SYSTEMATIC MARINE SPATIAL PLANNING AND MONITORING IN A DATA POOR ENVIRONMENT: A CASE STUDY OF ALGOA BAY, SOUTH AFRICA

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ABSTRACT

Globally the failure of traditional fisheries management approaches is evident through the increasing number of overexploited or depleted marine stocks. Past sectoral management has failed to address cumulative impacts of fisheries activities on ecosystem health. Ecosystem based approaches have been advocated as a viable alternative for sustainable management of marine ecosystems as they present a holistic and precautionary approach, which integrates management of multiple activities with that of maintaining ecological health. Although conceptually advanced, implementation has been poor due to the complexities of competing ecological and socio-economic management objectives. Marine spatial planning can facilitate the implementation of ecosystem based management as it is able to address the spatial heterogeneity of biological communities and anthropogenic activities. Ecosystem based management approaches aim to address the full range of anthropogenic drivers on the marine environment, including but not limited to fisheries, tourism, coastal development, and land and marine based pollution sources amongst others. Fisheries activities have a direct impact on the local marine environment and were therefore the focus of this study which forms a starting point for implementing ecosystem based management in Algoa Bay. It is envisaged that future research will build on this foundation and include additional anthropogenic drivers into the management and monitoring strategies developed in this study in order to achieve a truly holistic ecosystem approach to management in Algoa Bay.

Algoa Bay is situated centrally within the warm-temperate Agulhas bioregion along the east coast of South Africa and is the largest and best formed logarithmic spiral bay along this section of coastline. A large city, two commercial ports and several coastal settlements are located within Algoa Bay and a wide range of marine based activities occur within the area. A large section of the coastline is proclaimed as a National Park yet only two small offshore marine areas are formally protected. The development of a large marine protected area (MPA) adjoining the terrestrial section was proposed in the mid-1990s but a lack of adequate spatial data with which to quantify the fishery costs and conservation benefits led to wide scale public opposition and halted the declaration process.

The primary goal of this study was to obtain and analyse baseline data to understand spatial and temporal trends in the distribution and abundance of fish populations and fisheries activities in order to develop a spatial framework for marine conservation and management in a data limited situation using Algoa Bay, South Africa as a case study. Furthermore, it aims to contribute to the development of a monitoring framework to evaluate the success of implementation and the resultant changes in biological and socio-economic environments. This information will be used to re-initiate the stakeholder engagement process in the future.

A review was conducted of past research relevant to the current study area and spatial biophysical and fisheries data requirements were identified. Randomly stratified controlled angling and underwater visual census (UVC) surveys were used to assess reef fish community structure and the relative abundance and size of dominant species in seven selected reef study areas across Algoa Bay. Multivariate statistics and generalised linear models (GLMs) were employed to analyse the data from 453 controlled angling sites and 261 point counts. Reef associated fish community structure differed significantly across Algoa Bay and two main communities were distinguished, a sheltered western reef community in which non-reef dependent species were abundant and an exposed eastern reef community dominated by larger resident reef dependent species. Demersal fish communities over the trawlable grounds of Algoa Bay was assessed using research trawl survey data from 123 stations collected between 1986 and 2008. Multivariate analysis revealed that depth and location resulted in the greatest influence on community structure. Community structure was relatively stable in the long-term although differences were apparent as a result of changes in trawl gear configuration. Single species analyses indicated that depth and location influenced the relative biomass of 12 important fishery species, while depth, location and substrate type resulted in significant influences on mean length of these species. Trends in mean size indicated that the shallow sheltered western region of Algoa Bay is an important nursery area for several species.

Randomly stratified roving creel surveys were conducted to assess spatial and temporal patterns in recreational shore (including subsistence) fishing effort and catch, and were supplemented by aerial surveys conducted on an *ad hoc* basis. Analysis of data from 193 roving creel surveys revealed distinct spatial and temporal trends in recreational shore fishing effort, with effort aggregated around coastal access points and peaking over the main holiday period in summer. These patterns were confirmed through nine aerial surveys. Recreational shore fishery catch rate was highly variable and catch composition differed spatially. Skiboat club launch records were obtained and access point trailer counts were conducted to assess temporal effort trends in the recreational skiboat fishery. On-site access point interviews were conducted at launch sites during high use periods to obtain information on the spatial location of fishing sites, catch rate and catch composition. Launch records spanning a four-year period were obtained for the main recreational skiboat club based in Port Elizabeth, and 163 and 171 effort counts were conducted at two main beach launch sites respectively over a 12-month period. Recreational skiboat launching effort differed significantly between access points and seasons, being higher in the western region of Algoa Bay and during summer/autumn. Recreational skiboat effort was unevenly distributed between offshore fishing grounds with greatest effort occurring closer to access points on the western and eastern limits of the study area and limited effort occurring in the central less accessible regions of Algoa Bay. Catch rate and catch composition differed between launch sites. Spatial indices of recreational shore and skiboat effort were developed and integrated into a single spatial index of relative recreational importance.

Logbook, vessel monitoring system (VMS) and onboard observer data were obtained to assess the five commercial fisheries (chokka-squid jig, linefish, small pelagic purse seine, inshore demersal trawl and demersal shark longline fisheries) active within Algoa Bay. GLMs were used to assess long-term temporal and spatial trends in effort and catch rate in each sector. VMS and onboard observer data were used to assess spatial patterns in the distribution of fishing effort for each commercial sector. This revealed distinct spatial trends, which differed between sectors. Spatial indices of relative importance were developed for each sector and integrated into an index of relative commercial importance. Effort in the commercial linefishery displayed a general declining trend, while increases were apparent in the chokka-squid and small pelagic purse seine fisheries. No clear temporal trends in effort were apparent in the inshore demersal trawl and demersal shark long-line fisheries. Sectors targeted different species and catch rates were highly variable.

The decision support tool Marxan was used to conduct systematic conservation planning analyses and identify priority areas for conservation in Algoa Bay. Marxan uses a simulated annealing algorithm to identify minimum area requirements to achieve conservation targets while simultaneously minimising the cost of area selection using spatially explicit biophysical and socio-economic data. Spatial data for 36 conservation features was obtained from past studies, specialist workshops and research conducted during this study. These features were used for the identification of priority areas for conservation using quantitative targets for representation, which were aligned with national standards with refinement based on the local conservation importance of selected features and local management objectives. Eight planning scenarios were investigated, which assessed the influence of varying combinations of fisheries cost layers on the spatial selection of priority areas and the associated displacement of fishing effort. Incorporating spatially explicit fisheries data into the analyses resulted in considerable reduction in overall displacement of fishing effort compared to scenarios in which the spatial distribution of fishing effort was not considered *a priori*. The integrated recreational and commercial fisheries relative effort cost later resulted in least impact to fisheries with an overall displacement of effort of 14%, and a range from 4 to 18% across individual sectors while achieving all conservation targets. This resulted in a 23% reduction in the displacement of fisheries effort when compared to the scenario in which no spatially explicit cost data were considered. Assessment of the conservation value of no-take zones currently proposed for Algoa Bay as part of the Addo Elephant National Park (AENP) expansion revealed that despite an 11% displacement of fisheries effort, two of the conservation features were not present, and that insufficient quantities of 14 features were available to meet the desired target levels for representation. This indicates that refinement of the no-take zones currently proposed in light of the results of the systematic conservation planning outputs would result in considerable improvement in local conservation efforts at minimal additional cost to fisheries. Algoa Bay was also well sited to improving marine conservation on a regional scale thereby contributing to regional marine conservation objectives. Results from the systematic conservation planning analyses will be used to justify and support decisions regarding the establishment of new no-take zones in Algoa Bay as well as to facilitate stakeholder engagement.

A Pressure-State-Response monitoring framework was developed and indicators selected for evaluating future changes in ecosystem components. Baseline data were used to assess spatio-temporal variability, and the influence of other explanatory factors on selected indicators and power analyses were conducted to determine future sample size requirements in order to develop a spatially stratified and statistically robust monitoring framework. Available sources of fisheries data were assessed and recommendations made for improving data quality through verification using complementary data sources. Provisional target and limit reference points were recommended for each indicator to evaluate the implementation of future management measures and evaluate the performance on the state of resources and distribution and intensity of fishing effort. The proposed monitoring framework is based on five key steps, i) setting ecosystem objectives, ii) selecting indicators and defining reference points, iii) designing a sampling protocol, iv) delegating monitoring responsibilities, and v) evaluating the implementation of actions/recommendations arising from ongoing monitoring to complete the adaptive management cycle. This framework will assist in evaluating progress towards the overall management goals for Algoa Bay and through adaptive management will allow continual improvement as more knowledge on the response of the ecosystem to management measures becomes available. The approach described in this study can be applied to improve management of marine ecosystems in other areas where baseline data have previously been lacking.

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LIST OF ACRONYMS AND ABBREVIATIONS

ADCP	Acoustic doppler current profiler
AENP	Addo Elephant National Park
AGDS	Acoustic ground discrimination systems
AIC	Akaike Information Criterion
BLM	Boundary length modifier
BRUVs	Baited remote underwater video
CPUE	Catch per unit effort
CV	Coefficient of variation
DAFF	Department of Agriculture, Forestry and Fisheries
DEA	Department of Environmental Affairs, formerly DEAT (Department of Environmental Affairs and Tourism)
DEAET	Department of Economic Affairs, Environment and Tourism (provincial)
DIFS	Department of Ichthyology and Fisheries Science, Rhodes University
EAF	Ecosystem approach to fisheries
EBM	Ecosystem based management
FoC	Frequency of occurrence
GLM	Generalised linear model
GPS	Global positioning system
IDZ	Industrial development zone
IEI	Index of economic importance
IRCI	Index of relative commercial importance
IRRI	Index of relative recreational importance
ITI	Index of total importance
IUU	Illegal, unreported and unregulated fishing
LRP	Limit reference point
MLRA	Marine Living Resources Act
MLS	Minimum legal size
MPA	Marine protected area
MSL	Mean sea level
MSP	Marine spatial planning
NMLS	National marine linefish system
NMB	National Biodiversity Assessment
NMBMM	Nelson Mandela Bay Metropolitan Municipality
NPAES	National protected areas expansion strategy
ORV	Off-road vehicle
ROV	Remotely operated vehicle
PSR	Pressure-State-Response
PU	Planning unit
TAC	Total allowable catch
TAE	Total allowable effort
TRP	Target reference point
UVC	Underwater visual census
SAEON	South African Environmental Observation Network
SAIAB	South African Institute for Aquatic Biodiversity
SANHO	South African Navy Hydrographic Office
SANParks	South African National Parks
SCP	Systematic conservation planning
SPF	Species penalty factor
SPPSF	Small pelagic purse seine fishery
VMS	Vessel monitoring system

CHAPTER 1
GENERAL INTRODUCTION

1.1 Global state of marine ecosystems

Marine ecosystems are continually being placed under increasing pressures from the growing global population. This has led to the overexploitation of many of the world's fish stocks and overcapitalisation of commercial fishing fleets in order to meet the growing demands (Caddy 1999). Currently 53% of the global fish stocks are fully exploited, while 32% are overexploited or depleted, with only 15% moderately or underexploited (FAO 2010). In the past the impact of fisheries on fish stocks was perceived to be minimal due to the vast expanse of the oceans and the perceived size of the target populations, and 'pollution' and 'environmental change' were identified as the major causes of stock collapses (Pauly *et al.* 2005b). However, the direct and indirect impacts of fisheries activities on the stocks and marine ecosystems can no longer be ignored, with several studies documenting the role they have played in the degradation (Thrush *et al.* 1998; Blaber *et al.* 2000; Jackson *et al.* 2001; Thrush and Dayton 2002; Myers and Worm 2003; Pauly *et al.* 2005b; Roberts 2007; Worm *et al.* 2007; Lotze 2007; Myers *et al.* 2007; Crowder *et al.* 2008; Worm *et al.* 2009; Baum and Worm 2009; Ferretti *et al.* 2010).

Continued increasing fishing pressure has resulted in a 22% increase in the proportion of fish stocks considered overexploited, depleted or recovering and a concomitant decrease in the proportion of moderately exploited stocks over a 34-year period (FAO 2010). These depletions have been driven by a massive increase in global fishing effort due to the development of industrial fisheries during the 1950s and 1960s, which resulted in rapid growth in global harvests of capture fisheries (Pauly 2008; Allsopp *et al.* 2009). Initially stock declines were masked by spatial shifts in fishing effort away from areas of localised depletion (Pauly *et al.* 2002; Roberts 2007). This was facilitated through technological advances in vessels and fishing gear, which allowed fleets to target previously unfished resources further afield and at greater depths thereby maintaining high catch rates and contributing to the serial depletion of the resources (Pauly *et al.* 2002; Pauly *et al.* 2005b; Roberts 2007). Global annual harvest peaked in the late 1980s and by the late 1990s the few remaining habitats which had more recently been exposed to fishing pressure began showing signs of overexploitation (Pauly *et al.* 2005a). The global expansion of fisheries has led to the targeting and exploitation of stocks throughout their distributional range and across all life stages, with almost no natural refuges remaining in which they are afforded protection. This has led to increasing global realisation of the dire state of marine ecosystems, the role fisheries have played in this decline, and the urgent need for improved fisheries management strategies in order to manage ecosystems and stocks sustainably in the future.

1.2 Management of marine ecosystems

Fisheries management has traditionally focused on single species or sectoral approaches based on stock assessments which aim to maximise the fishery harvest with little regard for the impacts on non-target species and interactions between ecosystem components (Pikitch *et al.* 2004). The failure of these management approaches is now widely recognised (Curtin and Prellezo 2010) and there is growing awareness of the cumulative effects of the multitude of anthropogenic activities on marine and coastal ecosystems (UNEP 2006; Crain *et al.* 2008; Halpern *et al.* 2008a; Halpern *et al.* 2008b; Selkoe *et al.* 2009; Halpern *et al.* 2009; Douvere 2010). This realisation has led to a paradigm shift in the approach to fisheries management during the mid 1990s with a move away from using single species approaches in isolation towards adopting a holistic integrated approach incorporating social, economic and biological aspects. This has been referred to as ecosystem based management (EBM) or the ecosystem approach to fisheries (EAF) and requires that all the goods and services of the ecosystem are taken into consideration during assessment and management of marine ecosystems (FAO 2003). There has been increasing emphasis placed on addressing poor ecosystem health and declining fish stocks, and the shortfalls of traditional fisheries management through EBM (Pikitch *et al.* 2004; Powers and Monk 2010). EBM not only aims to address cumulative ecological effects, but also to identify and resolve conflicts between user groups (Ehler 2008). Although there is widespread consensus that EBM approaches are advantageous and should be adopted, few examples of successful EBM implementation exist. This is largely due to the complexities arising from competing objectives between sectors within a management area and a lack of direction and clear implementation guidelines (Crowder and Norse 2008; Douvere 2008).

1.3 Spatial planning in marine ecosystems

EBM requires that marine habitats, human activities and critical process areas are identified (Pikitch *et al.* 2004) and therefore requires implementing a spatially based approach for integrating management of habitats, user groups and activities through time. Spatial zoning is therefore a key aspect of EBM (Katsanevakis *et al.* 2011) and the field of marine spatial planning (MSP) has developed rapidly in recent years now playing a central role in integrating socio-economic, ecological and environmental aspects and objectives into management (Gilliland and Laffoley 2008). Quantitative conservation planning approaches have been developed and used successfully for zoning terrestrial ecosystems, yet until relatively recently had not been employed in the management of marine ecosystems. The lag in adopting systematic conservation planning (SCP) approaches for marine applications has been due to the absence of adequate spatial data and the difficulties in obtaining information on the distribution of sub-tidal habitats, species and community distributions and human activities (Spalding *et al.* 2007). The value of quantitative approaches for zoning human activities, development of conservation networks and design of protected areas is, however, widely recognised (Margules and Pressey 2000), and several recent marine assessments have adopted these approaches successfully, providing guidance for future projects. Strategic planning outcomes can be presented graphically and are therefore easy to communicate to stakeholders in a clear and non-technical manner. As such they provide a useful platform for public engagement which is central in the EBM process. SCP has been used successfully in the design of marine protected areas and reserve networks in California (Klein *et al.* 2008a; Klein *et al.* 2010), the Gulf of Mexico (Gutiérrez-Moreno *et al.* 2008), the Caribbean (Agostini *et al.* 2010), Australia (Fernandes *et al.* 2005) and in the Southern Ocean (Lombard *et al.* 2007). These approaches have also been used to evaluate the representivity of existing MPA networks within South Africa (Clark and Lombard 2007) and elsewhere (Stewart *et al.* 2003; Stewart and Possingham 2005).

Marine Protected Areas (MPAs) have become an integral component in the precautionary management of fisheries (Browman and Stergion 2004) particularly in situations where data and knowledge is limited. Historically natural refuges existed due to inaccessible or distant waters which ensured that a proportion of large adult spawning fish were protected from the fishery (Pauly *et al.* 2002). However, the development and expansion of fisheries and rapid technological advancements has largely eliminated natural protection, and formal protection of adult breeding populations through the proclamation of MPAs has become crucial to sustain productivity and support adjacent fisheries through larval dispersal and spillover of adult individuals (Russ and Alcala 1996; Roberts *et al.* 2001; McGilliard and Hilborn 2008; Stobart *et al.* 2009; Cudney-Bueno *et al.* 2009; Pelc *et al.* 2010). Although the efficacy of MPAs has been questioned, there is mounting evidence as to the benefits of MPAs for fisheries management and the conservation of biodiversity (Mosquera *et al.* 2000; Lester *et al.* 2009). In order to ensure that the benefits of MPAs are realised they need to be appropriately sized and located to include a range of representative habitats. This can be achieved through the design of a reserve network linking habitats over a broad geographical scale. Furthermore, protection of the resources needs to be ensured through appropriate management and enforcement. Currently slightly over 1.1% of the world's oceans are represented within MPAs (Toropova *et al.* 2010). Although the level of protection afforded to marine ecosystems within exclusive economic zones (200nm of coastlines) increases to 2.9%, and that of continental shelf areas is 4.3% (Toropova *et al.* 2010) there remains an urgent need to increase the representation of marine ecosystems in MPA networks. MSP is a key tool which can be used in the design of MPA networks which are inclusive of habitats representative of the broader bioregion and taking cognisance of the range of human activities within an area.

1.4 Monitoring, evaluation and adaptive management

Adaptive management is an iterative process which considers data from ongoing monitoring programmes to continually improve management action so as to achieve the desired goals (Nichols and Williams 2006). It is a central component of MSP, SCP and EBM and requires ongoing evaluation against clearly defined management objectives in order to assess progress towards achieving the desired outcomes (Day 2008). Baseline data provides the reference point for long-term evaluation. Assessments are often compromised through inadequate historical data leading to the 'shifting baseline problem' in which the pristine state of the resources is forgotten and future comparisons are made against an altered state (Pauly 1995). It is often not possible to attain historical data on the 'original' state of resources prior to the effects of exploitation (Myers and Worm 2003; Roberts 2007); however, establishing reference or baseline conditions based on the best available information against which future changes can be evaluated in light of management strategies is essential and forms the basis for long-term monitoring programmes (Jennings 2005; Spellerberg 2005; Blanchard *et al.* 2010).

Monitoring the implementation of EBM approaches is required to ensure the overall objectives are met and allows for continual improvement (Curtin and Prellezo 2010). Fisheries dependent data are often used in temporal assessments of target resources. However, such data are affected by the changes in fishing gear, techniques, vessel capabilities and regulations (technology creep). These and other factors influence the catchability of target species, affecting abundance estimates, and are difficult to quantify and incorporate into analytical models. Furthermore, changes in regulations and market demands can influence catch rates, species composition and total harvest, compromising assessments of the state of resources. Effective long-term monitoring of resource state can therefore only be achieved through fisheries independent standardised protocols implemented by management or research organisations. However, these are often costly and difficult to implement due to the inherent natural variability in marine communities.

Baseline surveys and data can therefore be used to optimise the design of long-term marine monitoring protocols in order to minimise costs yet ensure sufficient statistical robustness to detect long-term temporal trends in parameters of interest. Indicators which can easily be monitored are often selected and used to represent a parameter of interest. The identification of measurable indicators and the implementation of a cost-effective monitoring programme that relates ecosystem objectives and reference points which trigger management actions have been major problems associated with EBM (Gislason *et al.* 2000). Causal relationships between indicators of state and drivers of change need to be known or established to evaluate future changes. The selection and evaluation of indicators (Caddy 2004; Jennings 2005; Shin *et al.* 2005; Methratta and Link 2006; Link *et al.* 2010; Blanchard *et al.* 2010; Shin and Shannon 2010; Shin *et al.* 2010b) and design considerations for standardised monitoring protocols (Thompson and Mapstone 2002; Willis *et al.* 2006; Beever 2006; Bennett 2007; Bennett *et al.* 2009) has been the focus of investigations in recent years. Within South Africa, little fisheries independent baseline information exists for many marine communities and dedicated baseline surveys are required prior to the establishment of long-term monitoring programmes.

1.5 Marine spatial planning and conservation in South Africa

The South African coastline is fairly well protected with approximately 23% designated as MPAs; however, only 9% is fully protected in no-take zones (Lombard *et al.* 2004). Past designation of MPAs was on an *ad hoc* basis with little strategic planning on a national level to ensure adequate representivity of habitats and biodiversity (Attwood *et al.* 1997). Five inshore bioregions have been defined along the South African coastline (Lombard *et al.* 2004) (Figure 1.1) and a detailed assessment of the MPA network within the warm temperate Agulhas Bioregion along the south-east coast of South Africa revealed that the representivity of habitats in the existing MPA network was poor and that expansion should focus on including habitat types inadequately represented (Clark and Lombard 2007). As part of the National Protected Area Expansion Strategy (NPAES), South Africa aims to maintain ecosystem processes through the development of MPA networks which integrate terrestrial, riverine, estuarine, inshore and offshore protected areas where possible (DEA 2010). This will not only allow for improved connectivity between ecosystems but also improved management and enforcement through already established protected area management capacity. Key areas for the expansion and development of new MPAs therefore exist adjacent to terrestrially managed protected areas. Systematic conservation planning (SCP) will play an increasingly important role in the future expansion of the MPA network in South Africa to ensure that MPAs complement each other in terms of the species and habitats they represent, both within and between bioregions.

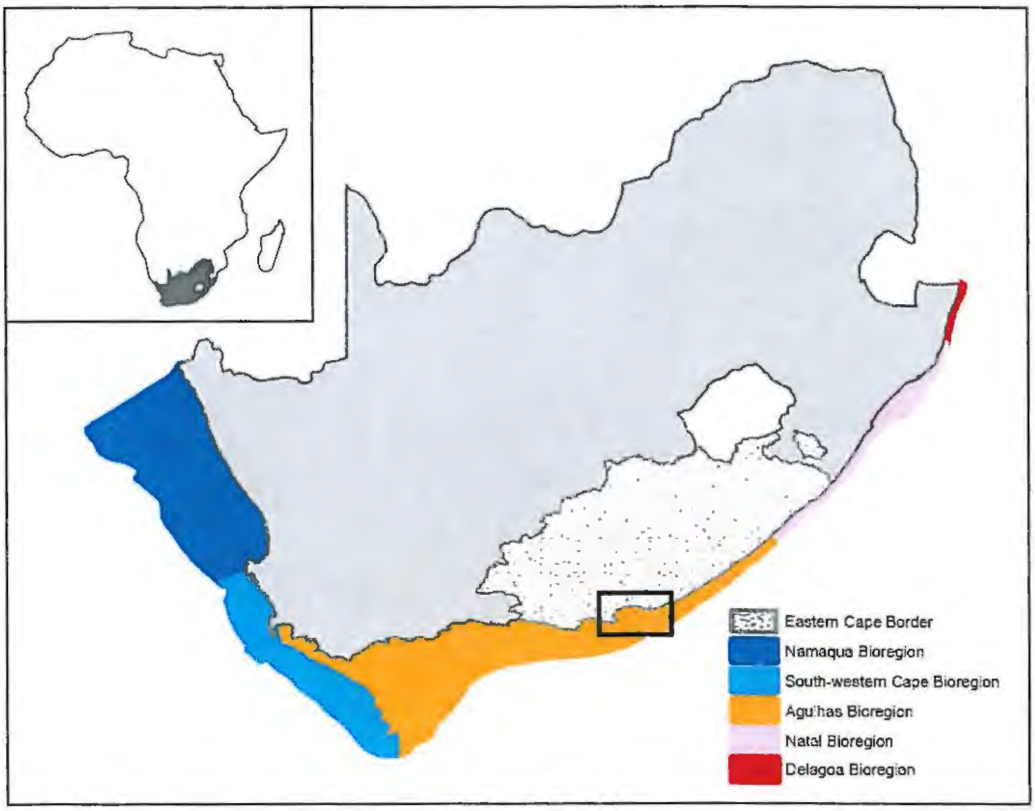


Figure 1.1. The five marine inshore bioregions of South Africa and the location of the study area (black box) within the Agulhas Bioregion and the Eastern Cape Province. The location of South Africa in the African context is illustrated in the top left insert. (Source: Lombard *et al.* 2004).

1.6 Case study: Algoa Bay

The Addo Elephant National Park (AENP) is located in close proximity to the city of Port Elizabeth in the Eastern Cape and extends from the inland mountain ranges to the coastline (Figure 1.2). The Park is managed by the South African National Parks (SANParks) whose marine responsibilities include management of large sections of the coastline between the Sundays and Bushmans river mouths, and two offshore island groups in Algoa Bay, which are included in the St Croix reserves and Bird Island MPA (Figure 1.2). The extent of formal protection currently afforded to the marine environment below the high watermark in Algoa Bay is limited to a 500m buffer around each of the St Croix islands, and the Bird Island MPA which includes approximately 70km^2 of inter- and subtidal habitat.

In order to enhance the level of formal protection afforded to marine subtidal habitats and biota in Algoa Bay, scientists and managers proposed a seaward expansion of the AENP boundaries to develop a large MPA adjacent to the existing terrestrial park. The first MPA boundary design was developed in the mid 1990s and was based on expert opinion (Kerley and Boshoff 1997). This was due to the absence of detailed marine biophysical data for subtidal habitats and the paucity of readily available and analysed fisheries data. A strategic environmental assessment was conducted as part of the AENP Expansion Project in 2000, which investigated both terrestrial and marine expansion options. During this process a further eight alternative footprint designs were prepared and evaluated by a working group comprised of 17 scientists and conservation managers (Newman and Klages 2001). This process resulted in the selection of the preferred option for the MPA footprint and was based on the expert judgement of the working group (Figure 1.2). This design option was favoured over others as it was a single contiguous area adjacent to the terrestrial park boundaries, had a simple boundary layout, and included both island groups, which were regarded as areas of high conservation importance within Algoa Bay. Although several other important biophysical features were considered to be represented within the proposed MPA footprint, the paucity of data (biophysical and fisheries) prevented any quantitative evaluation of the proposed reserve design. The absence of spatial biophysical data and limited knowledge of the extent of fisheries activities within the proposed footprint further complicated the designation of no-take zones (See Appendix 1 for MPA regulations), which were also designated based on the opinions of the working group. A public engagement process was initiated in order to foster stakeholder support and identify critical issues. The proposed design was poorly received by stakeholders due to the large spatial extent of the proposed footprint, inadequate consideration of socio-economic activities, and poor justification for the selection of the proposed no-take zones as these areas would have a significant direct impact on resources users. The resulting conflict between resource users and conservationists led to the engagement process being derailed, further reducing the political support for the establishment of a new MPA in Algoa Bay by the national regulatory authorities. The proposed MPA boundary as outlined in Figure 1.2 has not been formally accepted or adopted, and this study aims to evaluate the conservation value and socio-economic costs associated with this proposed design using a quantitative approach, while simultaneously investigating alternative designs in the broader Algoa Bay region which may provide better options for conservation and minimise impacts to the fisheries activities locally.

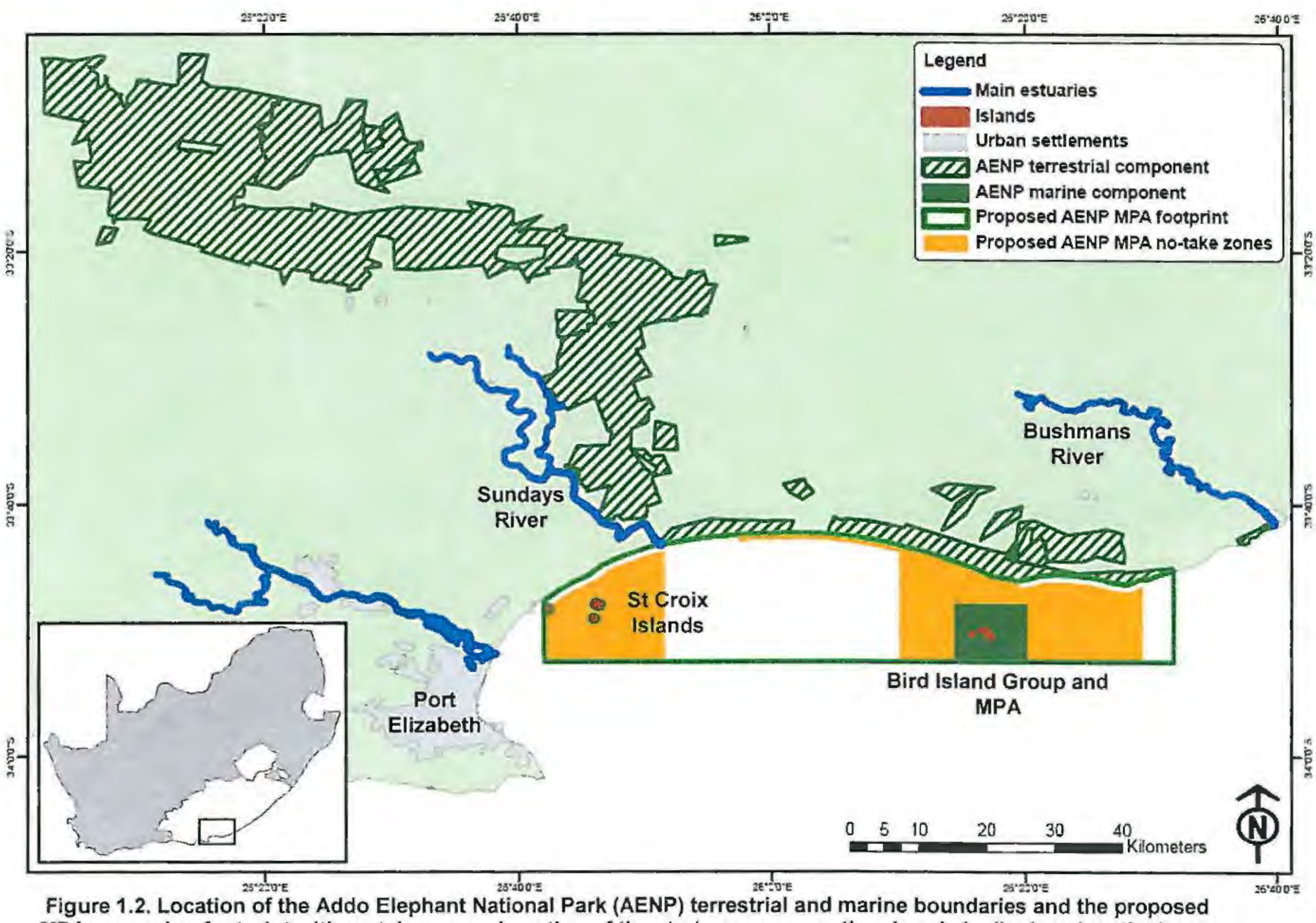


Figure 1.2. Location of the Addo Elephant National Park (AENP) terrestrial and marine boundaries and the proposed MPA expansion footprint with no-take zones. Location of the study area on a national scale is displayed on the insert map of South Africa (bottom left).

The failure of the initial MPA design proposal is considered to be due to the following critical shortcomings:

- Lack of defensible scientific reasoning and justification behind the proposed footprint and no-take zones
- Lack of alternative design scenarios presented to stakeholders during the engagement process
- Insufficient consideration of fisheries activities and their importance to local and regional economies
- Unclear decision process for evaluating tradeoffs between conservation and socio-economic objectives

The paucity of spatially explicit biophysical data and readily available and analysed fisheries data was a major constraint in the previous design process and highlighted the need for detailed research to be conducted in order to better understand the spatial and temporal characteristics of the biophysical environment and the fisheries activities occurring within the region. This study was therefore commissioned to address the data gaps which previously halted the declaration process and improve the available knowledge on the Algoa Bay ecosystem, thereby allowing a quantifiable and transparent planning approach to be adopted for the design of the MPA and no-take zones. This process would allow benefits and impacts of different design scenarios to be quantified and alternative MPA design options to be compared and presented for stakeholder review. Furthermore, to successfully evaluate the long-term conservation benefits and impacts to fisheries, an understanding of the variability in key parameters in the ecosystem was required to design a statistically robust monitoring protocol. This research project therefore aimed to address these issues and provide a defensible and transparent process with which alternative no-take MPA design options could be prepared to support decision-making, and to provide a solid platform from which to re-initiate the stakeholder engagement process. Furthermore, a framework for evaluating the effectiveness of spatial management initiatives in achieving the management objectives was required.

1.7 Thesis structure

The goal of this study was to obtain and analyse baseline data to understand spatial and temporal trends in the distribution and abundance of fish populations and fisheries activities in Algoa Bay in order to develop a spatial framework for conservation and management related to these fisheries. Furthermore, it aims to develop a monitoring framework to evaluate implementation and the resultant changes in the biological and socio-economic environments. Chapters 2 to 6 provide the baseline biophysical and socio-economic data which are required to conduct SCP in Algoa Bay (Chapter 7) and to develop a framework for monitoring and evaluating long-term responses to the implementation of new spatial management measures (Chapter 8) (Figure 1.3). Chapter 9 summarises the findings of the study. The content of each chapter is outlined briefly below.

Chapter 2 – This chapter provides an assessment of the **meteorological and oceanographic conditions** which drive productivity within Algoa Bay through the analysis of data collected by continuous monitoring platforms in Algoa Bay. Furthermore, it synthesises the available information from past research projects in order to **contextualise the biophysical and socio-economic environments, identify gaps** in the current knowledge, and highlight outstanding data requirements.

Chapter 3 – Knowledge on the distribution of **reef habitats and the reef ichthyofaunal community** structure within Algoa Bay is limited. This chapter identifies dominant reef complexes within Algoa Bay and investigates the spatial and temporal patterns in reef linefish communities.

Chapter 4 – This chapter describes and investigates factors influencing the community composition of **demersal ichthyofauna over the trawlable grounds** in Algoa Bay using research trawl data.

Chapter 5 – **Recreational fisheries** contribute significantly to the overall harvest of marine fisheries. This chapter investigates the spatial and temporal distribution of recreational shore and skiboat linefishing effort to determine key factors driving recreational use. Furthermore it assesses the community structure of the catch and quantifies annual harvest.

Chapter 6 – **Commercial fisheries** are managed on a sectoral basis and assessments are usually conducted on a national scale with little fine scale spatial information available. This chapter investigates fine scale spatial and temporal distribution and trends in catch and effort of five commercial fishery sectors in Algoa Bay utilising various sources of data to verify catch returns.

Chapter 7 – **Systematic marine conservation planning** was employed to integrate biophysical (Chapters 2, 3 and 4) and fisheries (Chapters 5 and 6) data to identify priority areas for the development of no-take zones in Algoa Bay.

Chapter 8 – This chapter assesses the outcomes from the planning exercise and provides a preliminary framework for **future monitoring** based on a Pressure-State-Response model.

Chapter 9 – This chapter summarises the findings and provides recommendations for future research.

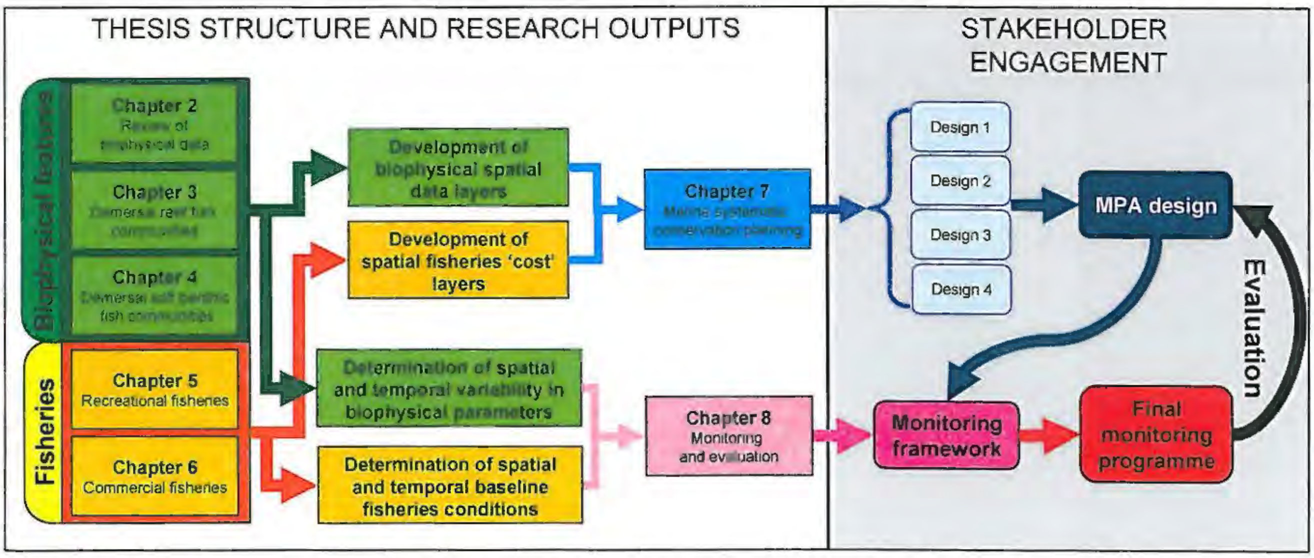


Figure 1.3. Schematic presentation of the thesis structure in context of the overall MPA design process showing links between key chapters and the stakeholder engagement process to follow.

CHAPTER 2

UNDERSTANDING KEY ATTRIBUTES OF THE STUDY AREA AND IDENTIFYING GAPS IN KNOWLEDGE

2.1 Introduction

Marine ecosystems are comprised of and influenced by physical, biological and socio-economic elements that are interdependent and have complex interactions with each other, which are often not readily apparent. Key drivers influence the structure and functioning of marine ecosystems thereby affecting the ecosystem services that they provide to human populations. Changes in the main driving forces or pressures on an ecosystem can result in cascading effects ultimately affecting the overall structure and health of the ecosystem (UNEP 2006). Human activities, in particular fisheries activities, through their cumulative and multiplicative impacts on the marine environment are the main drivers of ecosystem change. EBM aims to address the diverse requirements of society while maintaining healthy and functional ecosystems. In order to do so the key natural and anthropogenic elements which influence ecosystem productivity, structure and health need to be identified and understood. Future interpretation of ecosystem change is dependent on understanding these driving forces and will allow for continual improvement in management in order to achieve the overall objectives. This chapter synthesises the available information on Algoa Bay in order to better understand the biophysical and human dimensions and identify gaps in knowledge where additional information is required. Although this is largely based on review of available information from past studies, primary meteorological and oceanographic data were obtained and analysed to contribute to the current knowledge of Algoa Bay.

2.2 Data and methods

Meteorological data recorded by a weather station on Bird Island were obtained from the South African Weather Services for the period 1 March 2006 to 31 October 2010. Data received included hourly recording of air temperature, atmospheric pressure, wind direction and speed, rainfall and humidity, which were imported into an Access database and screened for outliers. Data were analysed in Excel and wind roses were prepared using WRPlot View. Oceanographic data recorded by continuous monitoring platforms located in Algoa Bay were obtained from the Elwandle Node of the South African Environmental Observation Network (SAEON). Current direction and velocity data were recorded by an acoustic doppler current profiler (ADCP) moored in approximately 30m of water to the south-east of Bird Island, which was configured to sample the water column in half meter bins at ten-minute intervals. The data spanned the period from December 2008 to May 2010 with intermittent gaps when the ADCP was removed for servicing. Water column temperature data were obtained from a SAEON thermister string moored in approximately 70m of water to the south-east of Bird Island with temperature recorders located at ten-meter intervals to within ten meters of the surface. Surface water temperature was obtained from an intertidal temperature recorder located at Woody Cape. Standard cosine-lanczos filters were applied to wind and current data using Ocean Data Tools prior to the preparation of vector plots. Surfer was used for graphical presentation of temperature and current speed and direction data. MODIS sea surface temperature satellite images were obtained for illustration of upwelling events.

2.3 Location of the study site

Algoa Bay is situated centrally within the warm-temperate Agulhas Bioregion in the province of the Eastern Cape of South Africa and is the largest and best formed logarithmic-spiral bay on the Cape south coast (Bremner 1983). For the purposes of this investigation the Algoa Bay study area is defined as extending from the 100m isobath off Cape Recife point (34°1'50"S, 25°42'20"E) in an easterly direction to the mouth of the Bushmans Estuary (33°41'40"S,26°39'45"E), including all of the marine environment on the shoreward side (Figure 2.1). This forms the core focus area of the study for the spatial planning analysis in Chapter 7, and is hereafter referred to as the 'Algoa Bay study area', or simply the 'study area'.

2.4 Physical characteristics of the study area

The coastline within the study area is approximately 150km in length consisting predominantly of sandy beaches (64%) interspersed with rocky outcrops (8%) and mixed rock and sand habitats (12%), with some stretches having been transformed through industrial development (16%) in the city of Port Elizabeth (Clark and Lombard 2007) (Figure 2.2). The sandy beaches are predominantly intermediate, with conditions typically between reflective (steep, course sand and low wave energy) and dissipative (flat, fine sand and high wave energy) states, and consist of well sorted fine to medium sized quartz sands (McLachlan *et al.* 1977; McLachlan *et al.* 1981a). Two prominent headlands lie to the east. The Woody Cape headland consists of calcareous sandstones being of Aeolian origin, while the Cape Padrone headland is comprised of quartzitic sandstone (Figure 2.1). The Alexandria Coastal Dune Field is situated along the northern shore of Algoa Bay and is the largest of its kind in South Africa, ranging from 2 to 3 km in width along approximately 50km of shoreline encompassing an area of approximately 120km^2 (Illenberger and Rust 1988; Watson *et al.* 1996) (Figure 2.2). The dunefield is a unique feature of Algoa Bay with transverse dunes ranging from 10-90m in height (Illenberger and Rust 1988). Seven estuaries of varying ecological characteristics and status occur within the study area (Figure 2.2). The Baakens and Papkuils estuaries are located within the city of Port Elizabeth and are canalised and in poor ecological condition (Whitfield 2000). The Coega Estuary was a temporary open/closed system which was in poor condition but has recently been transformed with the development of the deep water port of Coega. The Swartkops and Sundays estuaries are large permanently open systems considered in fair and good ecological condition respectively (Whitfield 2000) and are of high conservation importance (Turpie *et al.* 2002). The Boknes Estuary is situated in the eastern sector of the study area and is a temporary open/closed system in good condition, while the Bushmans Estuary, which forms the eastern border of the study area, is a large permanently open system in fair ecological condition (Whitfield 2000) and is also regarded as of high conservation value (Turpie *et al.* 2002).

Two island groups are located within Algoa Bay and are unique features along the South African east coast being the only islands between Cape Agulhas and Maputo in southern Mozambique. The Islands of the Cross (St Croix Islands) are situated within the western sector of Algoa Bay consisting of three separate outcrops, St Croix, Jahleel and Brenton islands, comprised of quartzitic Table Mountain Sandstone (Beckley and McLachlan 1979b; DEAET 1996) (Figure 2.1). St Croix is the largest of the three islands with an area of 0.12km^2 and rising to a height of 58 meters above MSL. The island supports the world's largest breeding colony of African penguins (*Spheniscus demersus*) (DEAET 1996; Pichegru *et al.* 2010). Although 17 plant species have been identified on the island, vegetation is limited to areas along the top ridge (DEAET 1996). Jahleel and Brenton islands are smaller unvegetated rocky outcrops. All islands drop steeply to the seafloor which is composed of consolidated sediments. The Islands of the Cross were proclaimed as South Africa's first island marine reserve in 1981 and included a marine component within a 300m radius around each island (DEAT 1981; DEAET 1996), which was extended to 500m in 1991.

Bird, Stag and Seal islands and a rock outcrop known as Black Rocks form the Bird Island Group situated approximately 10km offshore of the Woody Cape headland in the eastern sector of the study area (Figure 2.1). They are comprised of quartzitic Table Mountain Sandstone and the three islands, Bird, Stag and Seal, have a low relief and are sparsely vegetated with a mixture of shrubs. Thirty-three species of plants have been recorded; however, 20 are considered alien to the island group (DEAET 1996). Black Rocks is an unvegetated exposed rock formation. Bird Island supports the largest breeding colony of Cape gannets (*Morus capensis*) in the world (DEAET 1996), and together with Stag and Seal islands are important to several other birds including the African penguin and several species of migrant terns, including the endangered roseate tern (*Sterna dougallii*). Black Rocks is home to the eastern-most breeding colony of Cape fur seals (*Arctocephalus pusillus*) in the region (DEAET 1996). Guano harvesting was prolific on Bird Island from 1944 until the late 1980s but last occurred in 1989 (DEAET 1996; Urquhart and Klages 1996). The terrestrial component of the Bird Island Group was proclaimed a Provincial Nature Reserve in 1987 (ECPB 1999), and the Bird Island MPA was proclaimed in June 2004 (DEAT 2004). Both island groups were incorporated into the Addo Elephant National Park in 2005, with management of these areas becoming the responsibility of SANParks.

The sub-tidal environment of Algoa Bay is dominated by soft sediments, particularly course sands which are interspersed with fine silts and clays, with gravel beds limited to the rocky outcrops and island surrounds (Bremner 1978; Bremner 1991a) (Figure 2.2). The majority of the bay is between the 20- and 50-meter isobaths, with a maximum depth of 73m across its mouth (Harris 1978).

Although detailed soft sediment characterisation has been conducted within Algoa Bay (Bremner 1991b; Bremner 1991c; Illenberger 1992), reef complexes have been poorly charted (Newman and Klages 2001). Few sidescan surveys have been conducted in Algoa Bay, and are limited to the Cape Recife area (Buxton 1987), or have been conducted by private companies for prospecting purposes and the data are not publically available. Periodic surveys are conducted by the National Ports Authority in the shipping lanes and the dredge spoil dumping areas; however, these are monitoring surveys conducted over known soft benthic substrates and are limited to a small spatial area. Studies investigating sub-tidal soft benthic communities in Algoa Bay have indicated the presence of small isolated reef complexes in many areas of Algoa Bay (McLachlan *et al.* 1977; Cockcroft and Tomalin 1987). However, these have not been charted and it is likely that they are frequently inundated as a result of large scale sand movements typical of sandy beach ecosystems. The location of reef complexes is therefore limited to knowledge gained through the experience of commercial demersal trawlers, linefishermen and recreational skiboat anglers. The absence of detailed reef maps for Algoa Bay is a major limitation for future planning and management of the biological resources.

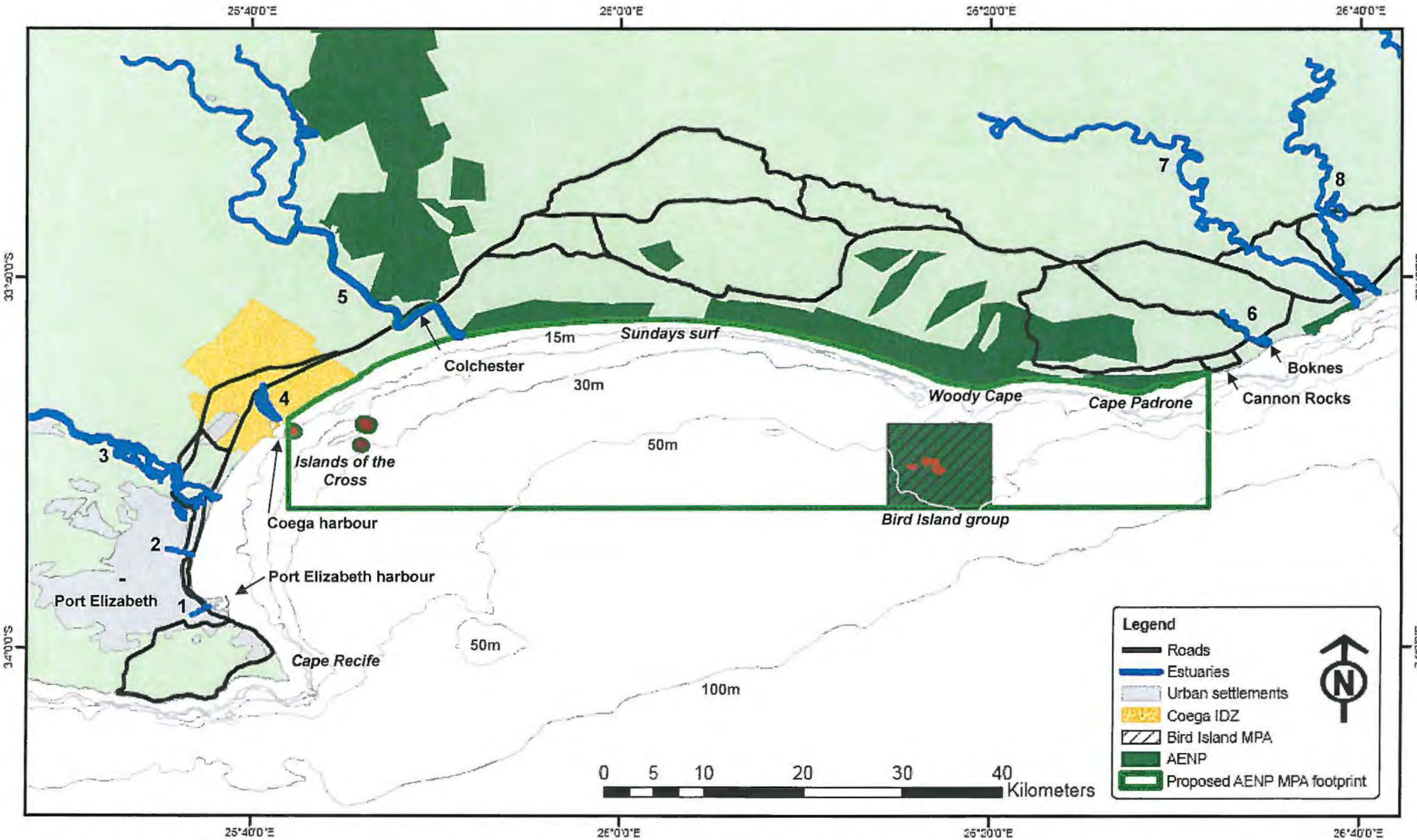


Figure 2.1. Key features of Algoa Bay (1=Baakens Canal; 2=Papkuils Canal; 3=Swartkops Estuary; 4=Coega Estuary; 5=Sundays Estuary; 6=Boknes Estuary; 7= Bushmans Estuary; 8=Kariega Estuary).

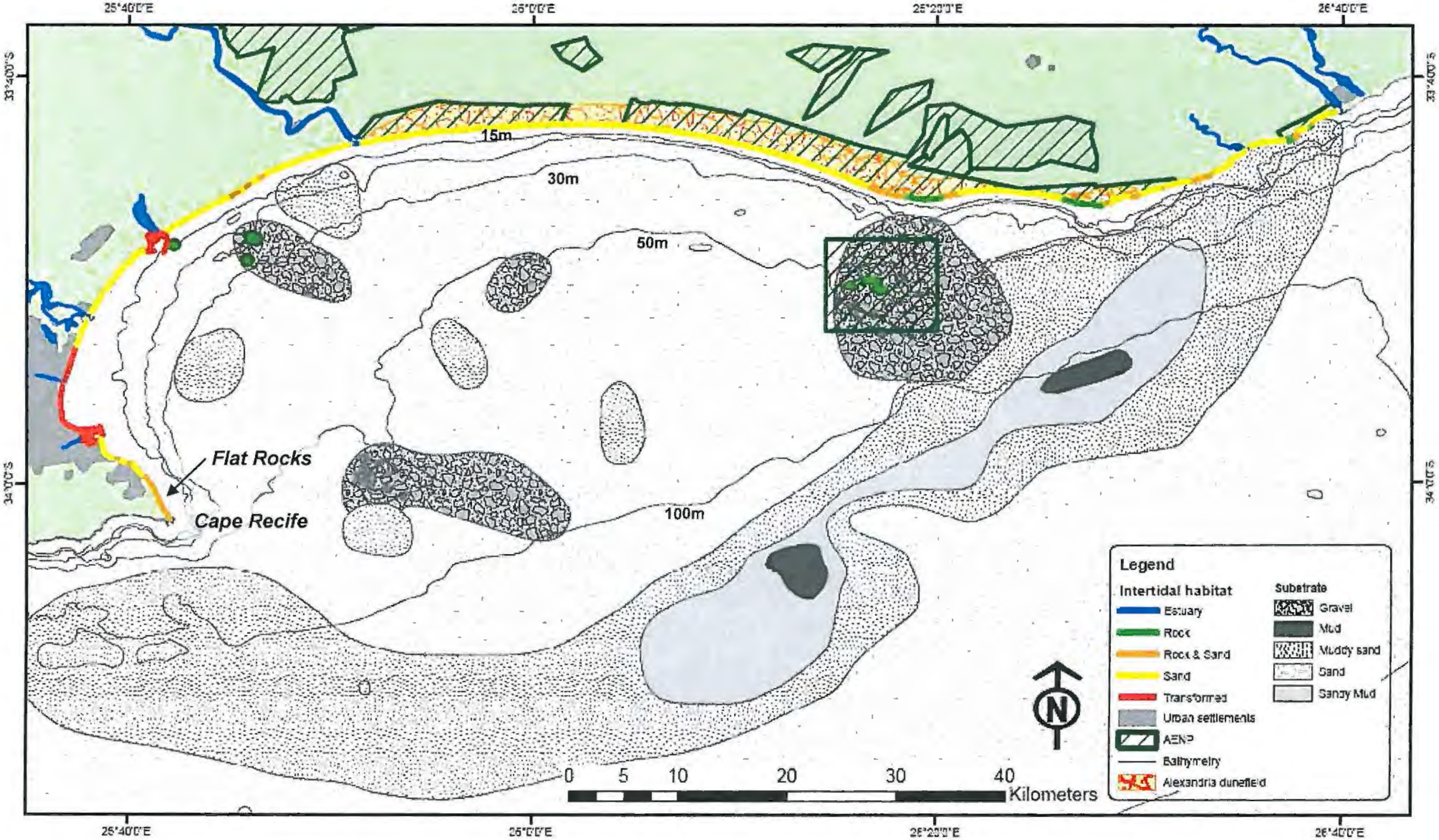
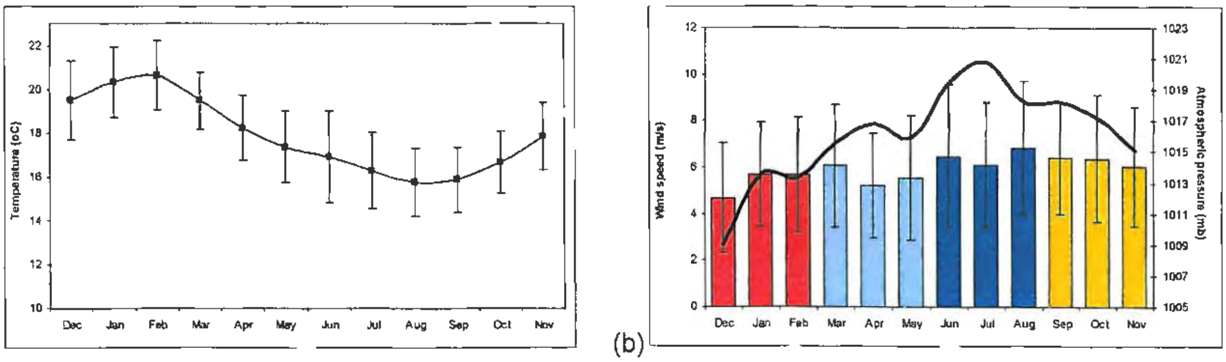


Figure 2.2. Coastal habitat characteristics of Algoa Bay. (Data sources: Intertidal habitat – Clark and Lombard 2007; subtidal substrate – Bremner 1978).

2.4.1 Meteorology

Mean summer air temperatures in Algoa Bay typically range from 12.9 - 25.5°C (Lubke and de Moor 1998). Mean daily air temperature recorded by the Bird Island weather station over the study period was highest between December and March, peaking at 20.6°C in February. Lower mean daily temperatures occurred between July and September, with lowest mean of 15.8°C in August (Figure 2.3). The minimum temperature of 5.8°C was recorded in October with a maximum of 42°C in December. Atmospheric pressure showed a marked increase over the winter months, peaking in July at 1020 mb, while lower pressures were prevalent over the summer months from December to February.

(a) 

(b)

Figure 2.3. (a) Mean monthly air temperature, (b) mean monthly wind speed (bars) and atmospheric pressure (line). Red bars=summer; light blue=autumn; dark blue=winter; orange=spring. Data from the Bird Island weather station (March 2006-October 2010).

Seasonal trends in wind patterns have been reported in Algoa Bay with strong south-westerly winds dominating in winter and easterlies increasing in the summer months (Beckley and McLachlan 1979b; Talbot and Bate 1987a; Schumann and Martin 1991; Roberts 2010). Generally winds tend to be strongest during the latter part of the year peaking during October and November with weakest winds in May and June (Schumann *et al.* 2005). However, high temporal and spatial variation in both wind speed and direction has been reported (Schumann *et al.* 1991). Mean wind speed recorded by the Bird Island weather station during the study period varied monthly, with weakest winds occurring in December (4.7m/s) and April (5.2m/s) and strongest winds in August (6.8m/s) and June (6.5m/s) (Figure 2.3b). Westerly winds occurred throughout the year but dominated during winter, with the easterly component increasing during spring and summer (Figure 2.4).

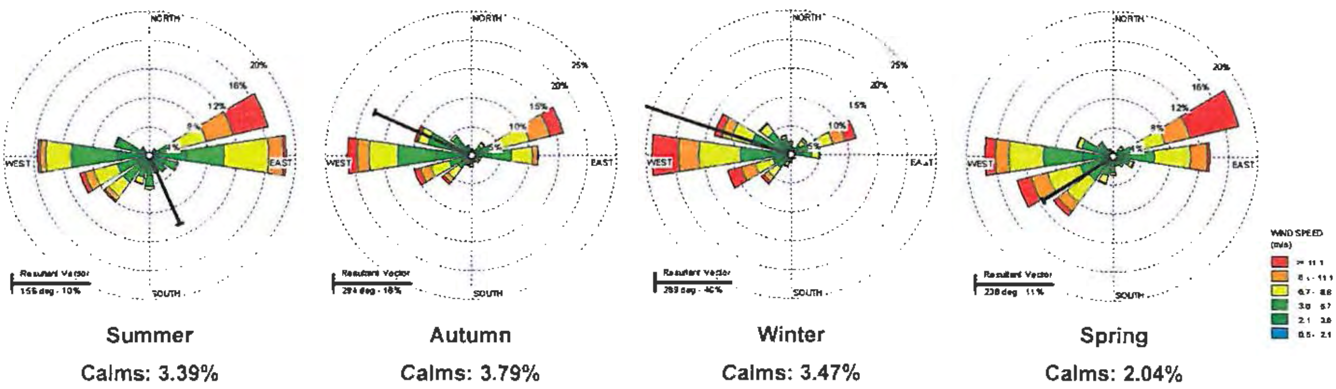


Figure 2.4. Seasonal wind directions and speeds from the Bird Island weather station (March 2006-October 2010) (calms indicates the proportion of time when no wind was detected).

2.4.2 Oceanography

Oceanographic conditions may vary over small spatial or temporal scales. Improved knowledge of these conditions and the factors which drive them contributes to understanding the responses of biological communities allowing improved interpretation of spatial and temporal variability in community structure and abundance for consideration in spatial planning and monitoring.

(a) Tides and swell

Tides within Algoa Bay are semi-diurnal with a mean spring amplitude of 1.6m and a maximum of 2.1m (Talbot and Bate 1987a). Freshwater inflow from rivers draining into Algoa Bay is negligible and has little influence on the local oceanographic conditions and salinity is relatively consistent throughout the bay (Schumann *et al.* 2005). Oceanic swell is bimodal being strongly influenced by the prevalent south-westerly winds during winter, with increasing occurrence of easterly winds in summer (Russouw 1984; Talbot and Bate 1987b; Schumann *et al.* 2005).

Two main driving forces influence the local oceanic conditions within Algoa Bay, namely the Agulhas Current acting on a large scale, and the predominant winds which drive the nearshore processes.

(b) Influence of the Agulhas Current

The Agulhas Current is a major western boundary current (Veronis 1973) which follows the shelf break of the Agulhas Bank along the 200m isobath (Lutjeharms 1981; Gründlingh 1983; Lutjeharms *et al.* 2000) where it transports warmer subtropical water in a south-westerly direction. In the Algoa Bay region the Agulhas Current runs approximately 80km offshore with the inshore shelf water being considerably cooler (Pearce 1977). Upwelling may occur along the inshore edge of strong boundary currents in regions where the shelf edge widens (Gill and Schumann 1979; Lutjeharms *et al.* 2000). The shelf edge begins to widen at Port Alfred, approximately 80km east of Algoa Bay (Figure 2.5), resulting in the movement of cooler water from deeper origin onto the shelf (Schumann 1987) which is expressed as a regular upwelling cell in the Port Alfred area (Lutjeharms *et al.* 2000; Lutjeharms 2007). This cold nutrient rich water moves southwards over the Agulhas Bank in the form of a cold ridge which follows the 100m isobath (Swart and Largier 1987). Intrusions of warm Agulhas Current water into Algoa Bay may also occur through meanders forming along the inshore boundary where the shelf widens (Figure 2.5) (Schumann *et al.* 2005; Roberts 2010). The meanders grow as they progress downstream creating shear edge features with warm water plumes and cyclonic eddies (Goschen and Schumann 1988; Lutjeharms *et al.* 2000). Westerly winds have been shown to push plumes of warm Agulhas Current water onto the shelf and into Algoa Bay at times (Schumann 1987; Goschen and Schumann 1988; Goschen and Schumann 1994) raising sea surface temperatures to 20-22^{o}C (Beckley 1983).

(c) Water column structure in Algoa Bay

Data from underwater temperature recorders (UTRs) located in 30 and 80m of water off the Woody Cape headland and Bird Island, respectively, were used to investigate the water column structure over twelve months during the study period to determine seasonal trends which may influence biological communities. The presence of a strong thermocline during summer is apparent with cooler water (<14°C) present from December to March at depths below 40m, with occasional cold water intrusions above 30m in depth recorded at both the inshore and offshore UTR strings (Figure 2.6). Surface waters, however, reached highest temperatures during January and February, despite the cold water intrusions suggesting movement of warm Agulhas Current water into the surface layers of the bay. Water temperature was more uniform throughout the water column during the winter and spring months. Local wind conditions have been reported to play a significant role in the generation of currents and temperature profiles within the inshore region of Algoa Bay (Harris 1978; Beckley 1983; Lutjeharms *et al.* 1989) and can lead to the expression of cold upwelled water in the surface layers (Rouault *et al.* 1995). Data from the Bird Island weather station suggests that the frequency of occurrence of easterly and westerly wind components is approximately equal for most of the year, except over winter when the westerly component increases in frequency and strength (Figure 2.6). Stronger westerly winds reduce the frequency of upwelling events and cause greater mixing of the water column leading to a more homogenous temperature structure (Schumann *et al.* 2005) which was evident in Algoa Bay during the study period (Figure 2.6).

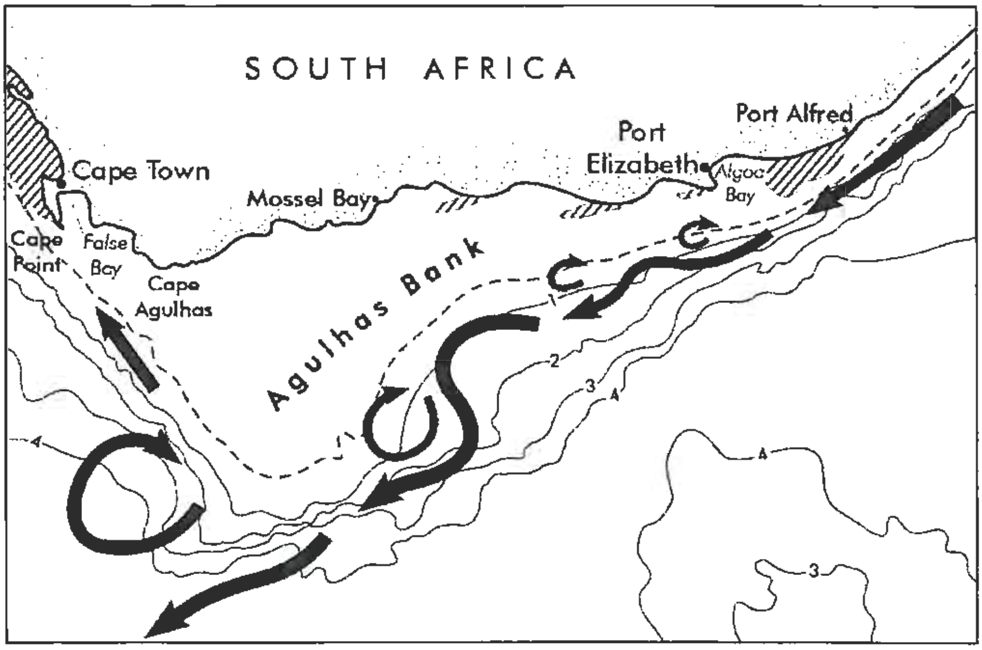


Figure 2.5. Schematic of the Agulhas Current (black arrows) indicating the formation of a meander and eddy features where the shelf begins to widen off Port Alfred. Grey shading indicates depths shallower than 1km, dashed line denotes the 200m isobath and hatching indicates areas of local upwelling (Source: Lutjeharms 2006).

Wind driven upwelling has been reported to be linked to strong easterly winds in summer (Schumann *et al.* 1988; Lutjeharms *et al.* 2000) which is evident through greater stratification, and stronger easterly winds recorded during spring and summer in the study area (Figure 2.6). This leads to the formation of intense thermoclines in the deeper waters (Schumann *et al.* 2005). North-easterly winds have been shown to result in the movement of cold upwelling waters in the nearshore and intertidal regions of Algoa Bay (Goschen *et al.* 2010). An example of such an event is displayed in Figure 2.7. During early March 2010 the sea surface temperatures were fairly uniform across Algoa Bay (Figure 2.7a); however, after two days of north easterly winds (Figure 2.7 top) the sea surface temperature began to decline off Woody Cape, which is evident through a small patch of cooler water visible in the satellite image (Figure

2.7b) and the change in intertidal water temperature measured at Woody Cape (Figure 2.7 bottom). Continued north-easterly winds led to a change in current direction which was most evident in the surface waters with current speed decreasing with depth, as well as the upward movement of cold water to the surface layers on 5 March 2010 (Figure 2.7c). Warm surface water trapped in the north-western region of Algoa Bay was pushed in an easterly direction along the coastline by a short period of westerly winds, resulting in a spike in surface water temperature at Woody Cape. However, near surface temperatures further offshore were not influenced by this movement of warm water, which was limited to the nearshore. This warm surface water, however, dissipated with cold water present in the surface layers for most of Algoa Bay on 8 March 2010. It is evident that the temperature of the deeper waters (50m and 70m) is consistently low and remains relatively stable throughout. The shallower water and in particular the surface and inshore intertidal waters are heavily influenced through wind induced upwelling with marked changes in water temperatures recorded. These patterns have also been observed in the adjacent Bay of St Francis (Sauer *et al.* 1991).

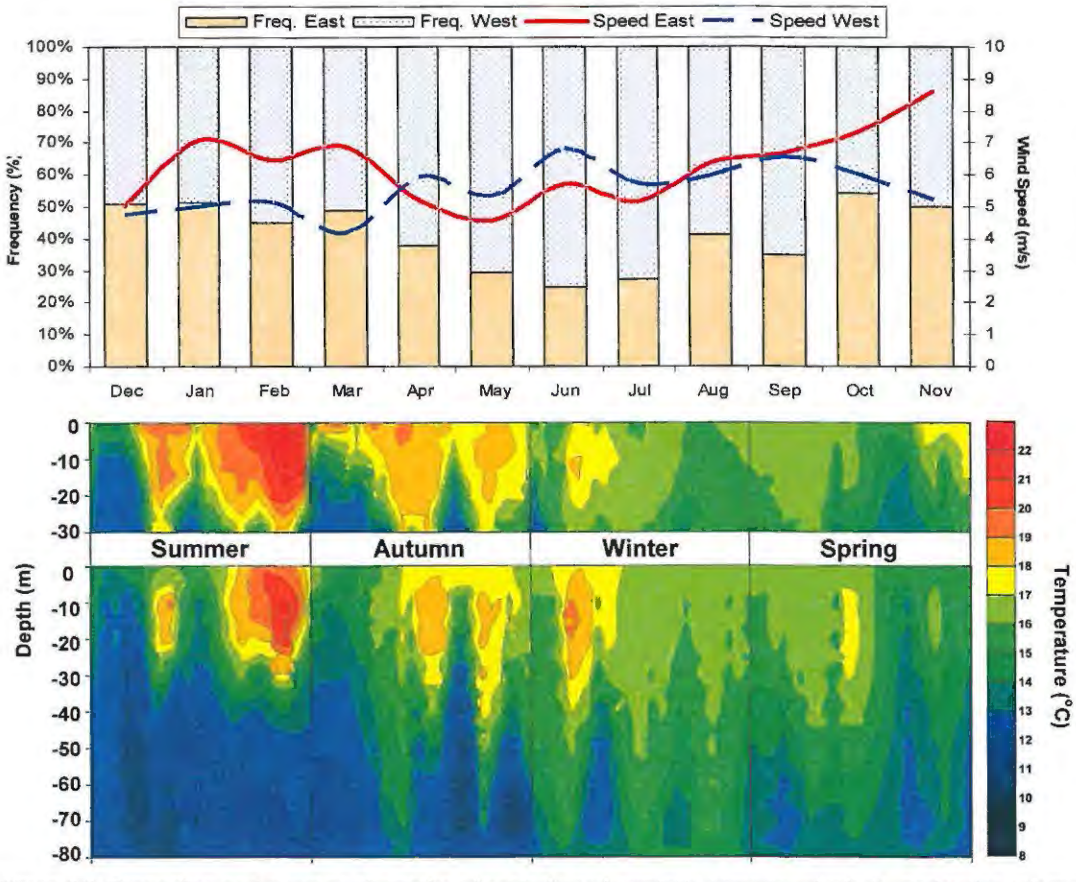


Figure 2.6. Proportion and average strengths of easterly and westerly component winds, and water column temperature profile from the 30m UTR string at Woody Cape and 80m UTR string at Bird Island (Data: November 2009-October 2010).

Periods of sustained easterly winds over summer can also result in upwelling of cold water to the west of Cape Recife (Schumann *et al.* 1988) which may be driven into Algoa Bay by westerly winds (Goschen and Schumann 1995) reducing sea surface temperature to between 11-13^{o}C (Roberts 2010). Average minimum water temperatures within Algoa Bay are in the order of 14-15^{o}C in winter, while maximum averages are in the range 20-22^{o}C in summer (Beckley 1983; Beckley 1988b; Schumann *et al.* 2005).

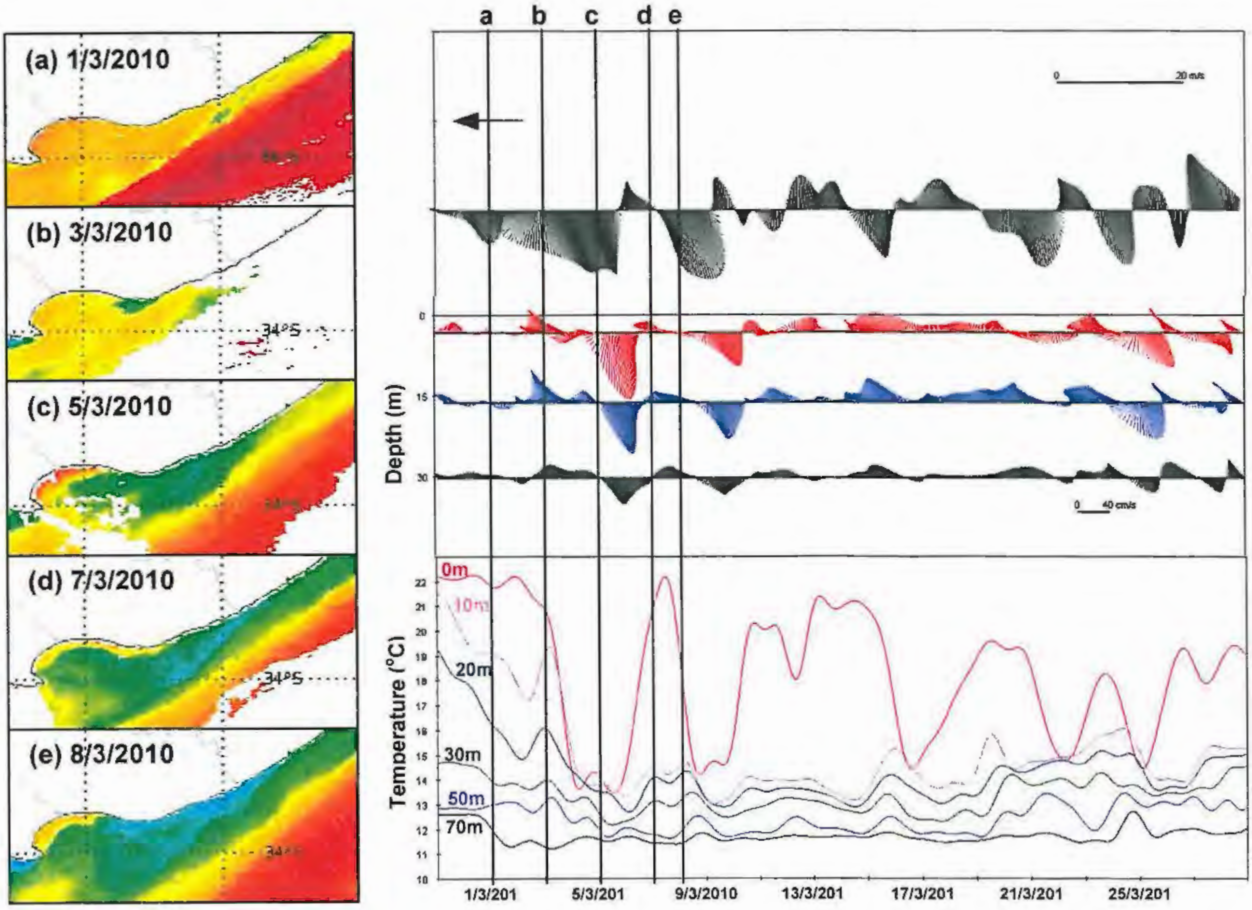


Figure 2.7. Satellite images of sea surface temperature from 1/3/2010-8/3/2010 (a-e) with wind vectors (top) (displayed in current direction), near surface and deeper water current vectors (middle), and temperature profiles (bottom) of the intertidal water at Woody Cape (0m) and sub-surface water temperatures from the deep water UTR off Bird Island.

(d) Current direction and velocity

Past studies in the western and central areas of Algoa Bay have documented surface currents flowing in predominantly north-eastward and south-westward directions (Harris 1978; Lutjeharms *et al.* 1986; Goschen and Schumann 1988), which coincides with the dominant wind directions (Schumann and Martin 1991; Schumann *et al.* 1991). Similarly, in the eastern sector of Algoa Bay alternating eastward-westward alongshore currents are common prevailing for up to three days at a time (Roberts 2010). Although surface currents flowed almost equally in an east-west direction, bottom currents tended to flow in a westerly direction. Several studies confirm that bottom currents are generally far slower than surface currents (Schumann *et al.* 2005; Pattrick 2007; Roberts 2010) and that they may not always flow in the same directions (Pattrick 2007; Roberts 2010). These differences are likely to be caused by the rapid influence of winds on surface waters which respond quickly to changes in wind directions, while deeper water currents are less influenced and take longer to respond (Roberts 2010). Currents recorded by an ADCP moored to the south-east of Bird Island confirmed the presence of higher velocity wind induced currents in the surface layers with reduced currents in deeper water (Figure 2.8). However, current direction was fairly consistent throughout the water column, and predominantly in a southerly and easterly direction (Figure 2.9). The duration and direction of wind forcing events therefore play an important role in establishing current patterns within Algoa Bay (Schumann *et al.* 2005), which may be further influenced by local bathymetric conditions and coastline features (Goschen and Schumann 1988; Schumann *et al.* 1988; Roberts 2010).

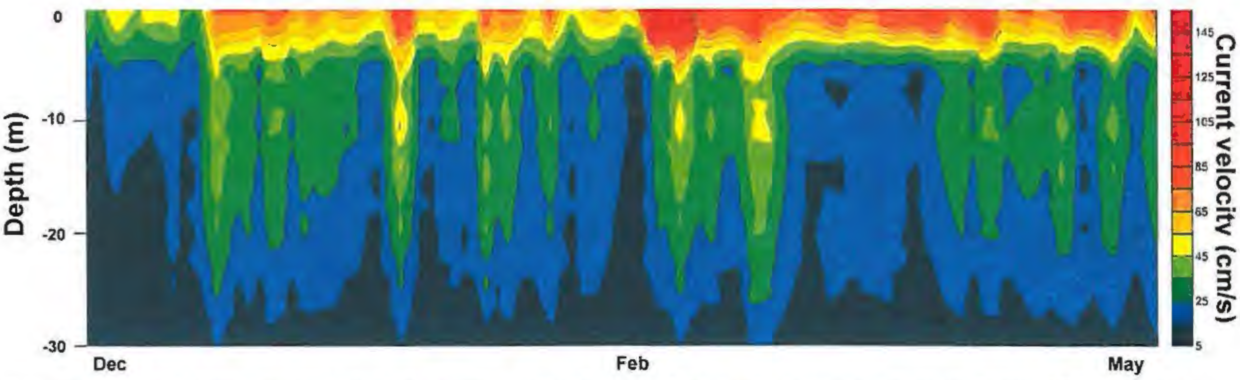


Figure 2.8. Short-term temporal trends in current velocity through the water column measured by an ADCP moored to the south-east of Bird Island (Data: December 2008-May 2009).

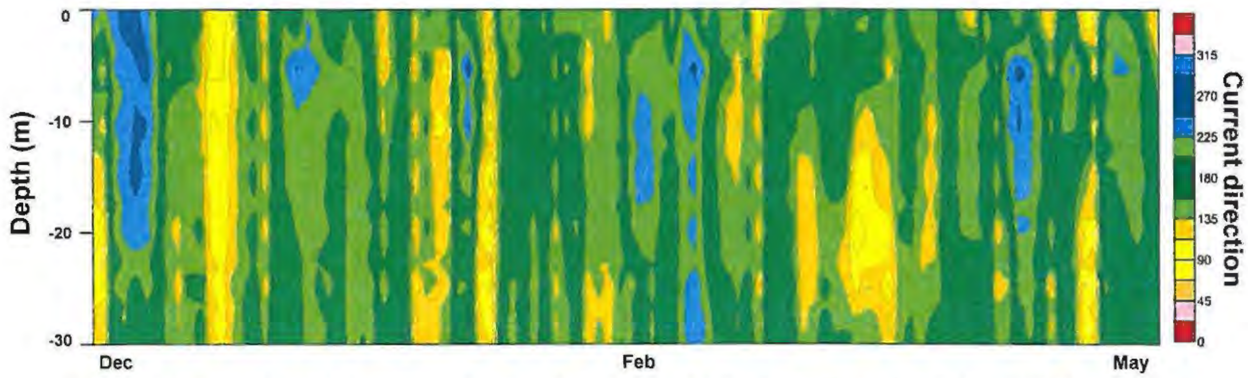


Figure 2.9. Temporal trends in current direction through the water column (Data: December 2008-May 2009).

Rip currents also play an important role in the flow of surface waters in the nearshore coastal regions and the formation and intensity of these currents is linked to swell size (Roberts 2010). To the east of Sundays Estuary they occur as semi-permanent features while in the western sector of Algoa Bay swell conditions have a more pronounced influence on the temporal and spatial expression of rip currents (Talbot and Bate 1987a). Westerly swell conditions typically increase the frequency and intensity of rips within Algoa Bay (Talbot and Bate 1987a) which may extend for 1-1.5km from the shoreline (Roberts 2010). However, they are generally restricted to the surface waters (Cook 1970) and are therefore unlikely to influence the movement of deeper waters significantly.

2.4.3 Summary of physical conditions

Oceanographic and meteorological forces are important drivers in Algoa Bay. Nutrient rich water derived from a combination of current and wind induced upwelling drives primary productivity. This primary production forms the foundation for the ecosystem supporting the higher trophic levels which in turn support numerous fishery activities. Oceanography also plays an important role in structuring biological communities, influencing the spatial and temporal distribution and abundance of species locally, and facilitating larval dispersal to adjacent areas. Understanding meteorological and oceanographic drivers in Algoa Bay is an important step towards understanding changes in secondary productivity expressed as changes in the state of biological communities. This can be achieved through ongoing monitoring as changes in biological communities are superimposed by other natural and anthropogenic drivers such as global climate change and changes in fishing pressure. Understanding such processes is important for spatial planning purposes.

2.5 Biological characteristics of Algoa Bay

2.5.1 Sandy beaches, the surf zone and nearshore

(a) Plankton

Sandy beaches and the associated surf zones comprise the dominant shoreline habitat within Algoa Bay and have been well studied, both qualitatively and quantitatively. Sandy beach ecosystems are heavily influenced by wave action (Branch and Branch 1983) and the predominant south-westerly swell direction has led to the formation of sheltered beaches with few rip currents in the western section of Algoa Bay, while in the eastern section the shoreline is more dynamic and many semi-permanent rip currents are present (Talbot and Bate 1987a). Due to the absence of macroalgae on the sandy shorelines surf diatoms (e.g. *Anaulus australis*) are the main source of primary production. Rip currents and the flow of nutrient rich groundwater into the surf zone drive the formation of dense aggregations of diatoms (McLachlan and Lewin 1981; Talbot and Bate 1988a; Campbell and Bate 1991; Campbell and Bate 1998) which undertake a diel vertical migration in order to maintain their position in the nutrient rich surf zone (Talbot and Bate 1986). By adhering to air bubbles during daylight they are maintained in the surface waters where they photosynthesise and their position in the surf zone is maintained by wave action and the physical barrier created by the surf zone (Talbot and Bate 1987b). However, at night they settle to the seafloor becoming epipsammic in order to prevent rip currents beneath the surface exporting them beyond the backline of the surf zone (Talbot and Bate 1988a; Talbot and Bate 1988b). Although this allows the formation of dense aggregations in surf zone (McLachlan and Lewin 1981; Talbot and Bate 1987b; Talbot and Bate 1988a), periodic losses may occur due to offshore winds and strong rip currents (Talbot and Bate 1988a) and approximately 11% of the primary productivity is lost to the nearshore (Campbell and Bate 1988) forming an important food source for filter feeding macrofauna (Branch and Branch 1983).

Several taxa comprise the zooplankton communities along the sandy beaches of Algoa Bay, including holo-, mero- and facultative planktonic forms (Romer 1986). Two mysids (*Gastrosaccus psammodytes* and *Mesopodopsis wooldridgei*) and one penaeid shrimp (*Macropetasma africanus*) form the bulk of the zooplankton biomass and occur in dense concentrations in the inshore waters (Wooldridge 1983). These taxa form an important trophic link connecting the surf zone phytoplankton blooms with higher trophic levels (Russow 1983; Lasiak 1983a). Zooplankton have developed behavioural mechanisms to ensure that they capitalise on the high food availability behind the surf zone after dusk (Wooldridge 1981; Wooldridge 1983; Cockcroft and McLachlan 1986; Webb and Wooldridge 1990).

(b) Microfauna, meiofauna and macrofauna

Microfaunal protozoans and bacteria play an important role processing particulate and dissolved organic matter as it filters through the sediments (McLachlan 1983). Meiofaunal composition in Algoa Bay is diverse, although nematodes and copepods dominate the community occupying different areas of the beach based on their physical and biological preferences (McLachlan 1977b; McLachlan 1983). Due to the instability of sandy beaches and the absence of macroalgal communities, no grazers are present and filter feeders, scavengers or predators comprise the macrofaunal communities (McLachlan 1977a; McLachlan 1977b; McLachlan 1983). Macrofaunal communities in the surf zone are impoverished and the intertidal region holds the greatest biomass of benthic fauna (McLachlan *et al.* 1981a) where the bivalve filter feeders, *Donax serra* and *Donax sordidus*, contribute to 95% of the total biomass (McLachlan 1977a; McLachlan 1983). The macrofauna are mobile, typically undertaking a tidal migration (McLachlan 1983), and contribute significantly to the higher trophic levels being a major source of food for numerous birds and fishes (Lasiak 1981; Lasiak 1983a; Lasiak 1983b). The sand prawn, *Callianassa kraussi*, occurs subtidally within the surf zone within the sheltered western sector of Algoa Bay where the absence of strong currents allows for the formation of semi-permanent burrows (Cockcroft and Tomalin 1987). It contributes significantly to the overall macrofaunal biomass in this region and is a major food source for benthic feeding fish (Cockcroft and Tomalin 1987).

(c) Ichthyofauna

The extensive surf zones in Algoa Bay are important habitats for larval fishes with estuarine dependent species dominating the communities (Watt-Pringle and Strydom 2003; Strydom and d'Hotman 2005). In contrast, larval fish communities in the nearshore (behind the surf zone) are dominated by marine species (Pattrick and Strydom 2008) with abundance showing clear temporal patterns peaking in spring and summer (Beckley 1985b; Pattrick and Strydom 2008). Nearshore larval communities are dominated by Gobiidae, but, the commercially important Engraulidae and Clupeidae families also contribute significantly to the community (Beckley 1986; Pattrick and Strydom 2008) and it has been suggested that Engraulidae spawn within Algoa Bay (Beckley 1986) and that Clupeidae spawn in close proximity to Algoa Bay (Pattrick and Strydom 2008). Larval fishes of the Sciaenid *Argyrosomus sp.* are also abundant in the nearshore, indicating preference for this habitat over the surf zone (Strydom and d'Hotman 2005; Pattrick and Strydom 2008) and estuaries (Beckley 1984b; Strydom *et al.* 2003). Based on the abundances of preflexion larvae it is thought that spawning of *Argyrosomus sp.* occurs locally within Algoa Bay (Pattrick and Strydom 2008) as was previously suggested (Smale *et al.* 1993).

Juveniles of several teleost species occur within the surf zone throughout the year, while larger individuals are less common, suggesting an offshore movement with growth and maturity (Lasiak 1983b; Lasiak 1984a; Lasiak 1986). The surf zone in Algoa Bay is therefore thought to serve as an important nursery area for several teleosts with the high productivity playing an important role in this function (Lasiak 1981; McLachlan *et al.* 1981a; Lasiak 1986). Differences in the juvenile species composition from the nearshore and surf zone of Algoa Bay and that of estuaries indicates the importance of the surf zone nursery areas for marine species (Lasiak 1981; Beckley 1984b).

Diverse fish communities representing several trophic levels occur along the sandy beaches of Algoa Bay (Lasiak 1981; McLachlan 1983; Lasiak 1983b; Lasiak 1984a; Romer 1990). However, communities are typically dominated by few species of planktivores or benthic feeders (Lasiak 1983b; Lasiak 1984b; Romer 1990). Filter feeding Mugilidae feed on the dense accumulations of diatoms within the surf zone, forming an important link between the primary producers and the top predators in the food web (Romer and McLachlan 1986). Benthic feeding teleosts include the blacktail (*Diplodus sargus capensis*), white and sand steenbras (*Lithognathus lithognathus* and *L. mormyrus*) spotted and olive (piggy) grunters (*Pomadasys commersonnii* and *P. olivaceum*) and the slender baardman

(*Umbrina robinsoni*) (Lasiak 1984b). However, the chondrichthyans, the lesser guitarfish (*Rhinobatos annulatus*) and the eagle ray (*Myliobatis aquila*), are the most important benthic predators in the surf zone of sandy beaches in Algoa Bay (Russow 1983). Piscivorous fish in the surf zone which prey on the smaller teleosts include the dusky kob (*Argyrosomus japonicus*), leervis (*Lichia amia*) and elf (*Pomatomus saltatrix*) (Lasiak 1984b). Several important fishery species occur along the sandy beaches, two being of particular socio-economic importance, namely the white steenbras and the dusky kob. These species are heavily sought after by recreational anglers and their stocks are currently depleted (Bennett 1993; Brouwer *et al.* 1997; Griffiths 1997a; Mann 2000).

2.5.2 Sub-tidal soft benthic communities

(a) Interstitial fauna

In comparison far less research has been conducted on the sub-tidal soft sediment communities in Algoa Bay. Early research indicated that the interstitial meiofauna was dominated by nematodes and herpacticoid copepods (McLachlan *et al.* 1977). In order to contribute to the understanding of the subtidal interstitial communities a research project has recently been established by the SAEON Elwandle Node to investigate spatial patterns in distribution and develop monitoring protocols for benthic invertebrate communities (S.Deysel *pers. comm.*).

(b) Ichthyofauna

Small-meshed trawl catches off the sandy beaches of Algoa Bay indicate that the nearshore ichthyofaunal species composition is dominated (by number) by teleosts including white seacatfish (*Galeichthys feliceps*) (28%), piggy (23%), kob (*Argyrosomus sp.*) (15%) and elf (11%) with elasmobranches only accounting for 4% of the catch (Wallace *et al.* 1984b). Ten estuarine associated species accounted for a large proportion of the catch by number (54%) (Wallace *et al.* 1984a), with juvenile white seacatfish, kob and elf prevalent indicating the nursery role of the nearshore coastal waters (Wallace *et al.* 1984a; Wallace *et al.* 1984b). Furthermore the abundance of kob in Algoa Bay was higher than in other areas along the Cape south coast indicating the importance of Algoa Bay for this species. The soft benthic feeders, white steenbras and spotted grunter, were notably absent from the nearshore (Wallace *et al.* 1984a) indicating their preference for the estuarine and surf zone environments in Algoa Bay.

The nearshore (<50m depth) regions of bays are important spawning and nesting areas for the chokka-squid, *Loligo reynaudi*, and several spawning locations and egg beds have been identified in Algoa Bay (Sauer *et al.* 1992; Sauer 1995). These sites are generally re-used over several spawning seasons (Sauer *et al.* 1992) and chokka-squid are heavily targeted in these areas by commercial vessels.

Considerable research has been conducted on the sandy beach ecosystems and the nearshore environments providing a good understanding of the food webs and flow of energy through the trophic levels. However, research on the offshore soft benthos in Algoa Bay has been limited due to the infrastructure and financial requirements of conducting such research, resulting in a large gap in the current knowledge of the Algoa Bay ecosystem.

2.5.3 Rocky intertidal shorelines

(a) Mega flora and fauna

Macroalgal communities in the littorinal zone on the Bird, St Croix and Jahleel rocky shorelines is dominated by purple laver (*Porphyra capensis*) interspersed with patches of sea lettuce (*Ulva rigida*) (Campbell 2009). The recent development for the Coega breakwater has, however, altered the composition and resulted in lower abundances of some algal species. Six other algal species are present in the upper and lower balanoid zones, while the cochlear zone is dominated by the limpet *Scutellastra cochlear* on all islands, with *Cheilosporum sagitattum* being the dominant algal species (Campbell 2009).

Limited mainland rocky shoreline habitat occurs within Algoa Bay and, unlike the sandy beaches, little research has been conducted locally. Studies have indicated that the intertidal communities on St Croix Island and the nearby mainland rocky shore are similar in composition (Beckley and McLachlan 1979b) and that zonation of macrofaunal and macroalgal communities is strongly influenced by wave exposure (Beckley and McLachlan 1979a; Beckley and McLachlan 1979b; McLachlan *et al.* 1981b). The shoreline between Cape Recife and Pollok Beach (inside Algoa Bay), known as Flat Rocks, is the only mainland rocky intertidal area within Algoa Bay (Figure 2.2) that has been studied in detail. It is sheltered from the dominant south-westerly swell and 13 macroalgal and 67 macrofaunal species have been identified, with zonation typical of other temperate South African rocky shores (McLachlan *et al.* 1981b; Bolton and Stegenga 1987; Beckley 1988a). Filter feeders, grazers and algae dominate the lower shore with carnivores and deposit feeders occurring throughout (McLachlan *et al.* 1981b). Macroalgal communities are more diverse on the exposed rocky shoreline of Cape Recife (23 species) compared to the sheltered shoreline inside the bay (13 species). A notable difference was the absence of the mussel, *Perna perna*, and several *Patella sp.* inside the bay which are common along exposed shorelines (McQuaid and Lindsay 2000), and is likely due to the sheltered nature of the shoreline inside Algoa Bay. Besides the two offshore island groups, the two rocky headlands, Woody Cape and Cape Padrone, represent the only other significant stretches of rocky shoreline within Algoa Bay, with only a few additional small isolated patches present (Figure 2.2). Although no intertidal research has been conducted on these two headlands, their communities are likely to be characterised by species common to exposed rocky shorelines due to their locality on the eastern side of Algoa Bay and the predominance of south-westerly swells.

(b) Ichthyofauna

The rocky shore intertidal zone is an important area for many ichthyofaunal species which may be resident in rock pools throughout the year, transient, using rock pools during specific periods of the year or their life cycle, or seasonal migrants washed down from the warm northern coastline by the Agulhas Current. Rock pool ichthyofaunal communities within Algoa Bay consist of up to 44 species from 20 families, with individuals typically being small, either small species, or juveniles of larger sparid species (Beckley 1985a; Beckley 1985c). Clinids (28%) and sparids (23%) dominate the intertidal communities and for some larger sparid species juveniles have only been observed in intertidal rock pools with none observed in estuaries, the surf zone or subtidal reefs, indicating the potential importance of rocky intertidal regions as nursery habitats. The juveniles of several economically important sparids occur within the intertidal area with some species showing high site fidelity (Watt-Pringle 2009).

2.5.4 Subtidal reef communities

(a) Macroalgae

Few detailed studies have been undertaken on the sub-tidal communities within Algoa Bay. Sub-tidal macroalgal assessments have been limited to the Bird Island area where 120 species and three distinct communities were identified (Anderson and Stegenga 1989). *Gelidium pteridifolium* dominates in the exposed regions, while *Plocamium corallorhiza*, *P. rigidium* and *Pachychaeta brachyarthra* are most abundant in the shallow waters, with a third deepwater community dominated by *Peysonnelia capensis* (Anderson and Stegenga 1989).

(b) Macro invertebrates

Virtually no information is available on the invertebrate community structure and diversity of subtidal reefs in Algoa Bay. The macrofauna is comprised of numerous soft-bodied sessile sponges, ascidians and bryozoans. However, due to the paucity of taxonomic information for these invertebrate groups and the limited subtidal surveys conducted both nationally and locally within Algoa Bay, the diversity and community structure is largely unknown (Parker-Nance 2003). Past visual assessments have documented shallow reefs to be dominated by seaweeds, with ascidians, octocorals, hydrozoans and sponges becoming more abundant on deeper reefs (Beckley and Buxton 1989).

The abalone, *Haliotis midae*, is a high-value species which has been commercially exploited in the Western Cape of South Africa since 1949 (Steinberg 2005). Although the perceived low abundances and patchy nature of abalone in the Eastern Cape prevented the development of a large scale commercial fishery (Tarr 2000), the Eastern Cape became a major source of abalone for the illegal markets in the mid 1990s (Raemaekers and Britz 2009). The growth and economic value of the illegal sector is highlighted by the shift in poaching activity from being largely shore-based and in close proximity to urban centres in the late 1990s to a highly organised boat-based poaching that involves travelling considerable distance at sea (Raemaekers and Britz 2009). The shallow reefs and rich macroalgal communities around Bird Island (Anderson and Stegenga 1989) support high densities of abalone. The inability of management authorities to control increasing illegal poaching activities from 2003 onwards led to the proclamation of the Bird Island MPA in June 2004 (DEAT 2004) in an attempt to halt illegal activities. However, despite the known high abundances of abalone and the high levels of illegal harvesting over the last decade, only two studies on the abalone resources in the area have been undertaken (SFRI 1986; Tarr and Anderson 1987). An initial assessment of abalone density at Bird Island indicated a range from 0.07 to 31.13 abalone.$10m^{-2}$ with an average density of 5.33±9.34 abalone.$10m^{-2}$. The high level of variation is due to the patchy distribution of abalone as a result of heterogeneous substrate and habitat types. Although the density was higher than initially suspected, the proportion of abalone above the then legal minimum size limit was low (38.5%) for a previously unexploited population, and the mean shell length was clearly smaller than that of populations in the Western Cape (SFRI 1986).

(c) Ichthyofauna

Few studies have investigated subtidal reef ichthyofaunal communities within Algoa Bay. Underwater visual censuses (UVC) conducted within the western region of Algoa Bay (extending to St Croix Island) investigated the non-cryptic ichthyofaunal composition at several reef sites, identifying 45 teleost and four elasmobranch species (Buxton 1987; Beckley and Buxton 1989). Sparids dominated the subtidal reef ichthyofauna with 22 species observed (Beckley and Buxton 1989). Several juvenile sparids were observed on subtidal reef complexes (Beckley and Buxton 1989) which have not been recorded in tide pools (Beckley 1985a; Beckley 1985c), estuaries (Beckley 1984b) or the surf zone (Lasiak 1981). Furthermore, juvenile abundance was typically greater on shallow inshore reefs suggesting that these areas may serve as an important nursery area for some sparid species (Buxton 1987).

Buxton (1987) identified three main community groups. The first shallow-water group consists of species with adults and juveniles occurring concurrently, including zebra (*Diplodus cervinus hottentotus*), blacktail, Mugilidae sp., bronze bream (*Pachymetopon grande*), Cape stumpnose (*Rhabdosargus holubi*), strepie (*Sarpa salpa*) and white musselcracker (*Sparadon durbanensis*) (Buxton 1987). The second group consists of juveniles of species occurring more commonly on deeper reefs which were present between December and April suggesting an inshore movement of recruits. This group was represented by fransmadam (*Boopsoidea inornata*), dageraad (*Chrysoblephus cristiceps*), roman (*Chrysoblephus laticeps*), black mussel cracker (*Cymatoceps nasutus*), blue hottentot (*Pachymetopon aeneum*), red steenbras (*Petrus rupestris*) and steentjie (*Spondyliosoma emarginatum*). The third group consists of species associated with sand patches including the white steenbras, sand steenbras, moony (*Monodactylus falciformis*), piggy and red tjor-tjor (*Pagellus natalensis*). The abundance of juvenile blacktail and strepie on subtidal reefs, tidal pools and the surf zone indicate that the shallow marine environment is a major nursery area for these species (Wallace *et al.* 1984a). A further four estuarine associated species not recorded by Beckley and Buxton (1989) are reported by Wallace *et al.* (1984a) from the nearshore reef areas, indicating the use of this area as a nursery area for some estuarine-associated species. No other offshore reef fisheries independent ichthyofaunal surveys have been conducted in Algoa Bay and further spatial information can only be obtained from commercial and recreational catch data, which have inherent biases.

2.5.5 Offshore pelagic environment

(a) Plankton

Very little research has been conducted on the neritic phytoplankton and zooplankton communities and the pelagic species which depend on them in the deeper waters of Algoa Bay. However, it has been recognised that the chlorophyll levels within localised areas around Algoa Bay are consistently more elevated than other regions along the east coast (Shannon *et al.* 1984; Probyn *et al.* 1994). This has been attributed to the persistent upwelling of cooler nutrient rich water resulting in higher levels of production in the Algoa Bay area than that of surrounding regions on the east coast (Shannon *et al.* 1984).

(b) Ichthyofauna

Phytoplankton production and the associated zooplankton communities are important food sources for pelagic fish species and largely determine their distribution patterns. Pelagic fish species which may occur in the Algoa Bay region include the sardine (also known as pilchard) (*Sardinops sagax)*, anchovy (*Engraulis encrasicolus)*, red-eye round herring (*Etrumeus whiteheadi*) and the horse mackerel (*Trachurus trachurus capensis*). Sardine is the most abundant and most important pelagic fishery species in the Algoa Bay region. The distribution and migration patterns of sardine are not well understood, but most of the biomass occurs along the south west coast and on the Agulhas Bank, but reaches as far east as Port Alfred (Coetzee *et al.* 2008). Prior to the late 1990s the bulk of the biomass was contained in the western region (van der Lingen *et al.* 2005); however, recent shifts in the distribution have been observed with an increasing proportion of the biomass occurring on the eastern Agulhas Bank from 1999 onwards (Coetzee *et al.* 2008). A similar eastwards shift in distribution has also been noted for anchovy (Roy *et al.* 2007) and it has been suggested that these trends may be linked to altered environmental conditions (van der Lingen *et al.* 2005; Roy *et al.* 2007) which have affected the availability of plankton, and therefore the distributional patterns of these species. Both species play an important role in the food web, providing the link between the primary producers and higher trophic levels.

The most common pelagic elasmobranchs found within Algoa Bay include the bronze whaler (*Carcharhinus brachyurus*), dusky shark (*Carcharhinus obscurus*) and the hammerheads (*Sphyrna sp.*), with the spinner shark (*Carcharhinus brevipinna*) being common in summer (Smale 1991; Heemstra and Heemstra 2004). The adults of these species usually occur in deeper waters while the juveniles are common in the coastal waters which serve as nursery areas (Smale 1991). They feed on a variety of pelagic and benthic fish and cephalopods. Although there is little published research on great white sharks (*Carcharodon carcharias*) (IUCN: Vulnerable) in Algoa Bay, the Bird Island Group is reported to be an area of high abundance (Klimley and Ainley 1996), particularly over winter months when seal pups are present. Research into their distribution and behavioural patterns around the Bird Island Group is currently being undertaken (M.L.Dicken *pers. comm.*).

(c) Avifauna

The islands within Algoa Bay and the Alexandria Dunefield provide important breeding and roosting sites for many seabirds. Two species are of particular importance, being apex predators in the marine ecosystem. These include the Cape gannet (IUCN: Vulnerable) for which Bird Island supports the largest gannetry worldwide, and the African penguin (IUCN: Endangered) for which St Croix is the largest breeding colony worldwide (Pichegru *et al.* 2010). The population of Cape gannets in Algoa Bay increased from 60 000 pairs between the mid 1990s and early 2000s to 90 000 pairs in 2005/6, while the numbers of African penguins declined from 20 000 during the 1990s to 10 000 between 2003-2006 (Crawford *et al.* 2008). Despite this decline St Croix remains the largest colony of African penguins in South Africa. The changes in population numbers is thought to be linked to changing environmental conditions resulting in large ecosystem shifts which affect the availability of food sources.

Both species are central-place foragers when breeding, implying that they must return to the nesting site to feed the chicks (Crawford *et al.* 2008). Sardine contributes significantly to the diets of both species (Crawford and Crous 1982; Crawford *et al.* 1983; Adams and Klages 1999; Pichegru and Ryan 2008; Crawford *et al.* 2008) and the eastward shift in the pelagic fish populations has had differing effects on each species, with the African penguin population declining and the Cape gannet population increasing (Crawford *et al.* 2008). This is thought to be linked to their foraging distances (approximately 40km for African penguins and 250km for Cape gannets) and it is likely that the distribution and density of pelagic stocks have not shifted eastwards sufficiently to benefit the African penguin populations (Crawford *et al.* 2008).

(d) Marine mammals

A breeding population of approximately 4 000 Cape fur seals occurs on Black Rocks within the Bird Island Group (DEAET 1996; Newman and Klages 2001). Annual counts have indicated large variation in the number of pups; however, no clear trend in the population status is apparent, and latest counts in 2004 indicated 423 pups in the colony (Kirkman *et al.* 2007). The diet of Cape fur seal in Algoa Bay consists primarily of the chokka-squid, which comprised 35%, followed by the shallow water hake (*Merluccius capensis*) 24%, horse mackerel 13% and the panga (*Pterogymnus laniarius*) 8% (Castley *et al.* 1991). Horse mackerel is the only major prey item which is also targeted by Cape gannets and African penguins and there is therefore little overlap in diets and minimal competition between fur seals and these species (Castley *et al.* 1991).

Bottlenose (*Tursiops truncates*), common (*Delphinus delphis*) and humpback (*Sousa chinensis*) (IUCN: Near threatened) dolphins occur within Algoa Bay. Common dolphins occur in low densities in the Eastern Cape throughout the year (Cockcroft and Peddemors 1990). Higher densities have been observed along the north-east coast during winter months only suggesting a seasonal migration which is likely to be linked to the migration of their major prey species, sardine, over this period (Cockcroft and Peddemors 1990). Humpback dolphins are more resident coastal dolphins which occur in shallow coastal waters and are reliant on the inshore resources (Karczmarski *et al.* 2008). Although it is evident that they occur in small groups and exhibit low population growth, little is known about the status of the different populations and it is thought that due to their inshore preference they are susceptible to anthropogenic impacts (Karczmarski *et al.* 2008). They have been reported to calf in Algoa Bay in summer (Karczmarski *et al.* 2008) and the inshore shallow reefs are their main foraging areas. Humpback dolphins show a preference for estuarine and reef associated teleost species, although cephalopods have also been recorded in their diet (Barros and Cockcroft 1991). Bottlenose dolphins are transient, occurring within 10km of the shoreline in Algoa Bay showing a preference for water depths of less than 30m (Ross *et al.* 1987). Little other research has been conducted on this species within Algoa Bay.

Four species of baleen whales are regularly observed within Algoa Bay, the southern right whale (*Eubalaena australis*), Bryde's whale (*Balaenoptera edeni*), minke whale (*Balaenoptera acutorostrata*) and the humpback whale (*Megaptera novaeangliae*). The southern right whale and humpback whale are migrants which move from the cold polar waters where they feed in summer, to warmer water along the

African east coast in winter where they calve. The minke whale is also a migrant species occurring more commonly offshore, while it is thought that the Bryde's whale may be resident in the region. An ongoing monitoring programme has been established in Algoa Bay to evaluate future trends in behaviour and abundances of all cetaceans (S.Plön *pers. comm.*)

2.5.6 Regional ecological importance of Algoa Bay

Algoa Bay has several biophysical characteristics which contribute to its ecological importance in the Agulhas Bioregion. The Port Alfred upwelling cell plays an important role bringing cool nutrient rich water to the surface which drives primary production in and around the Algoa Bay area. Surf zone diatoms, which are fed by nutrient rich waters originating from coastal aquifers, contribute significantly to the primary production in Algoa Bay. Prolific accumulations of surf zone diatoms have only been identified along twelve beaches in South Africa, with the Sundays surf being the longest and least impacted sandy shoreline where *A. australis* occurs (Newman and Klages 2001). The Alexandria Dunefield is a unique feature in Algoa Bay being the largest and least degraded dunefield in South Africa (Kerley and Boshoff 1997). Furthermore the rocky shoreline of Algoa Bay, although limited in extent, has diverse macroalgal assemblages due to the transitional nature between the south and east coast communities. The algal communities in this area have been poorly studied, yet 38% of recorded species are endemic to South Africa (Newman and Klages 2001). The long stretches of sandy beaches host diverse macrofaunal communities with 14 of the 25 major sandy beach species occurring along the Sundays surf beaches. The presence of numerous habitat types, including large open estuaries, the surf zone, intertidal and subtidal reefs and unconsolidated benthic substrates, within Algoa Bay and the interconnectivity between them contributes to the diverse ichthyofaunal communities present.

The Sundays Estuary is a large permanently open system which was ranked as 39^{th} out of 250 systems in South Africa in terms of its ecological importance (Turpie *et al.* 2002). The two island groups are unique along the east coast of South Africa. They support rich intertidal and subtidal diversity, and are important roosting and nesting sites for several bird species. In addition it is estimated that approximately 10% of the humpback dolphin population occurs within Algoa Bay (Newman and Klages 2001). This highlights the ecological importance of Algoa Bay and the need for a dedicated holistic management approach which takes into account the complex biophysical interactions to ensure future persistence and sustainable use of resources.

2.6 Socio-economic setting

2.6.1 General description

Three municipalities, the Nelson Mandela Bay Metropolitan Municipality (NMBMM), and the Sundays River and Ndlambe local municipalities, occur along the coastline within the study area. Port Elizabeth is situated in the NMBMM in the sheltered western corner of Algoa Bay (Figure 2.1) and is the third largest coastal city in South Africa with a population of 1.1 million residents (Anon 2010). The unemployment rate is over 35% and approximately 40% of the formal households are impoverished and access at least one social grant (Anon 2010). The Port Elizabeth Harbour is used by the agricultural and mining sectors as well as the motor industry, with approximately 1 260 ships handled per year. In an attempt to boost the regional economy, an Industrial Development Zone (IDZ) was developed on the northern shores of Algoa Bay with the concomitant Coega Port, a deep water port situated on its eastern border. The Coega Port became operational in October 2009 and is able to handle larger container ships which are unable to berth in the Port Elizabeth Harbour. Four coastal towns, Colchester, Cannon Rocks, Boknes and Bushmans River Mouth are situated within the study area (Figure 2.1). All towns are relatively small and are characterised by small resident populations with large influxes over peak summer holiday periods. The remainder of the coastline is either privately owned agricultural land, or proclaimed National Park falling within the Colchester and Woody Cape sections of the AENP (Figure 2.1). Few coastal access roads are present east of the Sundays Estuary and public access to large sections of the study area is therefore restricted.

Due to the surrounding urban and industrial development, storm water run-off and treated industrial and domestic effluent is discharged into the western sector of Algoa Bay. Furthermore, water quality is influenced by the high levels of shipping activity utilising the two ports. Two large marine outfalls discharge directly into Algoa Bay, namely the Cape Recife wastewater treatment works and the Fishwater Flats wastewater treatment works (Emmerson *et al.* 1983; Gardner *et al.* 1985; DWAF 2004). The Cape Recife outfall is situated on the western shores of Algoa Bay and discharges 5-10Ml of treated effluent per day (DWAF 2004). The Fishwater Flats outfall is considerably larger with approximately 112Ml discharged per day, which occurs via a 70m pipeline from the Brighton Beach Pier located to the west of the Swartkops Estuary (DWAF 2004). Dispersion and dilution is, however, reported to be rapid at both outfalls and near normal communities are present in close proximity to the outfalls (Watling and Emmerson 1981; Emmerson *et al.* 1983).

Industrial and domestic effluent is also discharged indirectly through outfalls into the Port Elizabeth Harbour, the Papkuils Canal and the Swartkops Estuary (Gardner *et al.* 1985; DWAF 2004). These all contribute to localised discharges of eutrophic freshwater which influences the distribution, abundance and composition of marine fauna and flora (Watling and Emmerson 1981; Emmerson *et al.* 1983). Industrial discharges into the Papkuils Canal has led to heavy metal contamination of the macrofauna affecting their distribution (Watling and Watling 1979; Watling and Emmerson 1981; Watling and Watling 1982; Watling and Watling 1983).

2.6.2 Fishery activities

(a) Recreational fisheries

The South African linefishery is a multi-user, multi-species fishery which targets approximately 200-250 species of which 95 contribute significantly to the commercial and recreational catch (DEAT 2005c). The recreational angling component of the linefishery is important in South Africa involving rock and surf, spear and offshore skiboat fishermen. Studies have been conducted on the recreational shore (Clarke and Buxton 1989) and skiboat fisheries (Smale and Buxton 1985) in Algoa Bay, with a national assessment conducted in the mid 1990s (Brouwer *et al.* 1997; Mann *et al.* 1997; Sauer *et al.* 1997; McGrath *et al.* 1997). These studies have indicated high and spatially heterogeneous levels of fishing effort, low catch rates and heavy targeting of certain species within Algoa Bay. Significant changes are likely to have occurred in the recreational fisheries as a result of new regulations and technological improvements which have allowed anglers to locate and target fish more successfully.

(b) Subsistence fisheries

Subsistence fishers have historically been overlooked and marginalised in South Africa (Branch 2002). However, with the proclamation of the Marine Living Resources Act (No. 18 in 1998) a Subsistence Fisheries Task Group was established to classify and identify subsistence fishing groups. An initial assessment estimated that approximately 29 233 people (<0.1% of the population at the time) could potentially be classified as subsistence fishers in South Africa, of whom 75% were located along the north-east coast of South Africa (Clark *et al.* 2002). Within the Algoa Bay region, several subsistence fishing applications have been received and processed over the last few years (A.Oosthuizen *pers. comm.*). However, no published information is currently available on the type of fishing employed, the spatial distribution of effort or the catch composition of subsistence anglers within the Algoa Bay region. A recent household survey of the surrounding communities indicated that the levels of subsistence fishing in Algoa Bay were low and that the effort was typically focused on the estuarine environments with limited subsistence fishing occurring along the coastline between Woody Cape and Cape Padrone (Chalmers, unpublished data).

(c) Commercial fisheries

Five commercial fisheries are known to operate within the Algoa Bay region (Sauer *et al.* 2003b). However, several fisheries are national fisheries with vessels operating along large areas of the South African coastline based on the distribution of the target stock. As a result published information usually pertains to the national fishery with little information available for localised coastal areas, which is a major limitation for local level planning and management.

(i) Commercial linefishery

The commercial linefishery is a boat-based fishery which targets approximately 40 teleost species with 20 being of major commercial importance to the sector (Lamberth and Joubert 1999). The fishery uses medium sized skiboats (5-8m in length) with two outboard engines which have an operational range of approximately 35km (Sauer *et al.* 2003b). They are trailerable vessels which are capable of launching from harbours, estuaries and beach launch sites. The national total allowable effort (TAE) for the 2008 fishing season was set at 455 vessels with a maximum crew of 3 450 (DEAT 2007d) with the number of rights holders in the Eastern Cape region (Cape Infanta to Port St Johns) being 80 and 87 in 2006 and 2007 respectively (DEAT 2007d). Of these only approximately 15 are known to operate regularly within Algoa Bay. Although national and regional level assessments of the commercial linefishery have been undertaken (Brouwer 1997; Brouwer and Buxton 2002), and past (Hecht and Tilney 1989) and recent (Donovan 2010) studies have investigated the dynamics of the Port Alfred commercial linefishery, no detailed studies have been conducted on commercial linefishery in Algoa Bay. The extent of commercial linefishing pressure in Algoa Bay is therefore a major gap in current knowledge.

(ii) Chokka-squid fishery

The chokka-squid is an economically important fishery species due to its high market value. The fishery is primarily based within the Eastern Cape due to the distribution of the stocks, which creates local employment and is a significant contributor to the local economy. Annual catches in the fishery are relatively low (3 500 – 11 000 tons) yet it is one of South Africa's most valuable fisheries due to high overseas market prices (Augustyn *et al.* 1992; Roel *et al.* 1998; Sauer *et al.* 2003b). The sector generates approximately ZAR[1]180 million per year (Sauer *et al.* 2003b), with between 85 and 90% of the resulting economic activity occurring within the Eastern Cape (Britz *et al.* 2001). The fleet consists of deck-boats with onboard freezer facilities which target chokka-squid over inshore spawning aggregations using jigs. Although considerable research has been conducted on the biology of the chokka-squid, few studies have investigated the spatial dynamics of the fishery in detail. Understanding the dynamics and importance of the chokka-squid fishery, the distribution of egg beds and seasonal occurrence of effort within Algoa Bay is important for the future management of the sector locally.

(iii) Small pelagic purse seine fishery

The small pelagic purse seine fishery (SPPSF) is a multi-species fishery and is South Africa's largest in terms of catch volume (DEAT 2005b) and second largest in terms of economic value (Augustyn *et al.* 1992). The fishery targets small short-lived species, with sardine (pilchard) and anchovy being the most important, accounting for 60-90% of the total landings (Fairweather *et al.* 2006b). There are approximately 100 vessels active within the fishery (Fairweather *et al.* 2006b) which supply eight fishmeal plants, six canning factories and 40 bait-packing facilities, which cumulatively provide employment for around 10 000 people (Sauer *et al.* 2003b). Only sardine occur in sufficient densities to warrant targeting along the Eastern Cape coast and in Algoa Bay, with the majority of national effort focused along the Western Cape coast. However, a few smaller vessels are based locally in Port Elizabeth and fish predominantly in Eastern Cape and Algoa Bay waters supplying the bait market. Concerns about the potential impact of the fishery on the ecosystem have recently been raised due to the declining trends in the African penguin populations in Algoa Bay. This led to the establishment of a pelagic MPA around St Croix Island (January 2009) where most of the Algoa Bay catch was previously landed (Fairweather *et al.* 2006b; Pichegru *et al.* 2009; Pichegru *et al.* 2010). Both Fairweather *et al.* (2006b) and Pichegru *et al.* (2009) provide assessments for the SPPSF nationally for different time periods which indicate the importance of Algoa Bay to the catch of sardine. Sardine is caught by a few smaller locally based vessels which do not travel to the distant productive grounds on the west coast and further spatial restrictions may therefore affect the economic viability of these vessels.

(iv) Demersal shark longline fishery

The demersal shark longline sector is an inshore fishery which generally operates in shallow water less than 100m in depth. Bottom-set gear is used to target two main species, the smooth-hound shark

[1] ZAR = South Africa Rand, US$1 ≈ ZAR8, October 2011

(*Mustelus mustelus*) and the soupfin shark (*Galeorhinus galeus*), with other species of Carcharhinids, Sphyrinids and batoids being landed as bycatch. Poor data currently exists for this fishery and considerable misreporting is thought to occur (C.Da Silva *pers. comm.*). Furthermore little is known of the activity of vessels locally within Algoa Bay thus warranting further investigation.

(v) Inshore demersal trawl fishery

The inshore demersal trawl fishery targets two main groups, the shallow-water hake (*Merluccius capensis) (M.paradoxus* is sometimes landed as incidental bycatch), and the east coast (or Agulhas) sole (*Austroglossus pectoralis*), and functions on a "dual quota" basis (DEAT 2005a; DEAT 2006). Although the fishery targets only two main species groups, the unselective nature of the fishery leads to the bycatch of several other species (DEAT 2005a), including many linefish species of which the populations are perceived to be in poor status. Although coastal bays are restricted trawling areas, only the sheltered western sector of Algoa Bay is protected and demersal trawling has been observed to occur in the eastern sectors of Algoa Bay. No detailed published information is available for the trawl fishery in Algoa Bay (although this data area available on request from DAFF) and the catch composition and contribution of threatened linefish species to the total catch in Algoa Bay is unknown. Nationally approximately ZAR100 million has been invested in assets for this sector with an annual landed catch of approximately ZAR60 million (DEAT 2005a).

2.7 Synthesis

Considerable research has been conducted within Algoa Bay. However, it has focused primarily on sandy beach ecosystems and the associated surf zones and nearshore environments with little to no research conducted on rocky intertidal and subtidal substrata. Although past research has made significant contributions to our understanding of the physical process which drive primary productivity and the distribution patterns of coastal and nearshore biota, our knowledge of sub-tidal communities is limited. Fishing intensity has increased significantly in recent years, yet our current knowledge of fishing activities within Algoa Bay is limited as local, finer scale fisheries assessments and monitoring programmes are scarce. In order to move towards an ecosystem based management approach in Algoa Bay taking fisheries related activities into account as a first step, high resolution spatial data on the distribution of the resources targeted by the fisheries, and the intensity and distribution of fishing effort is required. Baseline data are therefore needed to facilitate the development of a spatial management plan and to produce detailed monitoring protocols to evaluate the effectiveness of the plan. These monitoring programmes should measure temporal changes in pressures on the environment and the influence of these on the state of ecosystem resources. This will contribute to evaluating the effects of management interventions and further our understanding of other long-term drivers such as those of climate change.

The remainder of this study focuses on assessing fisheries as the main ecosystem drivers as the full range of factors required for inclusion in a fully holistic ecosystem based management approach was beyond the scope of this research. Additional factors such as water quality and pollution, coastal development and tourism need to be incorporated in the future through the adaptive management process. This review has contributed to the identification of available spatial data sources for use in marine spatial planning, and highlighted information gaps requiring further investigation to aid management and monitoring of fisheries resources and fisheries activities in Algoa Bay (Table 2.1).

Table 2.1. Marine spatial data sources available for spatial planning and additional fisheries related requirements identified in Algoa Bay.

Data for Chapter 7: Systematic Conservation Planning	Important spatial data gaps requiring further investigation
1. Intertidal habitat classification (Clark and Lombard 2007) 2. Subtidal substrate composition (Bremner 1978) 3. Bathymetry and depth categories (SANHO 1975) 4. Penguin foraging areas (Pichegru *et al.* 2010)	1. Reef locations 2. Reef fish community structure and abundance (Chapter 3) 3. Demersal fish communities (Chapter 4) 4. Recreational fishing activities (Chapter 5) 5. Commercial fisheries activities (Chapter 6)

CHAPTER 3
ASSESSMENT OF REEF LINEFISH COMMUNITIES IN ALGOA BAY

3.1 Introduction

Of the approximately 2 200 fish species occurring in the waters off southern Africa approximately 80% are found in the coastal waters between the shoreline and the 200m isobath (Smith and Heemstra 2003; Heemstra and Heemstra 2004). This region is also subject to high levels of fishing pressure which have increased considerably since the start of the boat-based linefishery in the mid 1800s (Thompson 1913; Griffiths 2000). Fishing is one of the major drivers of change in coastal ecosystems and has been reported to have considerable effects on marine communities globally (Jackson *et al.* 2001) and in South Africa (Götz *et al.* 2008; Götz *et al.* 2009b), and may lead to serial depletion of the fishery resources (Pauly *et al.* 1998; Pauly 2008).

Reef fish are particularly susceptible to overexploitation due to their longevity, slow growth rates, late maturity and high residency (Buxton 1993; Brouwer and Griffiths 2005a). Many species also exhibit complex life history strategies including serial hermaphroditism with size-based sexual dimorphism. These life history characteristics coupled with high levels of targeted exploitation along the South African coastline led to the collapse of the linefishery, with 18 of the 27 most heavily targeted linefish species now considered overexploited (Griffiths 2000; Atkinson and Clark 2005). A state of emergency was declared in the linefishery in December 2000 (DEAT 2000) which resulted in the subsequent reduction in commercial linefish effort. However, while size and bag regulations were amended for the recreational sectors, effort remained unlimited. The recreational sector therefore continued to contribute significantly to the overall harvest of linefish and a recent estimate of the recreational harvest indicated that it may be as much as twice that of the landed weight of the commercial sector (Atkinson and Clark 2005). Several reef associated species are also landed as bycatch in the commercial trawl fisheries, further contributing to their dire status.

MPAs have proven effective in protecting reef fish communities internationally (Willis *et al.* 2000; Mosquera *et al.* 2000; Halpern 2003; Willis *et al.* 2003; Lester *et al.* 2009) and locally within South Africa (Buxton and Smale 1989; Bennett and Attwood 1991; Cowley *et al.* 2001; Götz *et al.* 2009b; Mann 2010). The level of protection afforded to reef fish communities through MPAs is dependent on the representation and distribution of suitable habitat types and the spatial and temporal variability in assemblages. Selecting suitable locations for the protection of reef fish in MPAs is essential and requires detailed information (Kelleher 1999), particularly on the distribution of habitat types and the community structure on different reef complexes. There is a paucity of such information in South Africa, which has been a major limitation in the design and establishment of new MPAs for the protection of these species. The identification of suitable reef habitats and assessment of reef fish communities is a critical first step towards enhancing the design of reserve networks and protection afforded to reef communities through spatial management.

In many cases fisheries dependent data have been used where information on community composition is limited. However, there are many biases associated with such data (Smith *et al.* 2011). In particular the spatial accuracy of the data is questionable as fishermen may utilise different gears to target different species in particular areas and habitats, leading to a non-random sampling strategy (Smith *et al.* 2011). Catches are also biased by changes in fishery regulations (Donovan 2010) and influenced by the gear type used (Smith *et al.* 2011). Fisheries independent surveys avoid these biases through appropriate design of a randomly stratified sampling approach over time and space and allow for the standardisation of methods improving comparability between sites. Fishery independent surveys are therefore superior for conducting baseline assessments and for evaluating future changes in the state of the reef linefish resources.

Two fishery independent survey techniques have been widely used to assess reef fish communities. Underwater visual census (UVC) has been used internationally (Willis *et al.* 2000; Claudet *et al.* 2006; Smith-Vaniz *et al.* 2006; De Raedemaecker *et al.* 2010) and in South Africa (Smith 2005b; Mann *et al.* 2006; Bennett *et al.* 2009; Götz *et al.* 2009b). Similarly, controlled angling using catch per unit effort (CPUE) as an index of abundance has been used both internationally (Zeller and Russ 1998; Millar and Willis 1999; Haggarty 2005; Haggarty and King 2006) and locally in South Africa (Smith 2005b; Bennett *et al.* 2009; Götz *et al.* 2009b).

UVC techniques are considered superior to other survey methods (Kulbicki 1998) as they allow for the *in situ* estimation of reef fish abundance, community structure and diversity as well as the simultaneous assessment of benthic communities. UVC does, however, have inherent biases including underestimation of cryptic (Kulbicki 1998; De Girolamo and Mazzoldi 2001) and abundant species (Richards and Schnute 1986), and the distribution and behaviour of the species may be influenced by the presence of the observer (Cowley and Naesje 2004). Furthermore, within- and between-observer error may lead to bias in the datasets (Watson and Quinn 1997). The focus of the current study was, however, on dominant reef species targeted by the fisheries, these are typically larger conspicuous species which are not overly-abundant in the study area, reducing some of the bias associated with this method. Certain species may, however, be influenced by the presence of the observer but this remains constant throughout the study, and observer bias can only be reduced through the use of experienced scientific divers and sufficient training which formed part of this study.

Conducting UVC surveys in high energy coastal environments is strongly influenced by environmental conditions including currents, surge and visibility (Mann *et al.* 2006; Bennett 2007), and they are constrained to the depth and bottom time limitations of SCUBA diving. South African legislation also requires sizeable dive teams of certified scientific divers, creating additional logistical restrictions for conducting UVC. Recent advances have seen the use of underwater video techniques that may be deployed and operated remotely, which overcome these limitations (Willis *et al.* 2000; Willis and Babcock 2000; Lam *et al.* 2007). However, environmental conditions still play a major role when employing these methods, and financial constraints may limit the availability of such equipment to many research organisations at present.

Controlled angling on the other hand is influenced less by environmental conditions and can be conducted in areas where poor water visibility is common. It also allows for the accurate measurement of fish length, tagging of individuals for movement studies, is not constrained by the limitations of SCUBA or diving legislation, and is relatively inexpensive and simple to implement (Bennett 2007). A limitation is that controlled angling is selective for certain species, and excludes all herbivorous or corallivorous species which do not take baited hooks (Côté and Perrow 2006). However, unlike in tropical areas where herbivorous species are heavily targeted and comprise a significant proportion of the landed catch in coastal fisheries, herbivorous groups do not form a major target group for any fisheries sector in Algoa Bay and are therefore not influenced directly by these activities. Further limitations of controlled angling are that the size distribution sampled is influenced by hook selectivity (Côté and Perrow 2006) and post-release mortality may also occur due to handling stress or injury to fish (Grixti *et al.* 2008; Alós 2009). Although controlled angling does not allow for as detailed assessments of community structure as UVC, it provides a simple method for the assessment of key species targeted by the fisheries active in an area that can be conducted relatively easily over large geographical areas allowing large sample sizes to be attained. Controlled angling therefore provides a good means for the assessment of reef linefish communities available to, and targeted by fisheries activities, these being the key drivers under investigation in this study. A combination of both methods is therefore appropriate for obtaining baseline information on the linefish community structure and state of the targeted linefish resources, thereby contributing to future management of fisheries activities and monitoring of the targeted resources.

Ensuring adequate representation of reef fish communities within MPA networks requires that different community types can be identified and distinguished from each other. This is often difficult as natural variability is inherently high in reef fish populations (García Charton *et al.* 2000; Willis *et al.* 2006; Bennett *et al.* 2009) due to the influence of numerous biophysical and anthropogenic factors. In order for future management decisions to be effective in achieving their intended purposes, factors which contribute to the natural variability in populations need to be indentified and their effects on the community determined. Furthermore indicator species need to be identified as they are often used as proxies for assessing the state of the environment as they are more easily measurable (Vos *et al.* 2000). An understanding of the factors which contribute to spatial and temporal variability in abundance and size of indicator species will contribute to improving management of the resources and the design of long-term monitoring programmes, thereby increasing the power to distinguish real trends from natural variability.

Monitoring programmes aim to detect directional changes in communities and indicators in response to changes in management regimes and environmental and socio-economic drivers. Baseline data and control sites are essential for distinguishing trends in the state of the ecosystem in response to management actions from natural environmental drivers. MPAs are therefore an important component for long-term monitoring of reef fish communities as they provide a measure of the natural variability of species abundances and community structure, and allow natural directional changes to be identified and distinguished from responses to increasing pressure from socio-economic drivers and management interventions. MPAs thus serve as control sites and non-destructive fishery independent sampling protocols therefore need to be employed to allow for comparisons between no-take MPAs and exploited sites to be made.

Subtidal reef ichthyofaunal data within Algoa Bay are limited, arising from a few studies covering a small geographical area (Buxton 1987; Beckley and Buxton 1989). Little is therefore known regarding the spatial and temporal patterns in reef fish abundance and community structure within Algoa Bay, which has hindered local spatial management initiatives. This chapter therefore aimed to obtain baseline information on reef fish communities throughout Algoa Bay using a non-destructive randomly stratified survey design to investigate spatial and temporal trends and identify key biophysical factors influencing reef fish communities in an unbiased manner. The resulting baseline information is required for and will contribute to marine spatial planning in Algoa Bay (Chapter 7) and the development of long-term monitoring protocols for evaluation (Chapter 8). The main objectives of this chapter were to:

1. to identify and characterise the topography of reef habitat in Algoa Bay;
2. to determine spatial and temporal trends in reef linefish community structure; and
3. to evaluate key parameters influencing the relative abundance and mean size of dominant reef linefish to aid future spatial planning and monitoring in Algoa Bay.

3.2 Materials and methods

3.2.1 Reef mapping

(a) Identification of reef areas

Knowledge on the distribution and sizes of reef complexes within the Algoa Bay area is limited. Previous side-scan sonar work was conducted along the east coast of South Africa in the mid 1970s. This work focused primarily along the outer shelf edge with work conducted within Algoa Bay limited to a narrow band along the western sector between Cape Recife and the Port Elizabeth harbour (Flemming 1978; Buxton 1987) (Figure 2.1). In order to identify representative reef areas within Algoa Bay four sources of information were utilised:

1. The original side-scan sonographs from mapping conducted in 1983 for the western sector of the bay were obtained and digitised and rectified in ArcMap 9.2 (Environmental Systems Research Institute (ESRI)).
2. South African Nautical (SAN) Charts (SANHO 1975) were used to identify areas with elevated features within the bay potentially indicating high profile reef areas.
3. Point data with depth and substrate type were obtained from the South African Navy Hydrographic Office (SANHO). This consisted of eight main substrate types including coral, pebbles, rock, mud, sand, stone, shells and gravel, and combinations of each; and
4. Interviews were conducted with researchers and commercial and recreational anglers in order to obtain information on reef localities within Algoa Bay. This information was either obtained in a course format as sketches over SAN Charts, which were then digitised and rectified in ArcMap 9.2, or as Global Positioning System (GPS) coordinates which were imported into ArcMap 9.2.

All data were imported into ArcMap 9.2 and converted to spatial layers which were intersected with a 500mx500m grid in order to identify potential reef areas within Algoa Bay (Figure 3.2). Initially four sites of appropriate depth range (0-35m) were selected for further finer scale investigation (Bird Island (BI), St Croix (StC), Cape Padrone (CP) and Woody Cape (WC)), with an additional three sites (Bell Buoy (BB), Riy Banks (RB) and Evans (Ev)) included during later stages of the project.

(b) Bathymetric mapping of reef areas

Low cost bathymetric mapping was conducted as described by Götz (2005) in each selected area in order to obtain fine scale bathymetric information and verify substrate types. Bathymetric mapping was conducted for a total 20 sea-days across the study areas selected. A 6.8m skiboat fitted with a plotter and colour echo sounder were used to obtain spatially referenced depth data, while the substrate type (hard or soft) was interpreted from the colour display on the echo sounder unit and recorded by the researchers. A predetermined grid was surveyed in each study area with transect lines approximately 100m apart. Vessel speed was maintained at approximately $10km.h^{-1}$ to reduce GPS error (Götz 2005). Point data were entered into an MS Access database and imported into ArcView 3.2 (ESRI), where spatial and depth information was verified and corrected if necessary. Continuous seafloor maps were created for each study area by interpolation using the spline tension method in Spatial Analyst 2.0a (ESRI) and one-meter depth contours were created. Soft seafloors typically have a gently sloping continuous pattern and the one-meter contours in conjunction with additional data sources were used to identify and differentiate potential reef areas from soft substrata within each study area. Some of the reefs in Algoa Bay were reported to be of low profile and it is likely that they are periodically inundated by sand movement. Interpretation of the colour echo sounder reading does not account for sand over the reef surface as a hard substrate is still depicted on the unit. In some instances a SeaViewer remote camera lens was lowered to the seafloor in order to verify the substrate type visually on the attached monitor. Extensive diving surveys were conducted in the Bird Island area and assisted in verifying the accuracy of the mapping exercise. Poor visibility (<1m) in the other study areas limited the use of diving for verification purposes.

3.2.2 Assessment of linefish communities

Controlled angling surveys were the primary method used to assess the ichthyofaunal communities within each study area due to the poor water visibility inherent in the inshore regions of Algoa Bay. Where conditions allowed controlled angling surveys were supplemented with UVC. Controlled angling and UVC surveys were conducted between 1 February 2006 and 10 May 2009.

(a) Survey design

Sampling effort was stratified over study area, season, depth and reef profile (Figure 3.1). The calendar year was divided into two six-month seasons with summer extending from 1 November to 30 April each year. Within each study area a 100mx100m grid was used to stratify sampling effort over reef profile and depth using the data and maps produced during the mapping exercise (Figure 3.2 and 3.3). Depth was categorised as shallow (0-10m), medium (10-20m) and deep (20-30m), and profile as either high or low, creating a maximum of six strata within each study area (Figure 3.1). Each grid cell was assigned depth and profiles values determined from the continuous seafloor map, which was interpolated using Spatial Analyst in ArcView 3.2. Each grid cell was assigned a unique identification number and sampling sites for each stratum were selected using a random number generator. Prior to each seasonal sampling trip sites were selected for all study areas with the number of sampling sites selected per stratum within each study area proportional to the area. To avoid pseudoreplication, grid cells were never resampled during the same season (selection without replacement), although it was possible that they were randomly selected for the following seasonal survey. For each seasonal sampling trip additional sampling sites were selected from each stratum to account for instances when cells could not be sampled due to poor sea conditions or non-reef substratum. The centre point for each grid cell was loaded onto a handheld GPS and used to locate the sampling site within a given grid cell.

(b) Controlled angling surveys

All angling surveys were conducted while stationary on anchor using a 6.8m skiboat or 6m rubberduck. A Vemco minilog 8-TR temperature logger was secured above the anchor chain to record bottom temperature at one-minute intervals during the angling period. Angling was conducted by three anglers at a time for a 20-minute period at each site. Anglers were either trained scientists or experienced volunteers trained in controlled angling. An additional team member recorded all data and assisted with handling the fish where required. Certain species were tagged for movement studies. Sardine was the only bait used and all tackle was standardised consisting of a 5-ounce sinker and a 4/O mustard J hook. Barbs were flattened against the shanks to facilitate hook removal and prevent unnecessary injury (Côté and Perrow 2006). All fishing was conducted at least one hour after sunrise or before sunset to exclude crepuscular activity (Bennett 2007).

At each site the GPS coordinates, water depth, substrate type (as interpreted from the colour echo sounder), starting time and anglers' names were recorded. Details for each fish were recorded in capture sequence and included the species name, angler name, fork length (FL) to the nearest millimeter and tag details where necessary. Swimbladders of fish that showed obvious signs of barotrauma, or fish that floated on the water surface after release, were deflated by inserting a sterilised surgical needle through the body wall (Götz 2005). Capture mortality was recorded. Fish that were bleeding excessively were not released and were noted as a mortality. In addition fish that floated on the surface after deflation of the swimbladder and were unable to descend into the water column were also noted as a mortality. This estimate of capture mortality is a minimum estimate as no subsequent information on survival rates of released fish could be obtained (Götz *et al.* 2007).

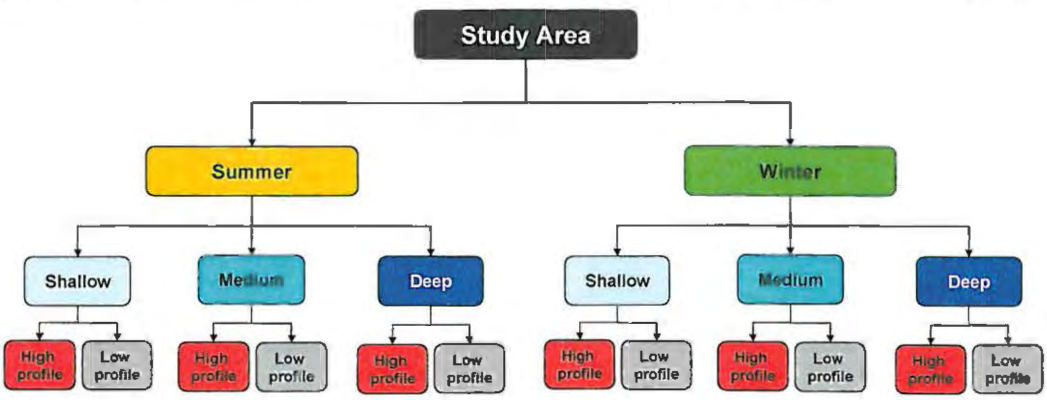


Figure 3.1. Stratified sampling design used in the field survey.

(c) Underwater visual census (UVC)

A weighted shot-line with surface marker buoy was deployed at each dive site. A Vemco minilog 8-TR temperature logger was attached above the chain on the shot-line to record bottom temperature at one-minute intervals for the duration of the dive. The boat skipper or dive supervisor recorded the GPS coordinates of the dive site when the shot-line was deployed as well as the start and end times of the dive. Two divers using SCUBA descended down the shot-line and on reaching the bottom each attached a dive reel to the anchor of the shot-line. The divers swam away from the shot-line in opposite directions until 10m away from the shot-line where each diver recorded the depth, substrate type, reef profile, dominant benthic growth and rugosity on an underwater slate (Götz 2005; Götz *et al.* 2009b). Depth was recorded from the diver's dive computer or submersible pressure gauge, and an average of all readings was taken as the site depth. Substrate was recorded as either rock where only reef was present or rock-sand if the reef was interspersed with sand patches. No counts were conducted over predominantly sandy substratum. Reef profile is the vertical relief of the rock formation and was categorised as high or low at each point count location by the diver. Dominant benthic growth was recorded as the most abundant species or functional group observed by the diver (e.g. bryozoans; coralline algae etc.). When detailed macro-benthic studies using an underwater digital camera and monopod were initiated (results not included in this thesis) dominant benthic cover at each dive site was obtained from digital photography but classified in the same manner as previously for the ichthyofaunal assessment. Rugosity is an estimate of the small-scale variation in surface height and was recorded as either high if the reef consisted of many crevices and holes, or low if the rock face was smooth. Visibility was estimated to the nearest meter by the divers using graduations on the dive reel line and the shot-line as visual reference. No counts were conducted if the visibility was below 3m.

Point counts were conducted by each diver and involved identifying the first species observed and conducting a continuous circular count of all individuals of that species which occurred within a maximum radius of 5m (Götz 2005; Bennett 2007; Götz *et al.* 2009b). On completion of the first rotation a second species was identified and the process repeated. This was continued until no further species were observed. Roman is a predatory reef sparid which is highly resident (Kerwath *et al.* 2007a; Kerwath *et al.* 2007b) and targeted by the offshore skiboat fishery (Griffiths 2000; Brouwer and Buxton 2002). Due to these characteristics it has been selected as an indicator species in prior reef fish assessments in South Africa, using both abundance and length to evaluate the impacts of fishing (Bennett *et al.* 2009). Roman length was therefore estimated for all fish in the area by each diver, and assigned to 5cm classes (Mann *et al.* 2006; Bennett 2007) on completion of the count. Where time allowed, each diver conducted a second point count swimming out from the shot-line at an angle of 90° to the first count.

3.2.3 Data analysis

(a) Diversity and community analyses

Analysis of diversity and community structure was conducted using the Plymouth Routines in Multivariate Ecological Research (PRIMER) Version 6.1.6 package. Four measures of diversity, species richness (S), Pielou's Evenness (J'), Shannon-Wiener diversity (H') and the Taxonomic Diversity indices were calculated on untransformed data.

Species richness (*S*) is the number of species captured or recorded at each site during controlled angling or diving surveys respectively.

The Shannon-Wiener Index (*H'*) is the most commonly used diversity index and is calculated as:

$$H' = -\sum_i p_i \log(p_i) \qquad \text{Equation 3.1}$$

where p_i is the proportion of the total count arising from the i^{th} species (Clarke and Warwick 2001b).

Pielou's Evenness Index (*J'*) provides a measure of the equitability of the number of individuals across species and is calculated as:

$$J' = \frac{H'}{\log S} \qquad \text{Equation 3.2}$$

where *H'* is the Shannon-Wiener diversity index and *S* is the number of species (Clarke and Warwick 2001b).

Taxonomic diversity (Δ) takes into account the relatedness of species using taxonomic distances and represents the average taxonomic distance apart of every pair of individuals in the sample (Clarke and Warwick 2001b) and is calculated as:

$$\Delta = \frac{\left[\sum\sum_{i<j} \omega_{ij} x_i x_j\right]}{\left[n(n-1)/2\right]} \qquad \text{Equation 3.3}$$

where ω_{ij} is the taxonomic distance between species *i* and *j*, x_i is the abundance of the i^{th} species and n is the number of individuals in the sample.

The diversity indices were checked for normality using a Kolmogorov-Smirnov Test and the homogeneity of variances with Levene's Test, if assumptions were met a students t-test or one-way ANOVA was conducted. If assumptions were not met non-parametric Mann-Whitney U tests for paired comparisons or a Kruskal-Wallis ANOVA for comparison of multiple groups was undertaken. Where significant differences between multiple groups occurred, post hoc testing was conducted by pairwise Mann-Whitney U tests with a Bonferroni adjusted level of significance, where the required alpha value is divided by the number of pairwise tests conducted calculated by the following equation:

$$n = \frac{K(K-1)}{2} \qquad \text{Equation 3.4}$$

where n is the number of tests conducted and K is the number of groups being compared (Zar 1999).

Non-parametric multivariate analyses were conducted on catch rate data from controlled angling surveys and count data from diving surveys which were square and fourth root transformed, respectively, prior to the calculation of Bray-Curtis similarity matrices (Clarke and Warwick 2001b). A higher transformation was chosen for the abundance data from UVC due to the higher variance of counts. This is desirable to prevent the results being biased by the counts of a few very abundant species (Clarke and Warwick 2001b). Non-metric multidimensional scaling (nMDS) ordination and hierarchical classification were used to display the influence of categorical explanatory factors on community structure graphically (Clarke and Warwick 2001b). Analysis of similarity (ANOSIM) was conducted to determine if significant differences in community structure existed between categorical groups, and the BIOENV procedure was used to investigate the relationship between continuous factors and community structure (Clarke and Warwick 2001b). The importance of individual species to the within group similarity and between group dissimilarity was investigated using the SIMPER routine. Species dominance plots were used to present the differences in community structure under different categorical factors.

(b) Relative abundance and size analysis

Generalised linear models (GLMs) were used to model the influence of explanatory factors on the relative abundance (CPUE or count) and size of individual species. The optimal combination of factors influencing each parameter was assessed by testing between competing models using the Akaike Information Criterion (AIC) (Akaike 1973). The AIC consists of two components, the negative log-likelihood, which measures the lack of model fit to the observed data, and a bias correction factor based on the number of model parameters, which aims to reduce model complexity, and is calculated as:

$$AIC = -2\ln\left[L(\theta_p|y) \right] + 2p \qquad \text{Equation 3.5}$$

Where: p is the number of free parameters, and $L(\theta_p|y)$ is the likelihood of model parameters given the data y (Johnson and Omland 2004).

Secondary interactions were not included due to the number of factors being investigated and complexities of each model. The sample size for each model was large and the number of factors included in the final models never exceeded one third of the sample size (Crawley 1993).

The data obtained from controlled angling and diving surveys are both discrete count response variables which follow the poisson distribution, for which the log-link function is commonly used (McCullagh and Nelder 1995; Dobson 2002), which takes the form:

$$g(\mu) = \log(\mu) \qquad \text{Equation 3.6}$$

where $g(\mu)$ is the linear predictor and μ is the population mean.

A feature of the Poisson model is that the expected mean is equal to the variance. Dispersion occurs if the variance is either greater (overdispersion) or less (under dispersion) than the mean for the dependent variable, which leads to deflated standard errors, increasing the possibility of Type 1 errors (Elhai *et al.* 2008). Overdispersion can be incorporated into a Poisson model by introducing a dispersion parameter into the relationship between the variance and the mean (Pedan 2001) in the form:

$$Var(y) = \phi\mu \qquad \text{Equation 3.7}$$

Where ϕ is the dispersion parameter and μ is the mean.

Dispersion is calculated as the ratio of the sum of residuals to the degrees of freedom (McCullagh and Nelder 1995) as follows:

$$\phi = \frac{Sum\ of\ residuals}{df} \qquad \text{Equation 3.8}$$

Dispersion values approximating 1 indicate a good model fit with little dispersion, whereas values above or below indicate over or under dispersion respectively. Where over or under dispersion occurred the Poisson model was scaled using Pearson Residuals to obtain a dispersion value approximating 1, with the standard error and model statistics being corrected accordingly. Poisson models incorporating a dispersion factor do not influence the values of the parameters estimates, only the estimates of standard error and confidence intervals which in turn affect the significance of model results.

A GLM of the following form was used to model the influence of explanatory factors on the catch rate from controlled angling surveys:

$$\begin{aligned} Log(catch) = {} & \beta_0 + \beta_1(year) + \beta_2(area) + \beta_3(season) + \beta_4(period) \\ & + \beta_5(temperature) + \beta_6(depth) + \varepsilon \end{aligned} \qquad \text{Equation 3.9}$$

where $\beta_{0\text{-}i}$ are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

Year, area, season and period were categorical factors while temperature and depth were continuous predictors. Year was taken as the calendar year in which the survey was conducted. Area was one of the seven study areas (RB, BB, StC, Ev, BI, WC and CP[2]) selected for the assessment of reef fish communities. Season was categorised as either summer or winter, with summer beginning on 1 November each year. A survey day was divided into three periods, morning (1 hour after sunrise to 11:00), midday (11:00-14:00) and afternoon (14:00 to 1 hour before sunset) with the period in which all or most of the sampling station was completed being recorded. Temperature for each site was calculated as the average temperature recorded by the Vemco minilog temperature logger secured above the boat anchor chain. Depth was taken as the mean station depth from the vessel's echo sounder.

[2] RB=Riy Banks; BB=Bell Buoy; StC=St Croix; Ev=Evans; BI=Bird Island; WC=Woody Cape; CP=Cape Padrone, see Figure 3.2 for locations

For the initial analysis across all survey areas within Algoa Bay substrate was not included as a factor as only sites which were positively identified as reef areas were included in order to standardise the comparison. After the initial assessment additional sites which were a combination of rock and rock-sand were included and an additional factor, substrate, was included in the GLM as required and the model took the form:

$$\begin{aligned} Log(catch) = \beta_0 + \beta_1(year) + \beta_2(area) + \beta_3(season) + \beta_4(period) \\ + \beta_5(temperature) + \beta_6(depth) + \beta_7(substrate) + \varepsilon \end{aligned} \qquad \text{Equation 3.10}$$

where $\beta_{0\text{-}i}$ are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

In order to investigate the effects of explanatory factors on count data from diving surveys, a GLM of the following form was applied:

$$\begin{aligned} Log(count) = \beta_0 + \beta_1(season) + \beta_2(period) + \beta_3(profile) + \beta_4(substrate) \\ + \beta_5(rugosity) + \beta_6(depth) + \beta_7(temperature) + \beta_8(visibility) + \varepsilon \end{aligned} \qquad \text{Equation 3.11}$$

where $\beta_{0\text{-}i}$ are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

All factors are the same as described in the section above. Additional categorical explanatory factors included reef profile and rugosity which were classified as high or low by the divers, and substrate which was recorded as rock if the count was conducted over solid reef, or as rock-sand if the reef was interspersed with sand patches or gullies. Visibility was estimated by the divers and depth was recorded by the diver's dive computer or submersible pressure gauge.

The measurement of fork length provides a continuous response variable which is normally distributed, for which the identity-link function is commonly used (Dobson 2002), and takes the form:

$$g(\mu) = \mu \qquad \text{Equation 3.12}$$

where $g(\mu)$ is the linear predictor and μ the population mean.

To model the effect of explanatory factors on fish length from controlled angling surveys, the following GLM was applied:

$$\begin{aligned} forklength = \beta_0 + \beta_1(year) + \beta_2(area) + \beta_3(season) + \beta_4(period) \\ + \beta_5(temperature) + \beta_6(depth) + \varepsilon \end{aligned} \qquad \text{Equation 3.13}$$

where $\beta_{0\text{-}i}$ are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

To model roman length from diving surveys, the GLM took the form:

$$\begin{aligned} forklength = \beta_0 + \beta_1(season) + \beta_2(period) + \beta_3(profile) + \beta_4(rugosity) \\ + \beta_5(depth) + \beta_6(temperature) + \beta_7(visibility) \\ + \beta_8(substrate) + \beta_9(area) + \varepsilon \end{aligned} \qquad \text{Equation 3.14}$$

where $\beta_{0\text{-}i}$ are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

Diagnostic plots and goodness of fit statistics were used to assess the appropriateness of each model.

3.3 Results

3.3.1 Reef mapping

At the onset of the project four areas representative of the nearshore and offshore (<35) reefs in Algoa Bay were identified and topographically mapped. These study areas were located within the proposed AENP MPA footprint and represented two offshore reef complexes, Bird Island (BI) and St Croix (StC), and two nearshore reef complexes, Cape Padrone (CP), and Woody Cape (WC) (Figure 3.2). In the final year of study three additional reef complexes were selected and mapped to increase representivity of reef habitats across the full spatial scale of Algoa Bay, including two reefs outside of the proposed MPA footprint. These study areas represented two additional offshore reef complexes, Riy Banks (RB) and Evans (Ev) and one nearshore reef, Bell Buoy (BB). Continuous seafloor maps were produced for each study area and results were used to stratify sampling effort over depth and reef profile (Figure 3.3).

3.3.2 Reef linefish communities in Algoa Bay

(a) Assessment of reef linefish communities by controlled angling surveys

A total of 453 controlled angling sites were targeted across all study areas within Algoa Bay at which 5 031 fish were captured, representing 44 species from 15 families (Table 3.1). Overall 19 species and 7 families recorded during the controlled angling surveys were not recorded during diving surveys. Only five species from the Sparidae family were common across all study areas and included santer (*Cheimerius nufar*), fransmadam (*Boopsoidea inornata*), roman (*Chrysoblephus laticeps*), steentjie (*Spondyliosoma emarginatum*) and blacktail (*Diplodus sargus capensis*) which cumulatively accounted for 79.6% of the total catch by number, with a further two species, white seacatfish (*Galeichthys feliceps*) and red tjor-tjor (*Pagellus natalensis*) occurring in all but the BI study area. The Sparidae family was the most speciose and abundant with 20 species recorded which accounted for 86.4% of the total catch by number. Santer was the most abundant species both across all areas, accounting for 45.1% of the total catch, and within each study area, comprising between 29.8 and 49.0% of the catch within each area. Fransmadam was the second most abundant species overall making up 16.5% of the total catch, but was only the second most abundant species in the BB, CP and WC areas. Roman was the third most abundant species overall, comprising 14.0% of the total catch, but was the second most abundant species in the RB, Ev and BI areas and third most abundant in the CP and WC areas. White seacatfish and silver kob (*Argyrosomus inodorus*) were the second and third most abundant species respectively in the StC area, while red tjor-tjor was the third most abundant species in the BB area.

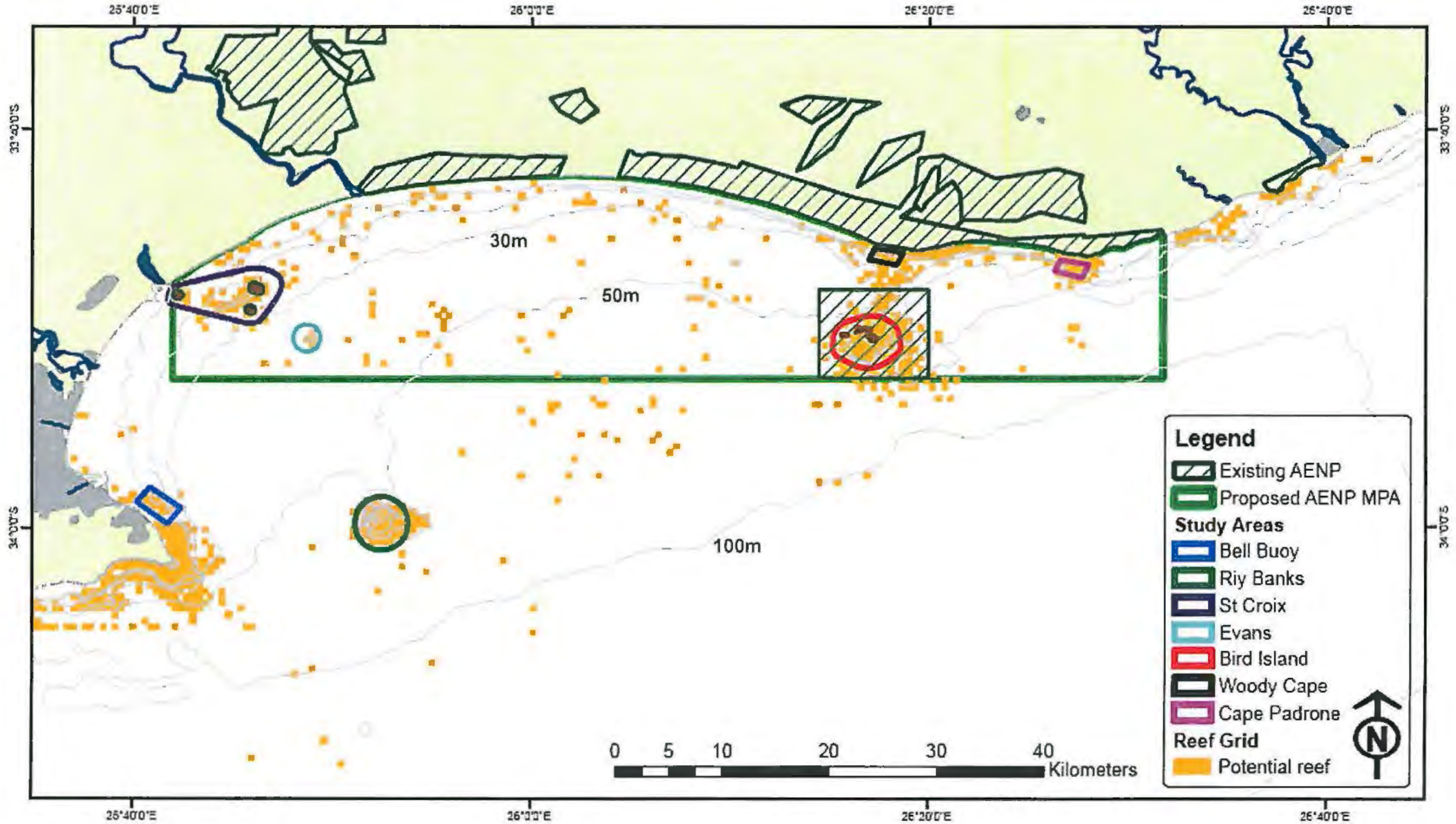


Figure 3.2. Potential reef areas identified and study sites selected for further investigation of reef fish communities in Algoa Bay.

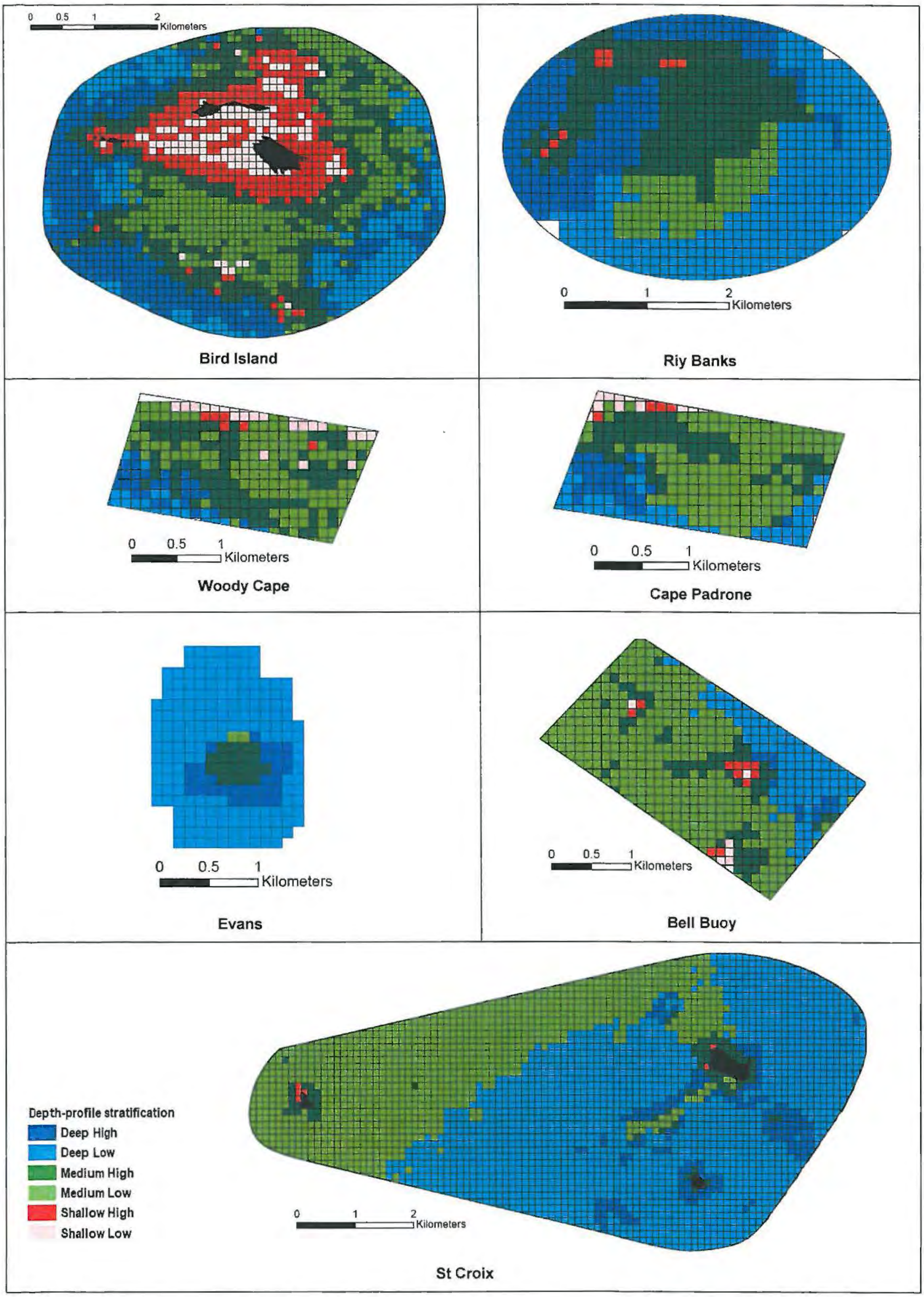


Figure 3.3. Stratification of sampling effort over depth and reef profile within each of the reef study areas in Algoa Bay.

Table 3.1. Sampling effort and species captured per study area during controlled angling surveys in Algoa Bay (* indicates families and species also recorded during UVC; number of fishing stations per area indicated under the area name, n= number of individuals caught). Species contributing to greater than 4% of the community in each area highlighted in grey.

Class	Family	Scientific name	Common name	Bell Buoy (stations=34)		St Croix (stations=101)		Riy Banks (stations=40)		Evans (stations=15)		Bird Island (stations=145)		Cape Padrone (stations=62)		Woody Cape (stations=56)		Total	
				N	%	n	%	n	%	n	%	n	%	n	%	n	%	n	%
CHONDRICHTHYES	Carcharhinidae	*Carcharhinus brachyurus*	Bronze whaler			3	0.3					7	0.4	1	0.2			11	0.2
		Carcharhinus obscurus	Dusky shark									7	0.4					7	0.1
	Triakidae	*Mustelus mustelus*	Smooth-hound			3	0.3	1	0.2			6	0.3	5	0.8	4	0.5	19	0.4
		Triakis megalopterus	Spotted gullyshark									2	0.1					2	0.0
	Rajidae	*Raja straeleni*	Biscuit skate			1	0.1											1	0.0
	Scyliorhinidae	*Haploblepharus fuscus*	Brown shyshark									1	0.1					1	0.0
		Poroderma africanum	Pyjama catshark *	1	0.4	1	0.1	1	0.2									3	0.1
	Sphyrnidae	*Sphyrna zygaena*	Smooth hammerhead			3	0.3											3	0.1
	Squalidae	*Squalus megalops*	Bluntnose spiny dogfish *	15	5.7	38	4.2			1	0.6			1	0.2	23	3.0	78	1.6
OSTEICHTHYES	Ariidae	*Galeichthys feliceps*	White seacatfish *	9	3.4	193	21.5	6	1.3	1	0.6			8	1.2	6	0.8	223	4.4
	Carangidae	*Seriola lalandi*	Giant yellowtail *									7	0.4			1	0.1	8	0.2
		Seriola rivoliana	Longfin yellowtail *			1	0.1											1	0.0
		Trachurus trachurus capensis	Horse mackerel	1	0.4													1	0.0
	Haemulidae	*Pomadasys olivaceus*	Piggy	12	4.6	27	3.0											39	0.8
	Pomatomidae	*Pomatomus saltatrix*	Elf	12	4.6	38	4.2			8	5.1					3	0.4	61	1.2
	Sciaenidae	*Argyrosomus inodorus*	Silver Kob	5	1.9	177	19.7							1	0.2	1	0.1	184	3.7
		Atractoscion aequidens	Geelbek			5	0.6	2	0.4			2	0.1	2	0.3			11	0.2
	Scombridae	*Scomber japonicus*	Mackerel	2	0.8	21	2.3											23	0.5
	Serranidae	*Acanthistius sebastoides*	Koester			1	0.1											1	0.0
		Epinephelus chabaudi	Moustache rockcod									1	0.1					1	0.0
		Epinephelus marginatus	Yellowbelly rockcod *											1	0.2	1	0.1	2	0.0
	Sparidae	*Argyrozona argyrozona*	Carpenter							8	5.1							8	0.2
		Boopsoidea inornata	Fransmadam *	41	15.6	18	2.0	77	16.6	20	12.7	396	21.6	215	33.4	63	8.2	830	16.5
		Cheimerius nufar	Santer *	101	38.4	268	29.8	149	32.2	77	49.0	783	42.7	307	47.7	582	75.4	2267	45.1
		Chrysoblephus anglicus	Englishman *									1	0.1					1	0.0
		Chrysoblephus cristiceps	Dageraad	4	1.5	7	0.8	4	0.9			17	0.9			6	0.8	38	0.8

Table 3.1 cont. Sampling effort and species captured per study area during controlled angling surveys in Algoa Bay (* indicates families and species also recorded during UVC; n = number of fishing stations per area, and number of individuals caught). Species contributing to greater than 4% of the community in each area highlighted in grey.

Class	Family	Scientific name	Common name	Bell Buoy (n=34)		St Croix (n=101)		Riy Banks (n=40)		Evans (n=15)		Bird Island (n=145)		Cape Padrone (n=62)		Woody Cape (n=56)		Total	
				N	%	n	%	n	%	n	%	n	%	n	%	n	%	n	%
	Sparidae	*Chrysoblephus gibbiceps*	Red stumpnose *					7	1.5			8	0.4	2	0.3			17	0.3
		Chrysoblephus laticeps	Roman *	1	0.4	16	1.8	128	27.6	30	19.1	431	23.5	59	9.2	39	5.1	704	14.0
		Cymatoceps nasutus	Black musselcracker *									6	0.3	10	1.6	2	0.3	18	0.4
		Diplodus cervinus hottentotus	Zebra *									1	0.1					1	0.0
		Diplodus sargus capensis	Blacktail *	5	1.9	2	0.2	14	3.0	1	0.6	11	0.6	4	0.6	1	0.1	38	0.8
		Lithognathus mormyrus	Sand steenbras *	1	0.4													1	0.0
		Pachymetopon aeneum	Blue hottentot *					23	5.0	1	0.6	18	1.0			1	0.1	43	0.9
		Pagellus natalensis	Red tjor-tjor *	33	12.5	46	5.1	2	0.4	1	0.6			3	0.5	4	0.5	89	1.8
		Petrus rupestris	Red steenbras *			1	0.1	2	0.4			8	0.4	1	0.2			12	0.2
		Polysteganus praeorbitalis	Scotsman *					5	1.1			43	2.3	1	0.2			49	1.0
		Polysteganus undulosus	Seventy-four *					17	3.7			11	0.6			1	0.1	29	0.6
		Pterogymnus laniarius	Panga			8	0.9	15	3.2	6	3.8							29	0.6
		Rhabdosargus globiceps	White stumpnose *	1	0.4													1	0.0
		Sarpa salpa	Strepie *	2	0.8	1	0.1					5	0.3					8	0.2
		Spondyliosoma emarginatum	Steentjie *	16	6.1	18	2.0	10	2.2	3	1.9	62	3.4	21	3.3	34	4.4	164	3.3
	Tetraodontidae	*Amblyrhynchotes honckenii*	Evileye puffer *									1	0.1	1	0.2			2	0.0
		Lagocephalus scleratus	Silverstripe puffer			1	0.1											1	0.0
	Triglidae	*Chelidonichthys capensis*	Cape Gurnard	1	0.4													1	0.0
			TOTAL individuals	263	100	898	100	463	100	157	100	1835	100	643	100	772	100	5031	100
			TOTAL species	19	43	25	57	17	39	12	27	24	55	18	41	17	39	44	

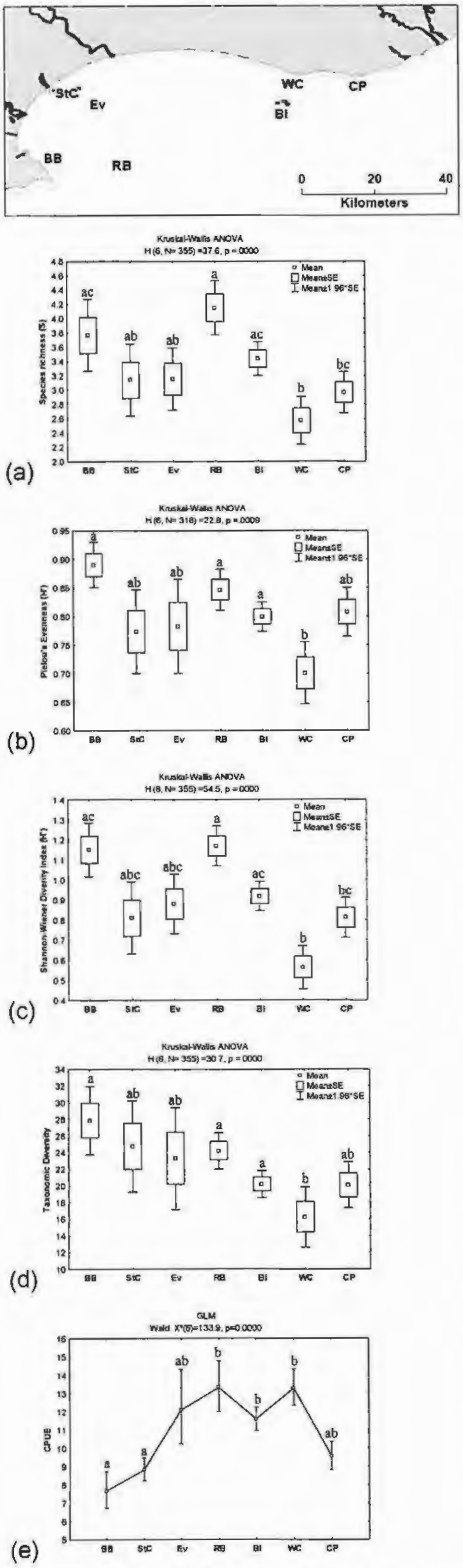


Figure 3.4. Spatial trends in linefish diversity and abundance in Algoa Bay from controlled angling surveys. (a)Species richness, (b) Pielou's Evenness, (c) Shannon-Wiener Diversity, (d) Taxonomic diversity, and (e) relative abundance. Study areas indicated in the map depicted on the x-axis, letters above denote significant differences.

(i) Diversity across study areas

Species richness differed significantly among areas (Kruskal-Wallis ANOVA, df=355, p<0.0001) (Figure 3.4a). Species richness at WC was significantly lower than RB, BB and BI, and CP was lower than RB.

The evenness of individuals across the number of species sampled represented by Pielou's Index differed significantly among areas (Kruskal-Wallis ANOVA, df=355, p<0.001) (Figure 3.4b), being highest for the BB area indicating the most even spread of individuals over species in this area. Pielou's Index was significantly lower in the WC area than at the BB, RB and BI areas.

Diversity measured by the Shannon-Wiener Index also indicated significant differences (Kruskal-Wallis ANOVA, df=355, P<0.0001) among areas showing similar patterns to species richness for each area (Figure 3.4c). Diversity was highest in the RB and BB areas, followed by BI which were all significantly greater than WC. The diversity in RB was also significantly higher than in CP.

Taxonomic Diversity differed significantly (Kruskal-Wallis ANOVA, df=355, p<0.0001) among areas showing a general decline from west to east across the study area (Figure 3.4d). Pairwise comparisons indicated that taxonomic diversity in the BB, RB and BI areas was significantly higher than in the WC area.

The relative abundance (standardised CPUE) of linefish (all species) differed significantly among areas (Wald $X^2(6)=133.88$; p<0.0001) (Figure 3.4e). Both BB and StC had significantly lower relative abundances than RB, BI and WC. Differences in CPUE between other areas were not significant.

(ii) Multivariate analysis of community structure

Hierarchical classification using Bray-Curtis similarities was used to aggregate the most similar areas for further analysis. Two principal communities were distinguished with the BB and StC areas grouping together at the 55% level of similarity (Principal Group 1), with the other study areas being more similar to each other and forming a second group (Principal Group 2) (Figure 3.5). The SIMPER routine showed that eight species contributed to 80% of the dissimilarity between Group 1 and Group 2 communities. Santer (19.8%), fransmadam (14.7%) and roman (14.1%) were the major contributors, cumulatively contributing 49% to the dissimilarity. Elf (6.8%), silver kob (6.7%), steentjie (6.3%), red tjor-tjor (5.9%) and white seacatfish (5.5%) also contributed significantly to the dissimilarity between Group 1 and 2 communities.

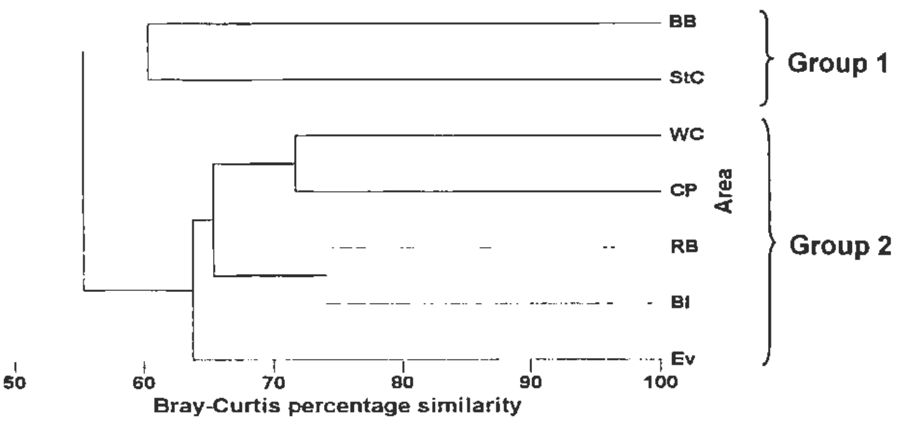


Figure 3.5. Hierarchical classification of controlled angling abundance data indicating the differentiation of two principal community groups.

In each area santer was the dominant species representing 39% and 49% in the Group 1 and Group 2 communities, respectively (Figure 3.6). Silver kob was the second most abundant species in the Group 1 community representing 14%, followed by fransmadam, white seacatfish and elf (*Pomatomus saltatrix*) which accounted for 10, 6 and 6% of the catch respectively. Fransmadam was the second most abundant species in Group 2 communities representing 20% of the catch, followed by roman, steentjie (3%) and scotsman (*Polysteganus praeorbitalis*) which represented 18, 3 and 1% of the catch respectively.

The differences between the two communities are further highlighted by the frequencies of occurrence (FoC) of certain species (Figure 3.7). In each principal group santer (91% Group 1 and Group 2), fransmadam (56% Group 1; 66% Group 2), steentjie (29% Group 1; 22% Group 2), blacktail (12% Group 1; 7% Group 2), bluntnose spiny dogfish (*Squalus megalops*) (6% Group 1; 4% Group 2) and dageraad (*Chrysoblephus cristiceps*) (6% Group 1; 5% Group 2) were recorded at a similar frequencies. Elf (44% Group 1; 2% Group 2), red tjor-tjor (32% Group 1; 3% Group 2), white seacatfish (26% Group 1; 4% Group 2), silver kob (24% Group 1; 4% Group 2) and piggy (*Pomadasys olivaceus*) (9% Group 1) were captured at angling sites more frequently in Group 1 communities than Group 2, with roman (26% Group 1; 64% Group 2) occurring more frequently in Group 2 communities. Black mussel cracker (*Cymatoceps nasutus*) (4%), blue hottentot (*Pachymetopon aeneum*) (10%), red stumpnose (*Chrysoblephus gibbiceps*) (5%), scotsman (9%) and seventy-four (*Polysteganus undulosus*) (6%) were only captured in the Group 2 study areas.

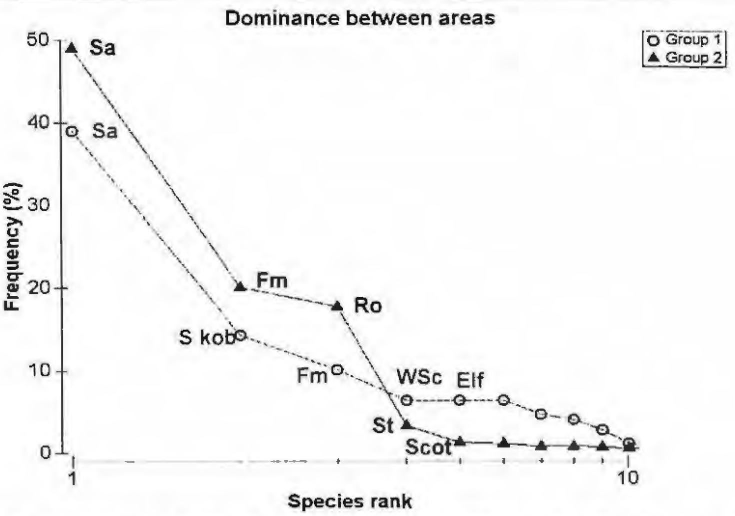


Figure 3.6. Dominance plots for the two distinguishable communities (Sa=santer; Fm=fransmadam; S kob=silver kob; Ro=roman; WSC=white seacatfish; St=steentjie; Scot=scotsman).

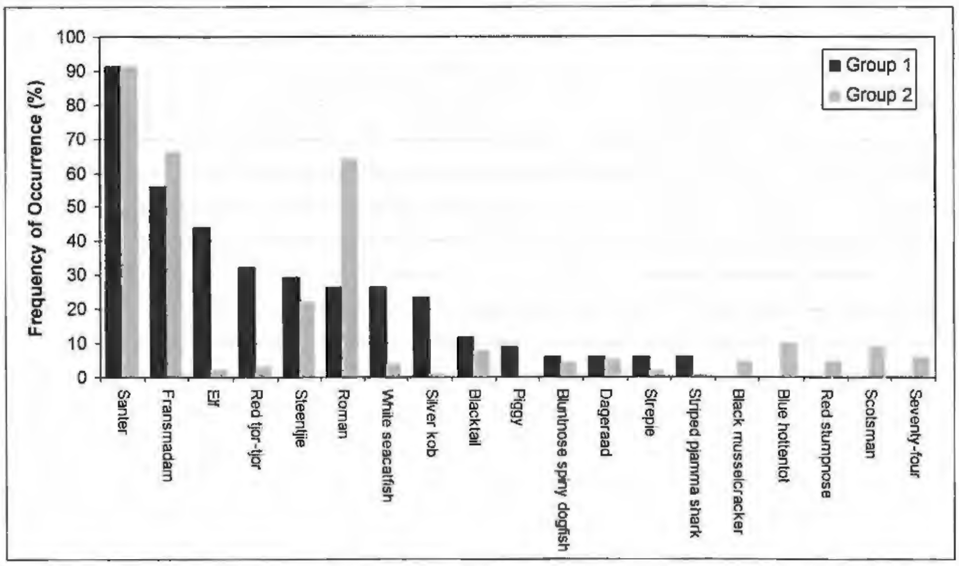


Figure 3.7. Frequency of capture for each species during controlled angling for the two principal community groups identified through multivariate analysis.

Santer was the only important fishery species which was captured in sufficient numbers within all study areas to allow broad spatial comparison (Table 3.2). Clear spatial differences in mean length and catch rates of adult and juvenile fish were apparent between areas. Mean length was highest in the RB (315±43mm), BI (314±49mm), CP (313±53mm) and Ev (309±44mm) areas, being lower in the StC (293±54mm) and BB (260±48mm) areas (Table 3.2). The proportion of mature fish caught was lowest in BB (19%) and highest in the RB (65%). Adult CPUE ranged from 0.4 to 3.6 fish.angler-hour^{-1} being lowest in the BB area and highest in the WC area. Juvenile CPUE ranged from 1.1 to 6.2 fish.angler-hour^{-1} at RB and WC respectively.

Table 3.2. Mean length, percentage of mature population and GLM predicted CPUE for adult and juvenile santer in each study area.

Area	Bell Buoy	St Croix	Riy Banks	Evans	Bird Island	Woody Cape	Cape Padrone
Mean length	260±48	293±54	315±43	309±44	314±49	293±48	313±53
% Mature	19%	42%	65%	57%	57%	35%	56%
Predicted CPUE (adult)	0.4	1.3	2.3	3.2	3.2	3.6	2.5
Predicted CPUE (Juv.)	2.8	1.2	1.1	1.4	2.1	6.2	2.2

For the purposes of further detailed analysis the two groups were treated separately to minimise the between site variability and attempt to identify key factors influencing the community structure in each principal grouping.

Summary of key findings

- Reef study sites were identified and substratum confirmed in selected locations
- Reef linefish diversity varied spatially
- Two major reef fish communities were distinguished
- The dominant species, santer, was larger in Group 2 communities

(b) Principal Community Group 1

(i) Multivariate analysis of community structure

In order to standardise comparisons across all study areas, angling sites which were deemed to be placed on mixed rock and sand areas using the boat's colour echo sounder were excluded from the bay-wide comparison. Mixed rock and sand sites were only identified in the BB and StC areas and occur as a result of fragmented reef patches and large sand movement patterns in these areas of the bay. The sites excluded from the initial comparison were included in the assessment of communities in principal Group 1 and substrate type (rock or rock/sand) was included as an additional factor in the analyses.

No clear separation in the ordination of angling sites (Figure 3.8a) by area was evident, although some separation was apparent by depth category, with shallow and medium sites grouping towards the right and deeper sites towards the left of the plot area (Figure 3.8b). The abundance data of dominant species indicated some form of separation with santer, fransmadam and roman grouping together and occurring on the right hand side of the ordination, whereas white seacatfish and silver kob occurred on the left hand side, and red tjor-tjor occurred between the two different groupings, potentially indicating some form of community separation and habitat preference within Group 1 communities (Figure 3.9).

ANOSIM analyses revealed that area, substrate, season and depth all had significant effects on community structure (Table 3.3). Depth had the strongest effect; however, the magnitude of the differences for all factors (as indicated by the Global R value) was low and no clear separation was evident in the MDS ordination (Figure 3.8), suggesting a limited influence of these factors on community structure in Group 1 communities. The BIOENV procedure revealed a weak correlation between community structure and temperature ($r=0.039$).

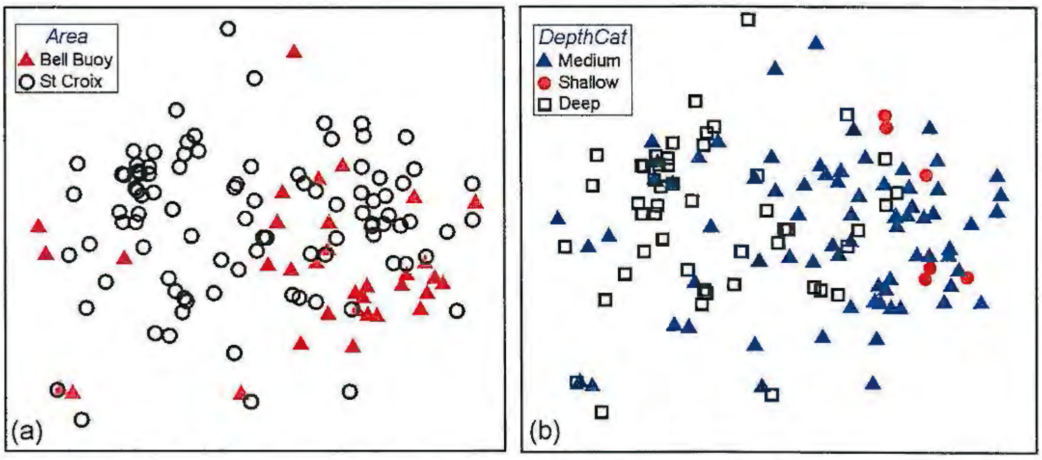


Figure 3.8. MDS ordination of angling sites by (a) area and (b) depth category in Principal Community Group 1 communities (Stress 0.15).

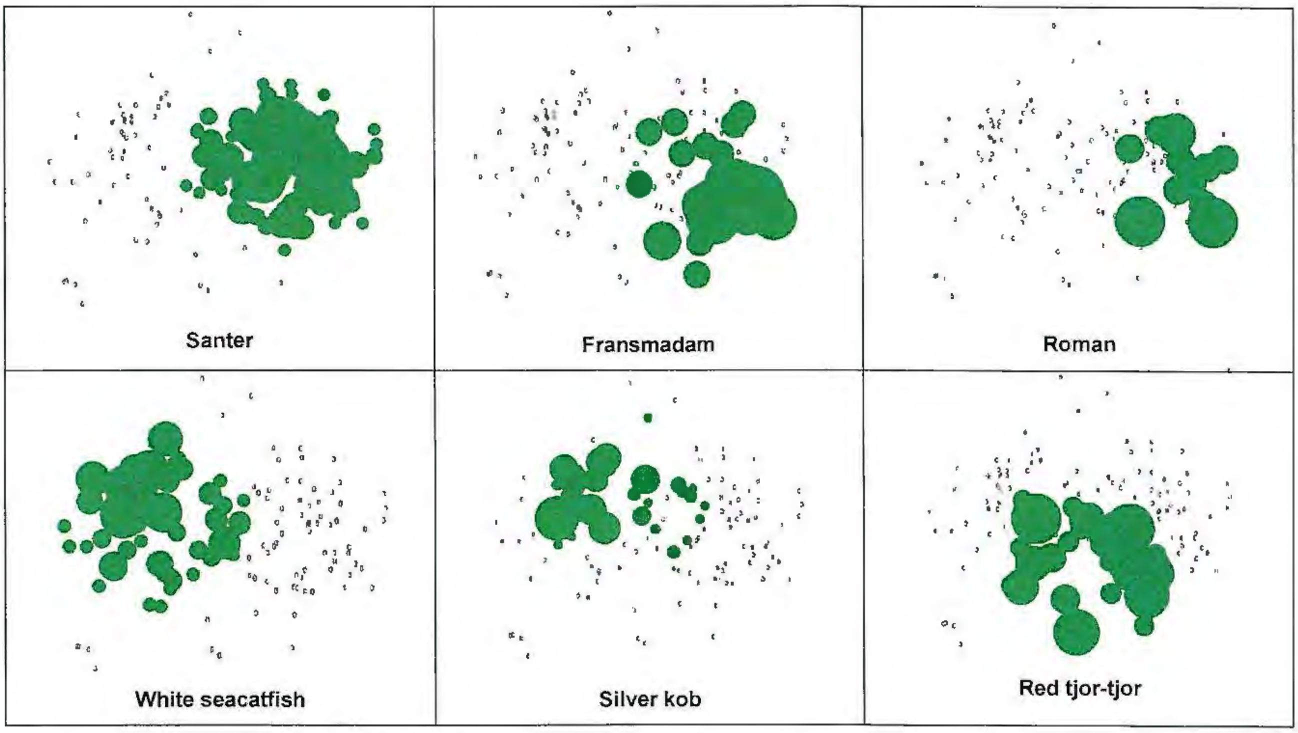


Figure 3.9. MDS ordination of Principal Community Group 1 with species bubble plots superimposed suggesting separation of sites based on individual species abundances (n=135). Larger bubbles indicate higher abundances.

Table 3.3. Results of ANOSIM of categorical factors on community structure in Principal Community Group 1.

Factor	Global R	p value	Significant pairwise comparisons (p<0.05)
Area	0.096	0.001 **	StC ≠ BB
Substrate	0.084	0.003 **	Rock ≠ rock/sand
Season	0.059	0.004 **	Summer ≠ winter
Depth category	0.184	0.001 **	Shallow ≠ Deep; Medium ≠ Deep
Period	0.015	0.150 n/s	-

n/s=not significant
* p<0.05
** p<0.01

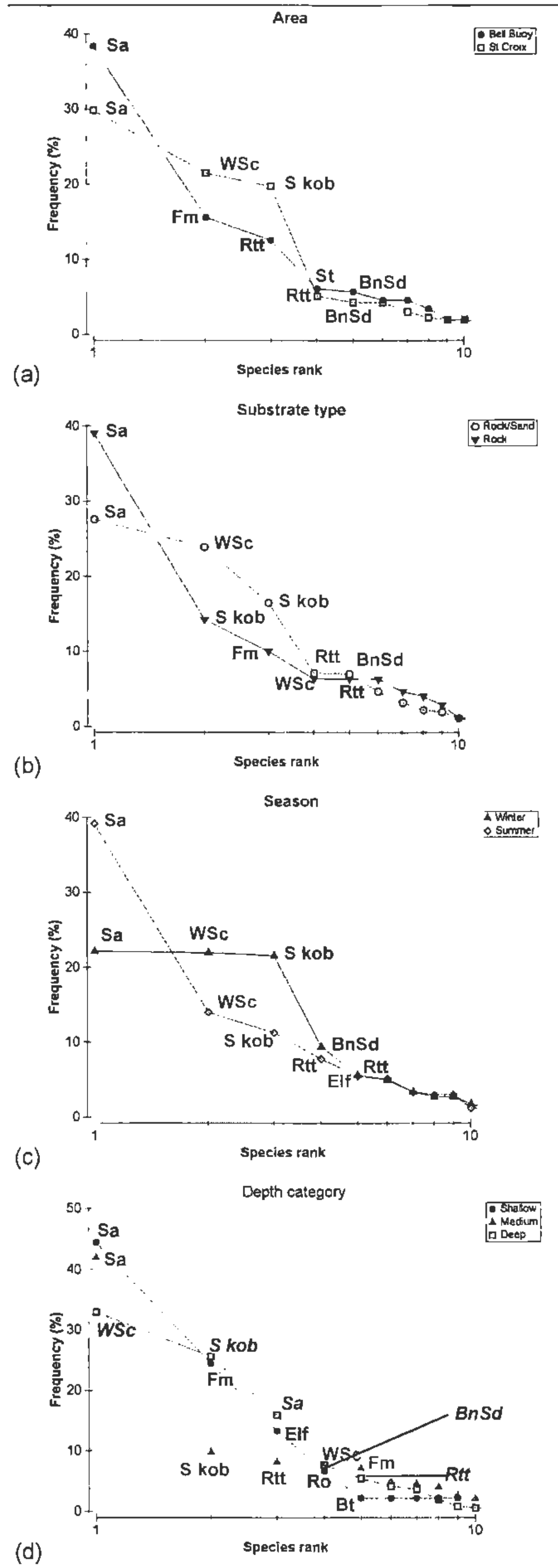


Figure 3.10. Influence of area, substrate, season and depth on community structure in Principal Community Group 1. (Sa=santer; WSc=white seacatfish; S kob=silver kob; Fm=fransmadam; Rtt=red tjor-tjor; St=steentjie; BnSd=bluntnose spiny dogfish).

Santer dominated both areas accounting for 38% and 30% of the catch in the BB and StC areas, respectively (Figure 3.10a). White seacatfish and silver kob accounted for 22% and 20% of the catch respectively in StC and although present in the BB community were not dominant species. Fransmadam (16%) and red tjor-tjor (13%) were the second and third most abundant species in the BB area.

Santer was the most abundant species over both rock (39%) and rock/sand (27%) substrate (Figure 3.10b). White seacatfish were more abundant over rock/sand (24%) than rock (6%), while silver kob (16% rock/sand and 14% rock) and red tjor-tjor (7% rock/sand and 6% rock) accounted for similar proportions in each habitat type. Fransmadam did not contribute significantly to the catch over rock/sand substrate while comprising 10% over rock. Bluntnose spiny dogfish contributed 7% to the catch in rock/sand areas but not significantly to communities over rock.

Santer was the most dominant species in summer accounting for 37% of the catch (Figure 3.10c) but decreased in winter when the abundance of white seacatfish and silver kob increased (22%).

Santer dominated the shallow (42%) and medium (44%) depth ranges with fransmadam (24%) and elf (13%) being second and third most abundant in the shallow and silver kob (10%) and red tjor-tjor (8%) in the medium depth range (Figure 3.10d). White seacatfish (33%) and silver kob (26%) were the two most dominant species at the deeper depth ranges followed by santer (16%).

(ii) Trends in measures of diversity

Area, season, period and depth category did not have an influence on the species diversity in principal Group 1 (Table 3.4). Substrate had a significant effect on the number of species (p=0.006) and the Shannon-Wiener Diversity Index (p=0.041) with both being higher over rock than rock/sand substrates. No correlations between temperature and diversity were significant.

Table 3.4. Influence of categorical and continuous factors on four diversity indices in Principal Community Group 1. Cells highlighted in green indicated significant differences at p<0.05.

Factor	Species richness	Pielou's Evenness	Shannon-Wiener	Taxonomic Diversity
Area	p=0.244 n/s	p=0.081 n/s	p=0.085 n/s	p=0.157 n/s
Substrate	p=0.006 * R>R/S	p=0.722 n/s	p=0.041 * R>R/S	p=0.059 n/s
Season	p=0.771 n/s	p=0.449 n/s	p=0.708 n/s	p=0.914 n/s
Period	p=0.559 n/s	p=0.939 n/s	p=0.728 n/s	p=0.997 n/s
Depth category	p=0.076 n/s	p=0.599 n/s	p=0.051 n/s	p=0.098 n/s
Temperature	p=0.758 n/s	p=0.719 n/s	p=0.831 n/s	p=0.279 n/s

n/s=not significant
* p<0.05
** p<0.001

Table 3.5. Influence of factors on the relative abundance (CPUE) of linefish in Principal Community Group 1 sampled at 135 angling sites. Factors not listed were excluded from the GLMs based on preceding AIC analysis.

Effect	df	*W*	p
All species (n=1161)			
Intercept	1	0.06	0.814 n/s
Year	3	10.23	0.017 *
Season	1	3.35	0.067 n/s
Period	2	5.72	0.057 n/s
Substrate	1	9.24	0.002 *
Temperature	1	2.64	0.104 n/s
Depth	1	3.16	0.075 n/s
All santer (n=369)			
Intercept	1	2.24	0.134 n/s
Year	3	2.71	0.438 n/s
Season	1	2.16	0.142 n/s
Period	2	7.53	0.023 *
Substrate	1	4.57	0.032 *
Temperature	1	4.41	0.036 *
Depth	1	0.51	0.475 n/s
Juvenile santer (n=234)			
Intercept	1	9.48	0.002 *
Area	1	9.24	0.002 *
Period	2	5.20	0.074 n/s
Substrate	1	3.95	0.047 *
Temperature	1	13.52	<0.001 **
Adult santer (n=135)			
Intercept	1	1.42	0.233 n/s
Year	3	4.77	0.189 n/s
Area	1	7.10	0.008 *
Season	1	3.88	0.051 n/s
Period	2	6.57	0.037 *
Substrate	1	5.37	0.020 *
Temperature	1	2.02	0.155 n/s
Depth	1	1.47	0.225 n/s

n/s=not significant
* p<0.05
** p<0.001

(iii) Trends in relative abundance (CPUE)

Results from GLM analyses revealed that CPUE for all species combined was influenced significantly by year and substrate (Table 3.5). Mean CPUE of all species declined significantly from 11.0 fish.hour^{-1} in 2006 to 7.8 fish.hour^{-1} and 8.1 fish.hour^{-1} in 2007 and 2008 respectively, with a subsequent increase in 2009 to 10.5 fish.hour^{-1} (Figure 3.11a). CPUE for all species was higher over rock (10.7 fish.hour^{-1}) than rock/sand (8.0 fish.hour^{-1}) substrate (Figure 3.14a).

Period, substrate and temperature all had significant effects on the CPUE of all santer (Table 3.5). Santer CPUE over morning and midday periods was similar (3.6 fish.hour^{-1}) but declined to 1.5 fish.hour^{-1} during the afternoon sampling period (Figure 3.13). Santer CPUE was higher over rock (3.5 fish.hour^{-1}) than rock/sand (2.1 fish.hour^{-1}) substrate (Figure 3.14b) and increased with increasing water temperature (Figure 3.15a).

Juvenile santer accounted for 63% of the catch in Group 1 communities and both juvenile and adult santer CPUE was influenced by area (Table 3.5), with higher CPUE of juveniles at BB than StC, while adult CPUE was higher at StC than BB (Figure 3.12). The CPUE of juvenile and adult santer was significantly higher over rock than rock-sand substrate (Figure 3.14b). Whereas juvenile santer CPUE increased in warmer water (Figure 3.15a), adult santer CPUE was not influenced significantly.

Table 3.5. cont. Influence of factors on the relative abundance (CPUE) of linefish in Principal Community Group 1 sampled at 135 angling sites. Factors not listed were excluded from the GLMs based on preceding AIC analysis.

White seacatfish (n=202)			
Intercept	1	0.54	0.460 n/s
Year	3	9.95	0.019 *
Area	1	1.74	0.188 n/s
Period	2	4.24	0.120 n/s
Substrate	1	2.8	0.094 n/s
Temperature	1	0.67	0.412 n/s
Depth	1	8.05	0.004 *
All silver kob (n=182)			
Intercept	1	0.11	0.740 n/s
Year	3	20.78	<0.001 **
Area	1	1.37	0.241 n/s
Period	2	3.14	0.208 n/s
Temperature	1	1.51	0.218 n/s
Depth	1	0.8	0.371 n/s
Juvenile silver kob (n=67)			
Intercept	1	0.13	0.724 n/s
Year	3	13.41	0.004 *
Period	2	1.58	0.453 n/s
Substrate	1	0.63	0.427 n/s
Temperature	1	2.52	0.112 n/s
Depth	1	4.54	0.033 *
Adult silver kob (n=115)			
Intercept	1	3.85	0.049 *
Year	3	9.35	0.025 *
Area	1	1.94	0.163 n/s
Period	2	3.21	0.200 n/s

n/s=not significant
* p<0.05
** p<0.001

Table 3.6. Influence of factors on the length of linefish in Principal Community Group 1 sampled from 135 angling sites. Factors not listed were excluded from the GLMs based on preceding AIC analysis.

Effect	df	W	p
Santer (n=366)			
Intercept	1	6437.12	<0.001 **
Year	3	21.2	<0.001 **
Season	1	26.34	<0.001 **
Area	1	51.01	<0.001 **
White seacatfish (n=202)			
Intercept	1	309.84	<0.001 **
Period	2	8.41	0.015 *
Temperature	1	3.23	0.072 n/s
Silver kob (n=182)			
Intercept	1	63.56	<0.001 **
Year	3	10.56	0.014 *
Substrate	1	9.01	0.002 *
Depth		21.62	<0.001 **
Temperature	1	2.82	0.093 n/s

n/s=not significant
* p<0.05
** p<0.001

Year and depth had a significant influence on CPUE of white seacatfish (Table 3.5). CPUE of white seacatfish declined progressively from 1.3 fish.hour^{-1} in 2006 to 0.43 fish.hour^{-1} in 2009 (Figure 3.11b) and increased with increasing depth (Figure 3.15b). Only 3.4% of the white seacatfish landed were below the length at 50% maturity (Mann 2000).

The relative abundance of all silver kob was only influenced significantly by year (Table 3.5), decreasing progressively from 2006 to 2009 (Figure 3.11c). No juvenile kob were caught in the BB area, but 37% of the kob caught in the StC area were below the length at 50% maturity (Griffiths 1997c).

Relative abundance of juvenile and adult silver kob were both influenced significantly by year (Table 3.5) decreasing from 2006 to 2009 (Figure 3.11c). In addition juvenile kob CPUE was influenced significantly by depth decreasing with increasing depth (Figure 3.15c).

(iv) Trends in size structure

The mean length of santer was influenced by year, season and area (Table 3.6). Santer mean length ranged from 261 to 274mm from 2006 to 2008 and increased substantially to 305mm in 2009 (Figure 3.16a). Santer were larger in StC (304mm) than BB (251mm) (Figure 3.17). Captured santer were larger during winter with a mean length of 295mm compared to 260mm in summer (Figure 3.20).

Period influenced white seacatfish length significantly (Table 3.6). The mean size of individuals caught during midday (288mm FL) was smaller than that of those caught during the morning and afternoon (304mm and 306mm respectively) (Figure 3.19).

Year, substrate and depth were significant factors influencing the mean length of silver kob (Table 3.6). The mean length in 2006 (335mm FL) was less than in 2007, 2008 and 2009 (375mm, 357mm, 357mm respectively) (Figure 3.16b). Silver kob caught over rock (371mm FL) were larger than those caught over rock/sand (342mm FL) substrate (Figure 3.18) and mean fork length decreased with increasing depth (Figure 3.21).

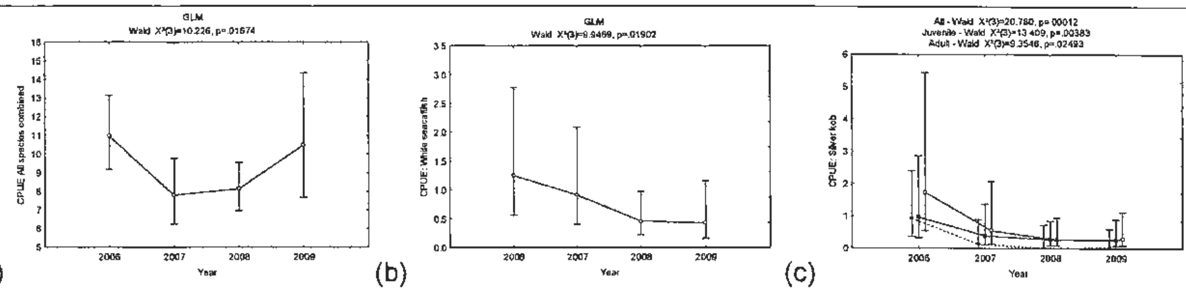


Figure 3.11. Influence of year on CPUE of (a) All species, (b) white seacatfish and (c) all silver kob (open circles), juvenile silver kob (closed circles dashed line) and adult silver kob CPUE (closed squares) in Principal Community Group 1.

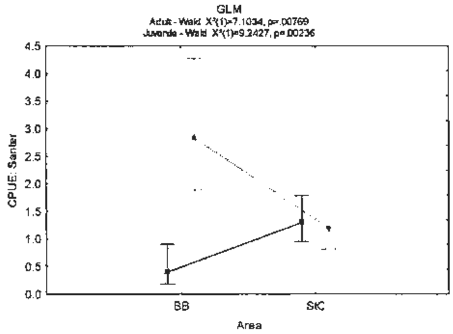


Figure 3.12. Influence of area on CPUE of adult (solid line and closed squared) and juvenile (dashed line) santer in Principal Community Group 1.

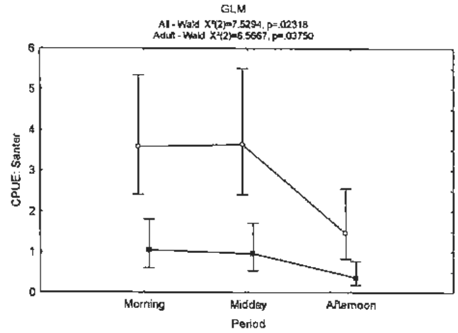


Figure 3.13. Influence of time period on the CPUE of all (open circles) and adult (closed squares) santer in Principal Community Group 1.

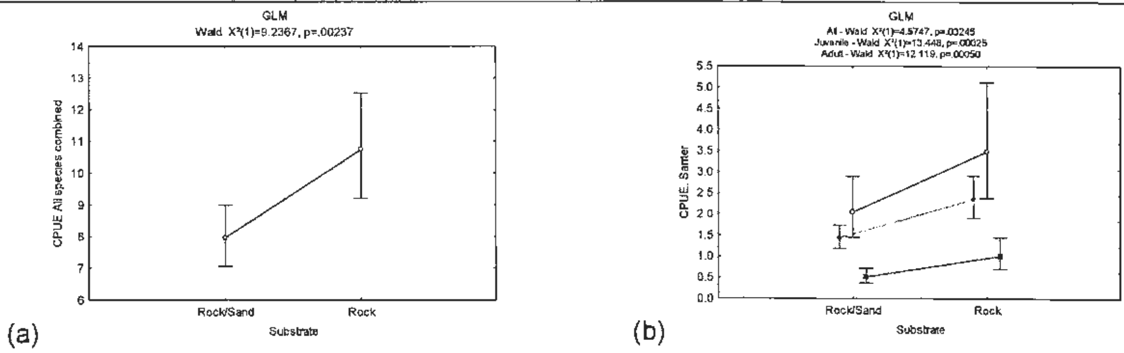


Figure 3.14. Influence of substrate on (a) all species CPUE and (b) all (open circles), adult (solid line and closed squares) and juvenile (dashed line) santer CPUE in Principal Community Group 1.

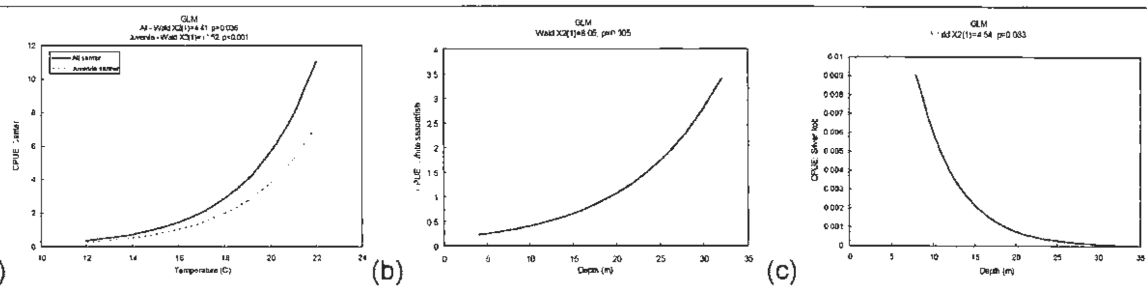


Figure 3.15. Influence of (a) temperature and santer CPUE, and (b) depth on white seacatfish and (c) silver kob CPUE in Principal Community Group 1.

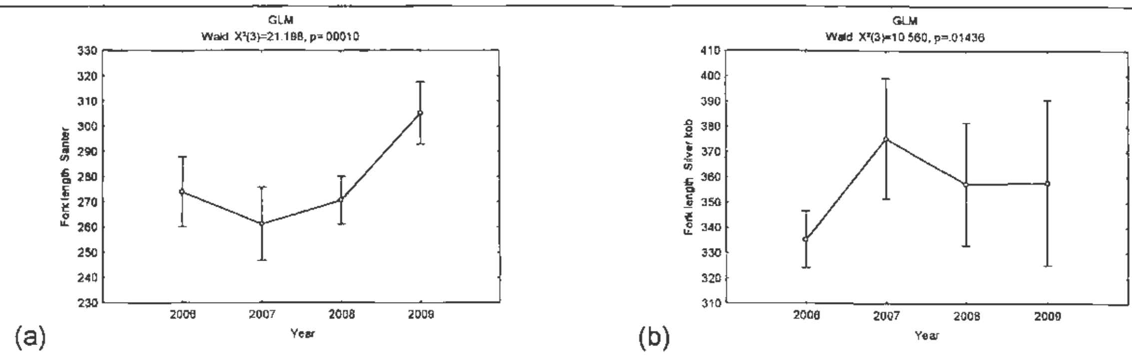


Figure 3.16. Influence of year on the fork length of (a) santer, and (b) silver kob in Principal Community Group 1.

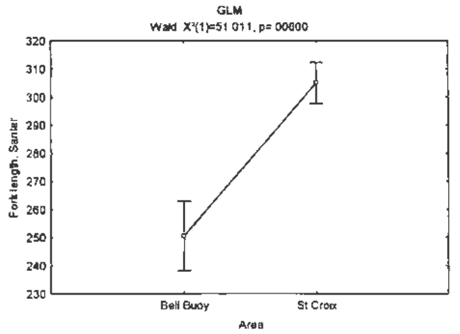


Figure 3.17. Influence of area on fork length of santer in Principal Community Group 1.

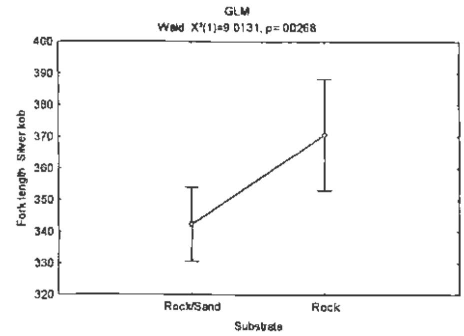


Figure 3.18. Influence of substrate on the fork length of silver kob in Principal Community Group 1.

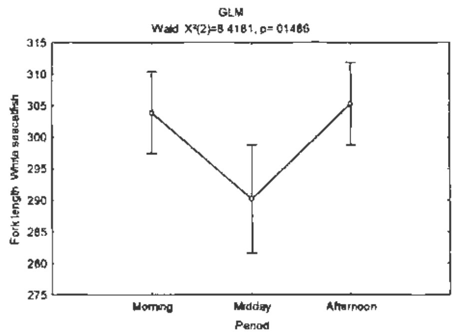


Figure 3.19. Influence of period on mean length of white seacatfish in Principal Community Group 1.

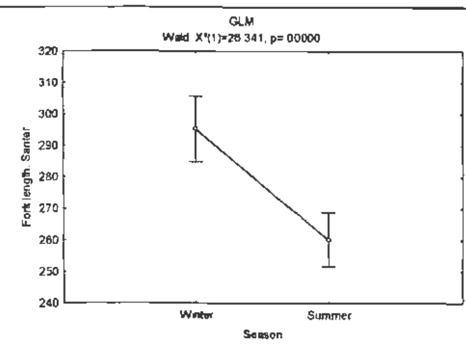


Figure 3.20. Influence of season on the mean fork length of santer in Principal Community Group 1.

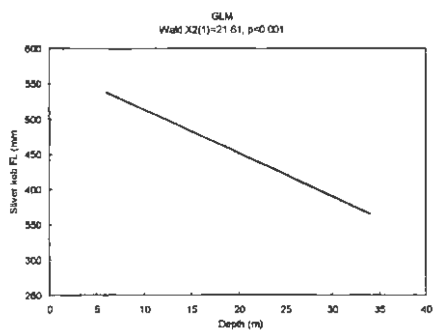


Figure 3.21. Influence of depth on the mean length of silver kob in Principal Community Group 1.

Summary of key findings – Community Group 1

- Explanatory factors had weak influence on Group 1 community structure
- Substrate type influenced diversity significantly
- High inter-annual variability in relative abundance of species
- Limited seasonal variability in relative abundance

(c) Principal Community Group 2

(i) Multivariate analysis of community structure

Angling sites in Group 2 showed no clear separation by area in the MDS ordination (Figure 3.22). However, the BI and RB sites are generally situated to the right while the CP and WC sites occur towards the left. This was confirmed by the groupings in the cluster dendogram where the two offshore reefs RB and BI, and the two inshore reefs WC and CP were most similar to each other (Figure 3.22).

The family Sparidae dominated the composition comprising between 94 and 98% of the catch by numbers in each area (Table 3.1). Santer was the most dominant and frequently encountered species being captured at 75% of the sites, while roman and fransmadam were only captured at 30% and 27% of the sites respectively (Figure 3.23; Figure 3.24). Species bubble plots suggest a spatial separation in the MDS ordination of fransmadam, roman, red stumpnose and scotsman from sites with higher santer abundance (Figure 3.25). Fransmadam was most abundant at sites where other species were scarce, while roman, scotsman and red stumpnose exhibit a high degree of overlap. Steentjie showed considerable overlap with both santer and fransmadam.

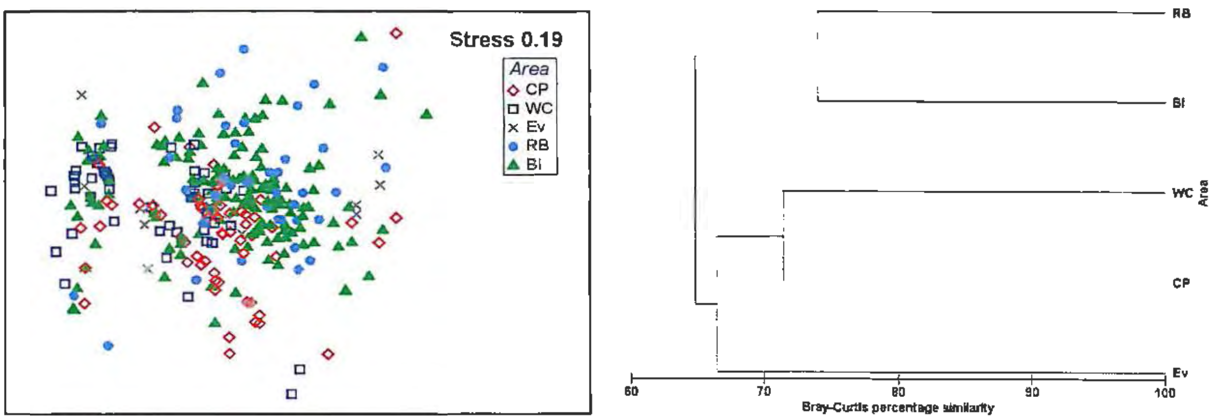


Figure 3.22. MDS ordination of angling sites in Principal Community Group 2 distinguished by area (left) and cluster dendogram illustrating closest groupings (right).

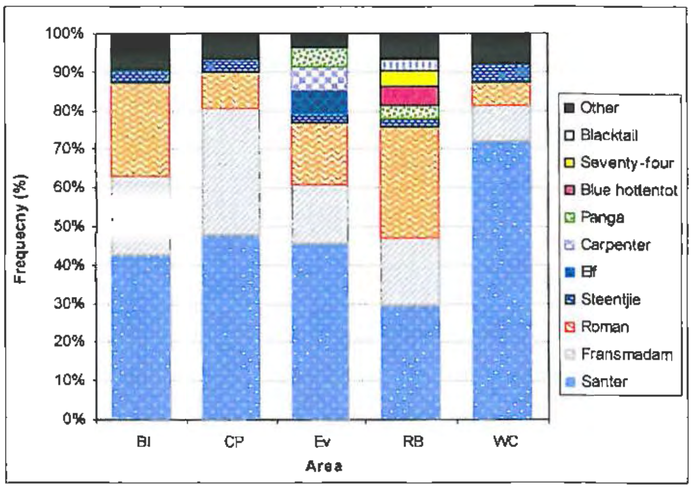


Figure 3.23. Dominance of species in each area based on abundance in Principal Community Group 2. (BI=Bird Island; CP=Cape Padrone; Ev=Evans; RB=Riy Banks; WC=Woody Cape).

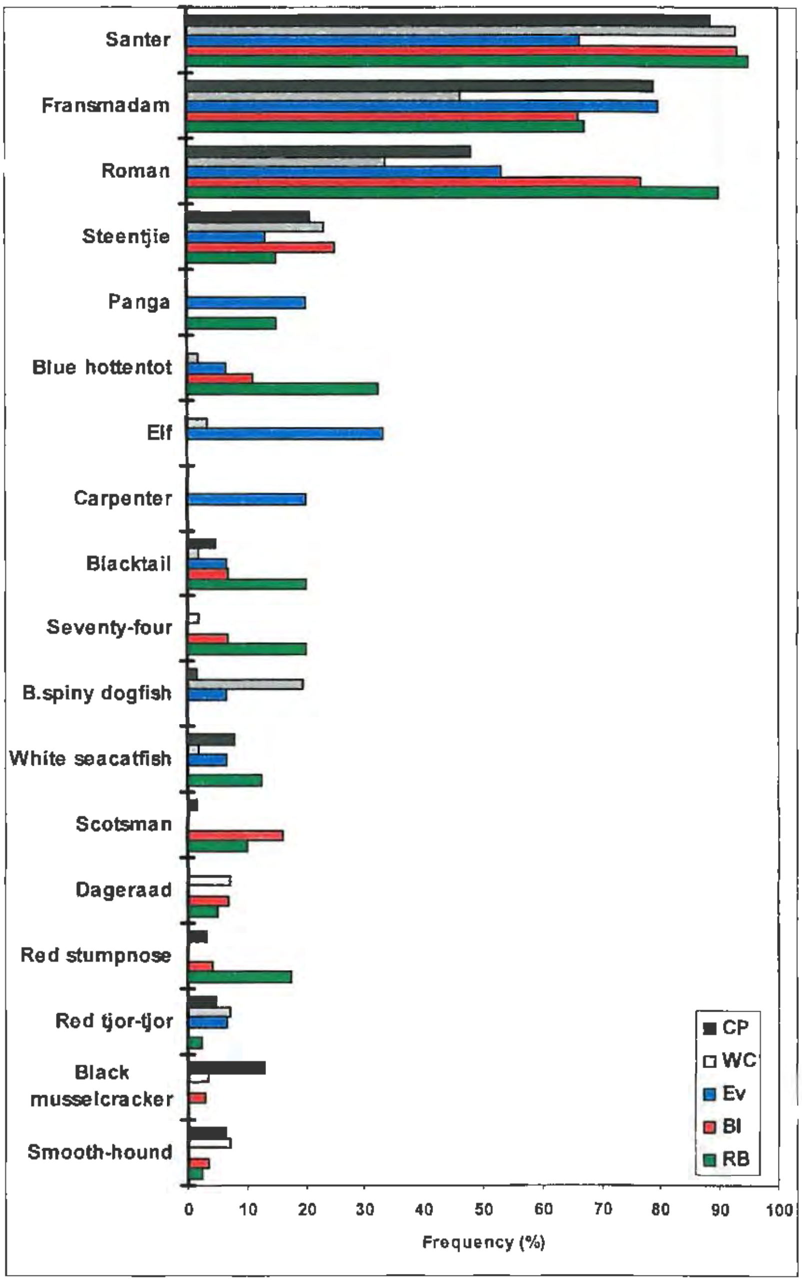


Figure 3.24. Frequency of main species in Principal Community Group 2 captured at angling sites within each area based on presence/absence data. (BI=Bird Island; CP=Cape Padrone; Ev=Evans; RB=Riy Banks; WC=Woody Cape).

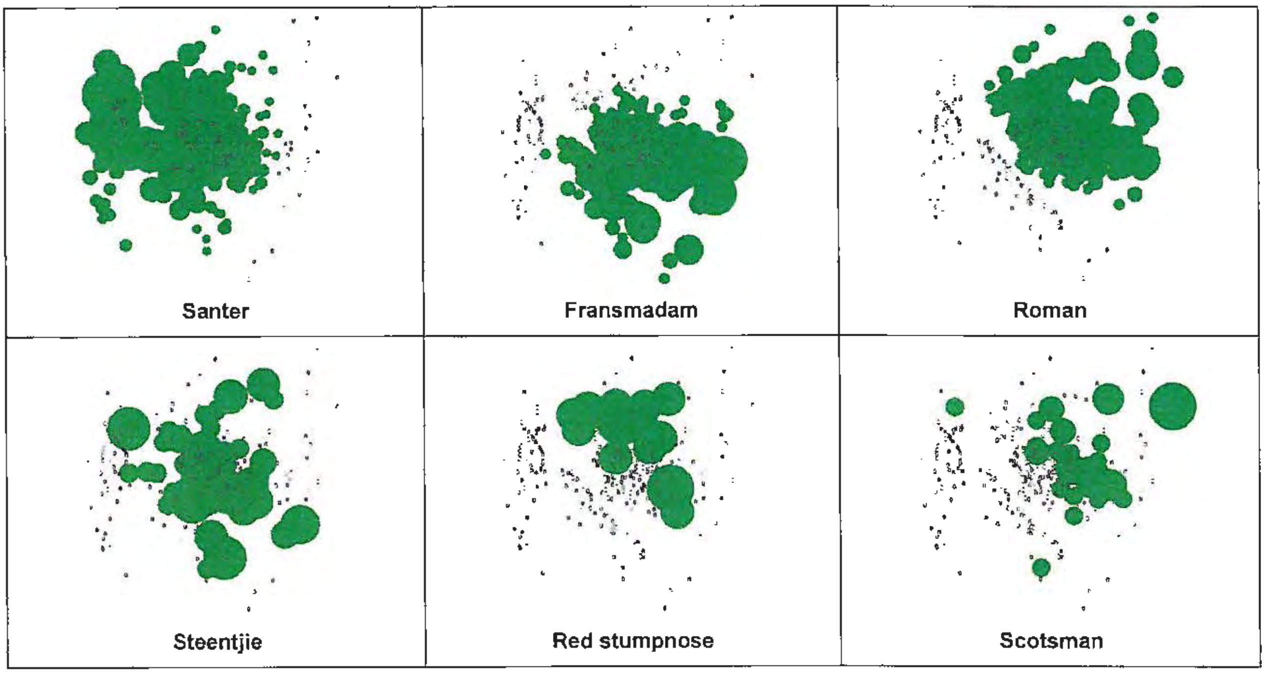


Figure 3.25. MDS ordination of Principal Community Group 2 with species bubble plots superimposed suggesting separation of sites based on individual species abundances (n=315). Larger bubbles indicate higher abundances.

Results of ANOSIM tests indicated that year, period and season did not have a significant effect on community structure within the Group 2 community (Table 3.7). Depth (p=0.001; Global R=0.089) and area had a significant effect (p=0.001; Global R=0.136), but the magnitude of effect caused by depth was small (Table 3.7). The BIOENV procedure revealed a weak correlation between community structure and temperature (r=0.136).

Spatially, pairwise comparisons indicated that WC was significantly different to all other sites (Table 3.8). The SIMPER routine indicated that santer contributed to between 24-30% of the dissimilarity between WC and other study areas, being far more dominant in the WC area where it accounted for 75% of the total catch. CP differed from both RB and Ev which was due to the higher abundances of roman and lower abundances of fransmadam at both RB and Ev. Differences in community structure between Ev and both BI and RB were due to lower abundances of fransmadam and roman and higher abundances of santer in Ev. BI communities were similar to both RB and CP. Cumulatively santer, roman and fransmadam accounted for between 76% and 90% of the catch in each area (Figure 3.23).

Table 3.7. Results of ANOSIM of categorical factors on community structure in Principal Community Group 2.

Factor	Global R	p value	Significant pairwise comparisons (p<0.05)
Area	0.136	0.001 **	See Table 3.8 below
Year	0.023	0.065 n/s	-
Season	0.009	0.057 n/s	-
Depth category	0.089	0.001 **	Medium ≠ Deep
Period	-0.003	0.58 n/s	-

n/s=not significant
* p<0.05
** p<0.001

Table 3.8. Results of pairwise ANOSIM comparisons between areas in Principal Community Group 2.

Area pairs	r statistic	p value
WCxRB	0.37	0.001 **
CPxRB	0.20	0.001 **
WCxCP	0.18	0.001 **
WCxBI	0.16	0.001 **
WCxEv	0.27	0.002 **
CPxEv	0.23	0.006 **
EvxRB	0.20	0.013 *
BIxEv	0.19	0.014 *
BIxCP	0.05	0.066 n/s
BIxRB	0.06	0.085 n/s

n/s=not significant
* p<0.05
** p<0.001

(ii) Trends in measures of diversity

There were significant differences between years in both species richness and Pielou's Evenness Index (Table 3.9). The number of species caught was lowest in 2006, increased in 2007 and 2008 but declined slightly in 2009 (Figure 3.26). Pielou's Evenness Index was lowest in 2007 and highest in 2009 (Figure 3.26). All diversity measures differed significantly by area (Table 3.9) and displayed similar trends with lowest mean values at WC and highest values at RB (Figure 3.28). The number of species was the only diversity metric which differed significantly by period with lowest number of species caught over midday and highest caught in the afternoon (Table 3.8; Figure 3.27). Season did not have a significant effect on any of the diversity indices, and there were no significant correlations between depth or water temperature with any of the diversity indices (Table 3.9).

Table 3.9. Influence of factors on measures of diversity in the Principal Community Group 2.

Factor	Number of species	Pielou's Evenness	Shannon Wiener	Taxonomic Diversity
Year	p=0.0295 * 2007 > 2006	p=0.0249 * not detected	p=0.4243 n/s	p=0.870 n/s
Area	p<0.001 ** RB > CP, BI, WC BI > WC	p=0.0295 * RB > WC	p=<0.001 ** RB > BI, CP, WC BI > WC	P<0.001 ** RB > BI, CP, WC
Season	p=0.591 n/s	p=0.289 n/s	p=0.540 n/s	p=0.467 n/s
Period	p=0.0108 * Afternoon > Midday	p=0.406 n/s	p=0.035 * Afternoon > Midday	p=0.305 n/s
Depth category	p=0.155 n/s	p=0.831 n/s	p=0.189 n/s	p=0.179 n/s
Temperature	p=0.201 n/s	p=0.061 n/s	p=0.631 n/s	p=0.765 n/s

n/s=not significant
* p<0.05
** p<0.001

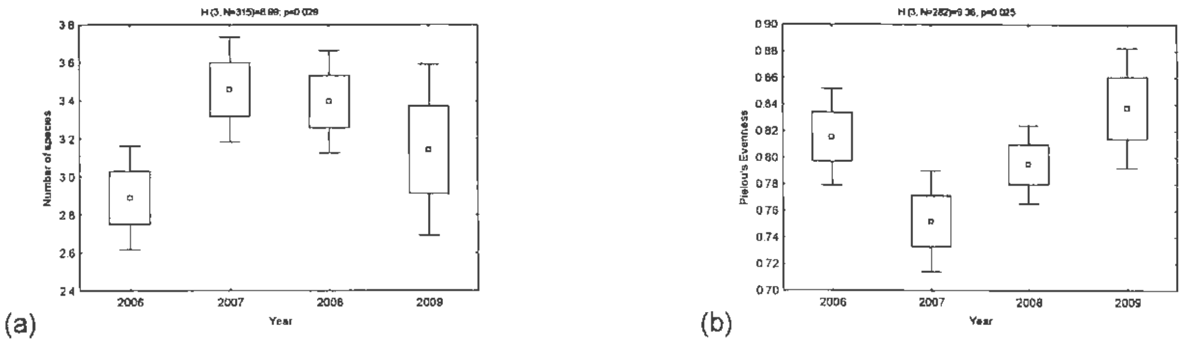


Figure 3.26. Annual differences in (a) Species richness, and (b) Pielou's Evenness Index for Principal Community Group 2.

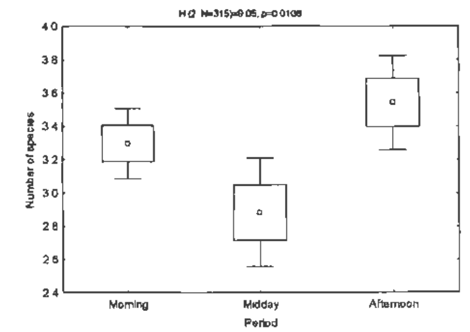


Figure 3.27. Influence of period on the number of species in Principal Community Group 2.

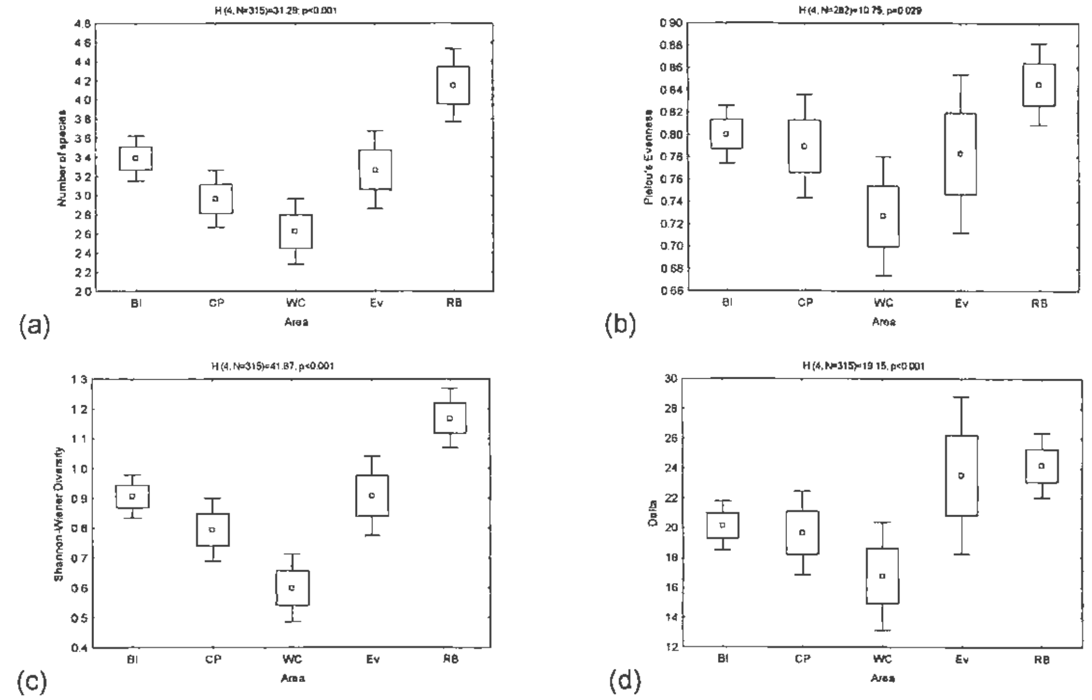


Figure 3.28. Differences in diversity by area in Principal Community Group 2, (a) Species richness, (b) Pielou's Evenness, (c) Shannon-Wiener Diversity, and (d) Taxonomic Diversity.

Table 3.10. Influence of factors on the relative abundance (CPUE) of linefish in Principal Community Group 2 sampled at 318 angling sites. Factors not listed were excluded from the GLMs based on preceding AIC analysis.

Effect	df	*W*	p
All species (n=3 888)			
Intercept	1	56.6	<0.001 **
Year	3	32.45	<0.001 **
Season	1	3.42	0.065 n/s
Area	4	15.8	0.003 *
Period	2	4.58	0.101 n/s
Depth	1	17.85	<0.001 **
Temperature	1	1.66	0.198 n/s
All Santer (n=1 904)			
Intercept	1	3.68	0.055 n/s
Year	3	21.18	<0.001 **
Area	4	64.22	<0.001 **
Temperature	1	2.91	0.088 n/s
Adult Santer (n=994)			
Intercept	1	61.55	<0.001 **
Year	3	19.13	<0.001 **
Area	4	7.34	0.119 n/s
Period	2	1.59	0.452 n/s
Depth	1	8.23	0.004 *
Juvenile Santer (n=911)			
Intercept	1	4.22	0.040 *
Year	3	15.53	0.001 *
Area	4	122.14	<0.001 **
Temperature	1	5.97	0.015 *
Depth	1	15.67	<0.001 **
Fransmadam (n=780)			
Intercept	1	117.1	<0.001 **
Year	3	6.24	0.101 n/s
Area	4	24.24	<0.001 **
Period	2	3.21	0.201 n/s
Depth	1	75.17	<0.001 **
Roman (n=689)			
Intercept	1	0.05	0.826 n/s
Season	1	25.11	<0.001 **
Area	4	64.72	<0.001 **
Depth	1	1.45	0.229 n/s
Temperature	1	1.01	0.314 n/s

n/s=not significant
* p<0.05
** p<0.001

(iii) Trends in relative abundance (CPUE)

Year ($p<0.001$), area ($p=0.003$) and depth ($p<0.001$) had significant effects on the CPUE of all species combined within the Group 2 community (Table 3.10). CPUE increased significantly from 9.2 fish.hour^{-1} in 2006 to 14.0 fish.hour^{-1} in 2007 and then decreased significantly in 2007 and 2008 to 12.4 fish.hour^{-1} and 11.5 fish.hour^{-1} respectively (Figure 3.29a). CPUE for all species at WC was significantly higher than at CP (Figure 3.30a) and decreased with increasing depth (Figure 3.31a).

Year ($p<0.001$) and area ($p<0.001$) influenced the CPUE of all santer significantly (Table 3.10). CPUE for santer showed similar trends to that of all species increasing significantly from 3.7 fish.hour^{-1} in 2006 to 6.7 fish.hour^{-1} in 2007 (Figure 3.29b). In addition CPUE in 2007 was significantly higher than 2008 (5.3 fish.hour^{-1}). The CPUE of all santer was significantly higher at WC than all other areas (Figure 3.30b). Of the santer landed 48% were below the size at 50% maturity (Mann 2000). Year ($p<0.001$) and depth ($p=0.004$) had a significant influence on adult santer abundance, with abundance decreasing with depth. Year ($p=0.001$), area ($p<0.001$), temperature ($p=0.015$) and depth ($p<0.001$) influenced the relative abundance of juvenile santer significantly. Juvenile santer abundance was significantly higher at WC (6.2 fish.hour^{-1}) than all other areas (1.1-2.2 fish.hour^{-1}) and increased with increasing depth (Figure 3.31b).

Area ($p<0.001$) and depth ($p<0.001$) were significant predictors of fransmadam CPUE (Table 3.10). Fransmadam CPUE was significantly higher at BI (1.6 fish.hour^{-1}) and CP (2.6 fish.hour^{-1}) than WC (0.9 fish.hour^{-1})(Figure 3.30c), and decreased with increasing depth (Figure 3.31c).

Season ($p<0.001$) and area ($p<0.001$) were significant predictors of roman CPUE (Table 3.10). CPUE was significantly higher in winter (2.2 fish.hour^{-1}) than summer (1.2 fish.hour^{-1}) ($p<0.001$) and significantly higher in the RB (3.3 fish.hour^{-1}) and BI (2.7 fish.hour^{-1}) areas than both the CP (0.9 fish.hour^{-1}) and WC (0.6 fish.hour^{-1}) areas (Figure 3.30d). Only one roman caught during the survey was below the size at 50% maturity (Mann 2000).

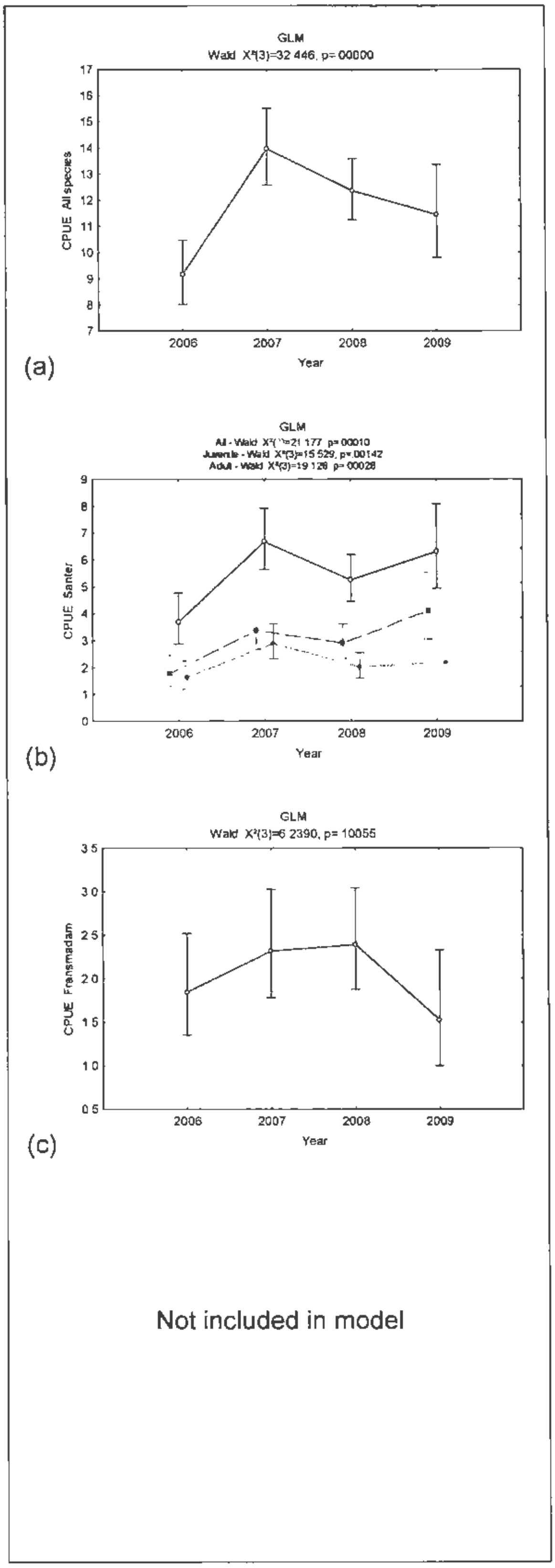


Figure 3.29. Influence of year on CPUE in Principal Community Group 2.

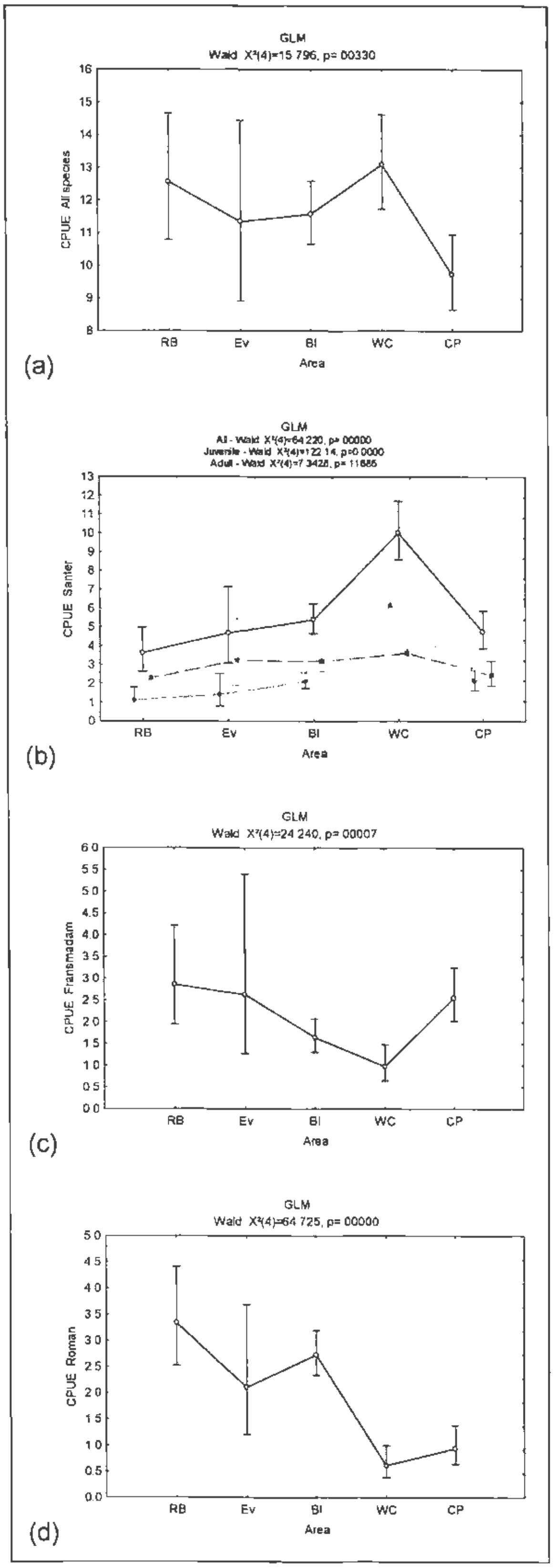


Figure 3.30. Influence of area on CPUE in Principal Community Group 2.

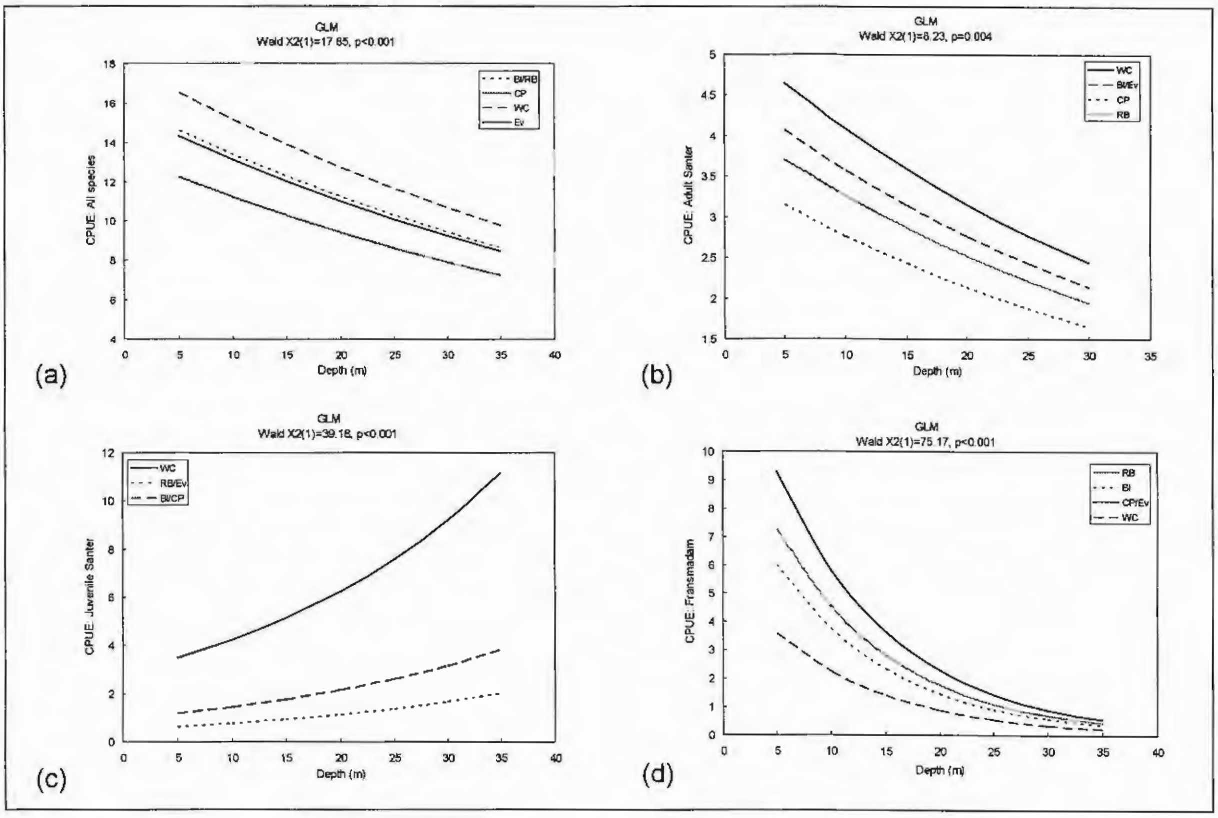


Figure 3.31. Influence of depth on CPUE of (a) all species and (b) adult santer, (c) juvenile santer and (d) fransmadam in Principal Community Group 2.

(iv) Trends in size structure

Year (p<0.001), area (p<0.001), period (p=0.041) and depth (p<0.001) were all significant predictors of santer length (Table 3.11). Mean santer length in 2007 (304mm) was significantly lower than in 2008 and 2009 (318mm and 314mm respectively) (Figure 3.32a). Mean santer length in WC (293mm) was significantly lower than in RB, Ev, BI and CP (312mm, 321mm, 314mm and 310mm respectively) (Figure 3.33a). Mean length decreased with increasing depth (Figure 3.34a). Santer caught during the midday (305mm) period were significantly smaller than during either the morning (313mm) or afternoon (311mm) periods (Figure 3.35).

Table 3.11. Influence of factors on the length of linefish in Principal Community Group 2 sampled from 318 angling sites. Factors not listed were excluded from the GLMs based on preceding AIC analysis.

Effect	df	W	p
Santer (n=1839)			
Intercept	1	5098.85	<0.001 **
Year	3	17.35	<0.001 **
Period	2	6.37	0.041 *
Area	4	67.62	<0.001 **
Depth	1	22.04	<0.001 **
Fransmadam (n=764)			
Intercept	1	23708.2	<0.001 **
Year	3	26.35	<0.001 **
Area	4	57.46	<0.001 **
Roman (n=683)			
Intercept	1	789.32	<0.001 **
Depth	1	6.3	0.012 *
Steentjie (n=128)			
Intercept	1	105.96	<0.001 **
Depth	1	7.81	0.005 *
Temperature	1	3.21	0.073 n/s

n/s=not significant
* p<0.05
** p<0.001

Both year (p<0.001) and area (p<0.001) had a significant influence on fransmadam length (Table 3.11). Fransmadam caught in 2006 (213mm) were significantly larger than in both 2007 and 2008 (202mm and 208mm respectively) (Figure 3.32b). Fransmadam caught in CP (218mm) were significantly larger than those caught in BI (207mm), Ev (200mm) and RB (199mm). Fransmadam caught in WC (213mm) were larger than in RB and Ev, and the mean size in BI was larger than in RB (Figure 3.33b).

Depth was the only factor which influenced roman (Figure 3.34b) and steentjie (Figure 3.34c) length significantly (Table 3.11), which decreased with increasing depth for both species.

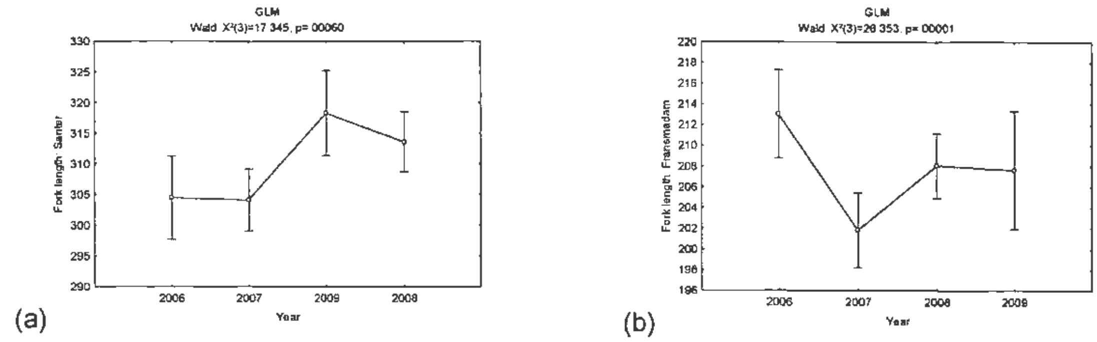


Figure 3.32. Influence of year on (a) santer, and (b) fransmadam length in Principal Community Group 2.

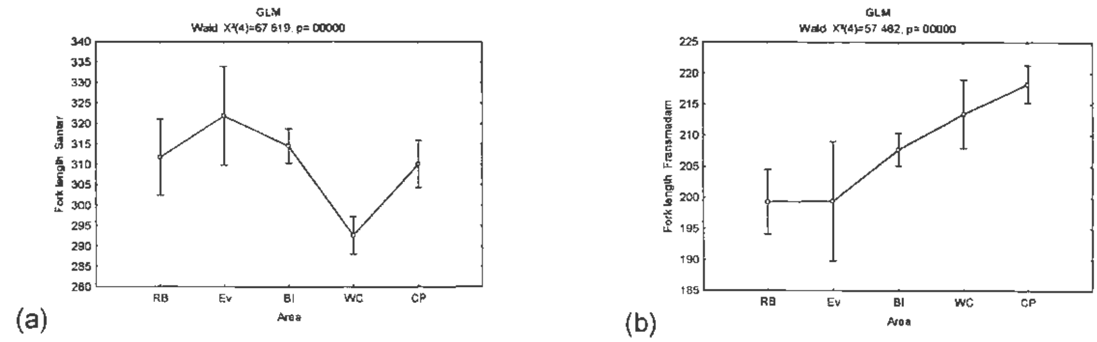
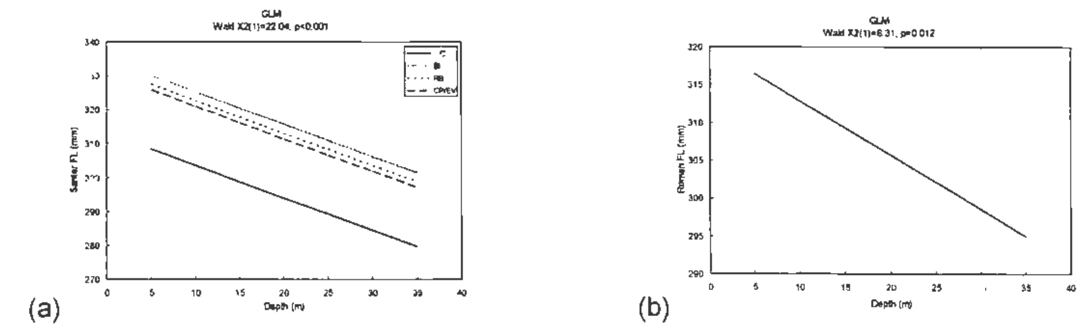


Figure 3.33. Influence of area on (a) santer, and (b) fransmadam length in Principal Community Group 2.



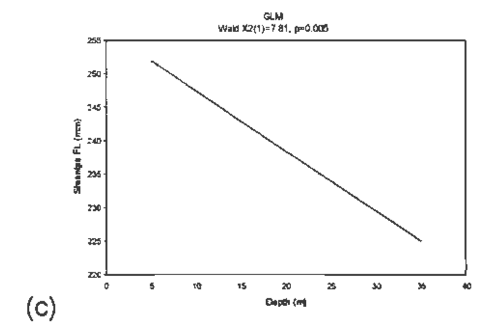


Figure 3.34. Influence of depth on (a) santer, (b) roman, and (c) steentjie length in Principal Community Group 2.

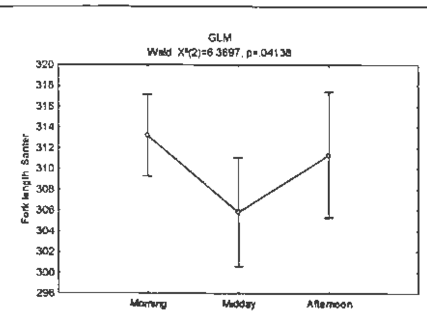


Figure 3.35. Influence of time period on santer length in Principal Community Group 2.

Summary of key findings – Community Group 2

- Reef dependent sparids dominated the community
- Some degree of spatial separation evident between offshore (RB and BB) and inshore (CP and WC) communities
- Spatial effects influenced diversity, with greater diversity in offshore areas
- Inter-annual variability in diversity and the relative abundance of species was high
- Location was the most significant predictor of relative abundance of species

Summary of key findings – Controlled angling survey

- Two distinct spatially separated reef fish communities were identified
- Relative abundance of all species was lower in Group 1 communities
- Santer populations were dominated by larger adult fish in Group 2 communities
- Several key biophysical factors influenced the relative abundance and the mean length of species within each community group, with location being the most important followed by substrate and depth (Table 3.12 and 3.13)

Table 3.12. Summary of key factors influencing diversity indices and multivariate statistics for reef fish communities.

Metric	Area		Substrate		Season		Period		Depth		Temp.	
	Grp1	Grp2	Grp1	Grp2	Grp1	Grp2	Grp1	Grp2	Grp1	Grp2	Grp1	Grp2
Species richness	×	✓	✓	n/a	×	×	×	✓	×	×	×	×
Shannon-Wiener Diversity	×	✓	✓	n/a	×	×	×	✓	×	×	×	×
Pielou's Evenness	×	✓	×	n/a	×	×	×	×	×	×	×	×
Taxonomic Diversity	×	✓	×	n/a	×	×	×	×	×	×	×	×
Multivariate	✓×	✓	✓×	n/a	✓×	×	×	×	✓	✓×	✓×	✓
Importance score	55		50		5		20		15		15	

Note: weak effects are indicated by both a tick and cross

Table 3.13. Summary of key factors influencing individual species abundance and mean size from controlled angling.

Species	Relative abundance						Mean size					
	Area	Subst.	Season	Period	Depth	Temp.	Area	Subst.	Season	Period	Depth	Temp.
Reef community Group 1												
All species	×	✓	×	×	×	×	n/a	n/a	n/a	n/a	n/a	n/a
Santer	×	✓	×	✓	×		✓	×	✓	×	×	×
White seacatfish	×	×	×	×	✓	×	×	×	×	✓	×	×
Silver kob	×	×	×	×	×	×	×	✓	×	×	✓	×
Reef community Group 2												
All species	✓	n/a	×	×	✓	×	n/a	n/a	n/a	n/a	n/a	n/a
Santer	✓	n/a	×	×	×	×	✓	×	×	✓	✓	×
Fransmadam	✓	n/a	×	×	✓	×	✓	×	×	×	×	×
Roman	✓	n/a	✓	×	×	×	×	×	×	×	✓	×
Importance score	50	50	13	13	38	0	50	17	17	33	50	0

(d) Assessment of linefish communities by UVC

A total of 93 dives were conducted during which 261 point counts were completed at 88 sites. In total 44 fish species from 16 families were observed of which 20 species and 9 families were not recorded in the controlled angling survey (Table 3.14). Poor diving conditions with water visibilities less than 3m for large periods of time resulted in 90% of the dives being conducted in the BI area with few conducted in the StC (6%), RB (3%) and WC (1%) areas. A summary of all fish species recorded is provided in Table 3.14 below.

Fransmadam dominated numerically in the BI (70.4%), RB (56.9%) and StC (36.8%) areas (Table 3.14). Steentjie was the second most dominant species in the BI area accounting for 8.8% followed by strepie (7.2%). Similarly steentjie was the second most dominant species in StC area followed by strepie (*Sarpa salpa*) which accounted for 27.1 and 6.9% respectively. Blue hottentot (26.7%) and barred fingerfin (*Cheilodactylus pixi*) (7.2%) were the second and third most dominant species in the RB area. Steentjie (53.3%) was dominant in the WC area, followed by santer (40.0%).

Due to the limited diving conducted in most study areas, spatial comparisons using UVC data were not possible and a detailed investigation of the influence of biophysical factors on community structure, relative abundance and size distribution of the ichthyofaunal community was undertaken using data from the BI area.

(i) Multivariate analysis of community structure

Although fransmadam were numerically more abundant, both roman and fransmadam were observed at 92.4% of the dive sites, followed by steentjie (69.6%), santer (68.4%), blue hottentot (62.0%) and twotone fingerfin (60.7%) (Figure 3.36). Blacktail, zebra, Cape stumpnose (*Rhabdosargus holubi)*, janbruin (*Gymnocrotaphus curvidens)*, strepie and Cape knifejaw (*Oplegnathus conwayi*) were observed at between 30 and 60% of the dive sites while red stumpnose, bronze bream (*Pachymetopon grande)*, scotsman, white musselcracker (*Sparodon durbanensis*) and barred fingerfin were recorded at between 10 and 30% of the dive sites.

ANOSIM tests indicated that season, depth and substrate resulted in significant differences in community structure, while rugosity, profile and period had no effect (Table 3.15). Although the seasonal effect was significant ($p=0.003$) the magnitude of the effect was low (Global R 0.09) and the seasonal differences between communities was considered minor. Depth on a categorical basis indicated strong differences between shallow and medium ($p=0.001$; r-statistic=0.237) and shallow and deep communities ($p=0.001$, r-statistic=0.419) while the BIOENV procedure indicated a weak correlation between depth as a continuous factor and community structure (r-value=0.258). Very weak correlations existed for temperature and visibility with r-values of 0.137 and 0.067 respectively.

The SIMPER procedure showed that the same 12 species, strepie, fransmadam, steentjie, blacktail, blue hottentot, zebra, santer, twotone fingerfin (*Chirodactylus brachydactylus*), Cape stumpnose, janbruin, Cape knifejaw and roman accounted for 80.2% and 78.1% of the dissimilarity between the medium and shallow, and the deep and shallow sites respectively. The same twelve species with the addition of red stumpnose accounted for 81.1% of the dissimilarity between substrate types (rock and rock-sand) and 81.4% of the dissimilarly between seasons.

Table 3.14. Fish species recorded during UVC (* indicates species also captured during controlled angling surveys; numbers in brackets=number of dives; number of point counts). Species contributing to greater than 4% of the community in each area highlighted in grey.

Class	Family	Scientific name	Common name	Bird Island (79; 231)		Riy Banks (3; 6)		St Croix (5; 20)		Woody Cape (1; 4)		Total (88; 261)	
				n	%	n	%	n	%	n	%	n	%
CHONDRICHTHYES	Odontaspididae	*Carcharias taurus*	Spotted ragged-tooth	4	0.0			1	0.1			5	0.0
	Scyliorhinidae	*Haploblepharus edwardsii*	Puffadder shyshark	1	0.0							1	0.0
		Poroderma africanum	Pyjama catshark *	3	0.0							3	0.0
		Poroderma pantherinum	Leopard catshark	1	0.0							1	0.0
	Squalidae	*Squalus megalops*	Bluntnose spiny dogfish *							1	6.7	1	0.0
OSTEICHTHYES	Ariidae	*Galeichthys feliceps*	White seacatfish *	1	0.0							1	0.0
	Carangidae	*Carangoides gymnostethus*	Bludger	5	0.0							5	0.0
		Seriola lalandi	Giant yellowtail *	36	0.2							36	0.2
		Seriola rivoliana	Longfin yellowtail *					4	0.5			5	0.0
	Chaetodontidae	*Chaetodon marleyi*	Doublesash butterflyfish	1	0.0			2	0.3			3	0.0
	Cheilodactylidae	*Cheilodactylus fasciatus*	Redfingers	10	0.0	2	0.3					12	0.1
		Cheilodactylus pixi	Barred fingerfin	17	0.1	46	7.2	4	0.5			75	0.3
		Chirodactylus brachydactylus	Twotone fingerfin	206	0.9	21	3.3	25	3.1			259	1.1
		Chirodactylus grandis	Bank steenbras	2	0.0							2	0.0
	Dichistiidae	*Dichistius capensis*	Galjoen	3	0.0							3	0.0
	Haemulidae	*Pomadasys striatum*	Striped grunter					15	1.9			17	0.1
	Mullidae	*Parupeneus rubescens*	Blacksaddle goatfish	25	0.1			1	0.1			26	0.1
	Oplegnathidae	*Oplegnathus conwayi*	Cape knifejaw	77	0.3			1	0.1			78	0.3
	Parascorpididae	*Parascorpis typus*	Jutjaw	1	0.0			2	0.3			3	0.0
	Serranidae	*Epinephelus marginatus*	Yellowbelly rockcod *	2	0.0							2	0.0
	Sparidae	*Boopsoidea inornata*	Fransmadam *	15 737	70.4	366	56.9	294	36.8			16 561	68.7
		Cheimerius nufar	Santer *	252	1.1			16	2.0	6	40.0	277	1.1
		Chrysoblephus anglicus	Englishman *	4	0.0							4	0.0
		Chrysoblephus gibbiceps	Red stumpnose *	46	0.2	1	0.2					47	0.2
		Chrysoblephus laticeps	Roman *	588	2.6	22	3.4	40	5.0			661	2.7
		Cymatoceps nasutus	Black musselcracker *	8	0.0							8	0.0
		Diplodus cervinus hottentotus	Zebra *	205	0.9	5	0.8	23	2.9			238	1.0
		Diplodus sargus capensis	Blacktail *	301	1.3	1	0.2	19	2.4			325	1.3
		Gymnocrotaphus curvidens	Janbruin	74	0.3	2	0.3	3	0.4			80	0.3
		Lithognathus mormyrus	Sand steenbras *					1	0.1			1	0.0
		Pachymetopon aeneum	Blue hottentot *	948	4.2	172	26.7	40	5.0			1 196	5.0
		Pachymetopon grande	Bronze bream	31	0.1							31	0.1
		Pagellus natalensis	Red tjor-tjor *					1	0.1			1	0.0
		Petrus rupestris	Red steenbras *	13	0.1			3	0.4			16	0.1
		Polysteganus praeorbitalis	Scotsman *	27	0.1	1	0.2					28	0.1
		Polysteganus undulosus	Seventy-four *	3	0.0							3	0.0
		Porcostoma dentata	Dane	3	0.0	1	0.2	2	0.3			6	0.0
		Rhabdosargus globiceps	White stumpnose *					2	0.3			2	0.0
		Rhabdosargus holubi	Cape stumpnose	110	0.5	1	0.2	28	3.5			143	0.6
		Sarpa salpa	Strepie *	1 612	7.2	2	0.3	55	6.9			1 683	7.0
		Sparodon durbanensis	White musselcracker	24	0.1							24	0.1
		Spondyliosoma emarginatum	Steentjie *	1 961	8.8			216	27.1	8	53.3	2 221	9.2
	Tetraodontidae	*Amblyrhynchotes honckenii*	Evileye puffer *	5	0.0							5	0.0
	Zanclidae	*Zanclus canescens*	Moorish idol	2	0.0							2	0.0
			TOTAL individuals	22 349	100	643	100	798	100	15	100	24 105	100
			TOTAL species	38	86	14	32	24	55	3	7	44	100

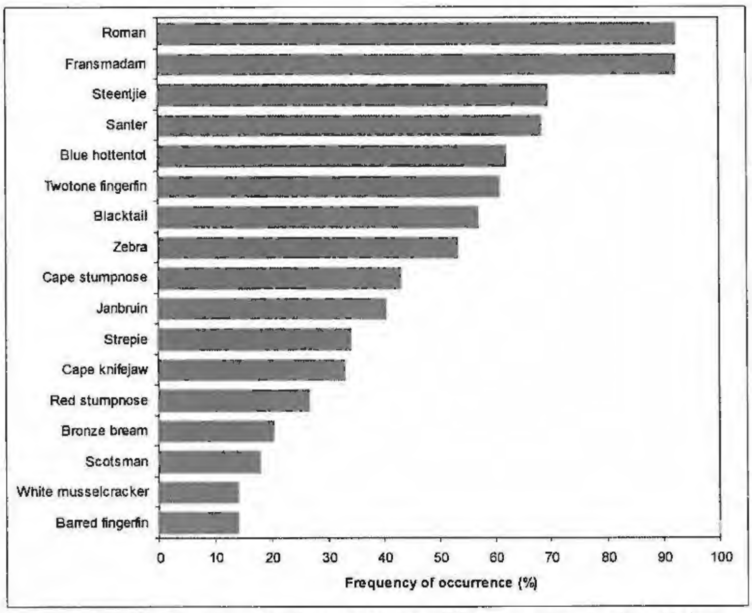


Figure 3.36. Frequency of occurrence of fish species during diving surveys in the BI area.

Table 3.15. Influence of factors on communities resulting from ANOSIM.

Factor	Global R	p value	Significant pairwise comparisons (p<0.05)
Season	0.09	0.003 *	Summer ≠ winter
Period	-0.007	0.569 n/s	-
Depth category	0.243	0.001 **	Medium ≠ Shallow; Deep ≠ Shallow
Profile	-0.051	0.889 n/s	-
Substrate	0.206	0.001 **	rock ≠ rock-sand
Rugosity	-0.115	0.955 n/s	-

n/s=not significant
* p<0.05
** p<0.001

Table 3.16. Influence of biophysical factors on the diversity of reef ichthyofauna during UVC.

Factor	Species richness	Pielou's Evenness	Shannon-Wiener	Taxonomic Diversity
Profile	p=0.018 * High > Low	p=0.025 * Low > High	p=0.635 n/s	p=0.163 n/s
Rugosity	p=0.008 ** High > Low	p=0.008 ** Low > High	p=0.279 n/s	p=0.054 n/s
Substrate	p=0.121 n/s	p=0.034 * Rock-sand > Rock	p=0.088 n/s	p=0.061 n/s
Season	p=0.001 ** Winter > Summer	p=0.877 n/s	p=0.020 * Winter > Summer	p=0.151 n/s
Period	p=0.028 * Afternoon > Midday	p=0.119 n/s	p=0.643 n/s	p=0.403 n/s
Depth	p=0.182 n/s	p=0.059 n/s	p=0.186 n/s	p=0.168 n/s
Temp.	p=0.008 ** r=0.299 (+ve)	p=0.716 n/s	p=0.117 n/s	p=0.216 n/s

n/s=not significant
* p<0.05
** p<0.001

(ii) Trends in measures of diversity

Reef profile and rugosity influenced the number of species (p=0.018; p=0.008) observed during UVC and Pielou's Evenness Index (p=0.025; p=0.008) significantly, but not the Shannon-Wiener (p=0.635; p=0.279) or Taxonomic Diversity indices (p=0.163; p=0.054) (Table 3.16). The number of species was greater over high profile and high rugosity reef and evenness was greater over low profile and low rugosity reef indicating the dominance of certain species over high profile, high rugosity reef. Pielou's Evenness was the only diversity metric influenced significantly by substrate type with (p=0.034) greater evenness of species over rock-sand substrate than solid rock. Season influenced both the number of species (p=0.001) and the Shannon-Weiner Diversity Index (p=0.020) significantly with more diverse fish communities observed during the winter months. Period was only a significant factor for the number of species (p=0.028) with larger numbers of species observed during the afternoon than midday sample periods. Depth did not influence diversity significantly and there was a weak positive correlation between the number of species and water temperature (p=0.008; r=0.299) but no relationship between temperature and any of the other diversity indices.

Table 3.17. Effect of factors on fish counts from 88 dives and 261 point counts. Factors not listed were excluded from the GLMs based on preceding AIC analysis.

Effect	df	*W*	p
Fransmadam (n=5 802)			
Intercept	1	10.03	0.002 *
Season	1	1.37	0.242 n/s
Period	2	2.42	0.299 n/s
Profile	1	3.02	0.082 n/s
Substrate	1	4.09	0.043 *
Rugosity	1	2.59	0.107 n/s
Temperature	1	0.16	0.687 n/s
Visibility	1	0.19	0.661 n/s
Depth	1	10.3	0.001 *
Steentjie (n=626)			
Intercept	1	2.35	0.125 n/s
Season	1	4.37	0.037 *
Period	2	2.91	0.232 n/s
Substrate	1	3.62	0.057 n/s
Rugosity	1	0.14	0.704 n/s
Temperature	1	4.04	0.045 *
Visibility	1	0.29	0.587 n/s
Depth	1	8.25	0.004 *
Strepie (n=605)			
Intercept	1	0.16	0.688 n/s
Season	1	0.28	0.594 n/s
Period	2	0.23	0.893 n/s
Profile	1	1.06	0.303 n/s
Rugosity	1	0.3	0.584 n/s
Temperature	1	0.76	0.384 n/s
Visibility	1	0.59	0.441 n/s
Depth	1	14.21	<0.001 **
Blue hottentot (n=321)			
Intercept	1	2.15	0.143 n/s
Season	1	4.64	0.031 *
Period	2	4.93	0.085 n/s
Profile	1	0.42	0.518 n/s
Substrate	1	0.5	0.481 n/s
Rugosity	1	6.7	0.010 *
Temperature	1	1.15	0.238 n/s
Visibility	1	3.39	0.065 n/s
Depth	1	8.24	0.004 *
Roman (n=202)			
Intercept	1	1.84	0.175 n/s
Season	1	7.9	0.005 *
Rugosity	1	13.78	<0.001 **
Visibility	1	2.12	0.146 n/s

(iii) Trends in relative abundance

Depth was a significant factor influencing the relative abundances of six of the nine most dominant species during UVC. Only roman, santer and twotone fingerfin were not influenced by depth (Table 3.17). Fransmadam, strepie, blacktail and zebra abundance decreased with increasing depth while the opposite was found for steentjie and blue hottentot.

Season had a significant influence on the abundance of steentjie (p=0.037) (Figure 3.37a), blue hottentot (p=0.031) (Figure 37b), roman (p=0.005) (Figure 37c) and twotone fingerfin (p=0.008) (Figure 37d) (Table 17) with the abundance of all these species being greater over the winter months.

Period influenced the abundance of santer (p=0.048) (Figure 3.38a) and zebra (p=0.009) (Table 3.17; Figure 3.38b) significantly with santer observed in higher abundances over midday than the morning and afternoon, while the opposite was recorded for zebra.

Profile was only a significant factor in predicting twotone fingerfin (p=0.020) abundance which was greater over high profile than low profile reefs (Table 3.17).

Fransmadam (p=0.043), twotone fingerfin (p=0.004) and zebra (p=0.016) were influenced significantly by substrate type (Table 3.17), with fransmadam and twotone fingerfin being more abundant over rock than rock-sand substrates while zebra were more abundant over rock-sand (Figure 3.39a-c).

Both blue hottentot (p=0.010) and roman (p<0.001) were recorded in greater abundances over high rugosity reef areas than low rugosity areas (Table 3.17; Figure 3.40a and b).

Water temperature was a significant predictor of steentjie abundance (p=0.045), with cooler temperatures leading to higher abundance of steentjie (Table 3.17). Visibility influenced the abundance of santer significantly (p=0.038) which increased with increasing visibility.

Table 3.17. cont. Effect of factors on fish counts from 88 dives and 261 point counts. Factors not listed were excluded from the GLMs based on preceding AIC analysis.

Blacktail (n=116)			
Intercept	1	0.25	0.62 n/s
Season	1	2.27	0.132 n/s
Temperature	1	2.81	0.094 n/s
Depth	1	61.24	<0.001 **
Santer (n=86)			
Intercept	1	7.16	0.007 *
Season	1	2.59	0.107 n/s
Period	2	6.08	0.048 *
Temperature	1	3.49	0.062 n/s
Visibility	1	4.31	0.038 *
Twotone fingerfin (n=70)			
Intercept	1	14.3	<0.001 **
Season	1	6.91	0.008 *
Profile	1	5.42	0.020 *
Substrate	1	8.2	0.004 *
Zebra (n=67)			
Intercept	1	9.47	0.002 *
Season	1	3.7	0.054 n/s
Period	2	9.39	0.009 *
Profile	1	2.47	0.116 n/s
Substrate	1	5.85	0.016 *
Depth	1	14.68	<0.001 **

n/s=not significant
* p<0.05
** p<0.001

Table 3.18. Effect of factors on roman length from 88 dives and 261 point counts. Factors not listed were excluded from the GLMs based on preceding AIC analysis.

Effect	df	*W*	p
Roman (n=497)			
Intercept	1	525.41	<0.001 **
Season	1	12.06	<0.001 **
Profile	1	7.12	0.008 *
Rugosity	1	9.25	0.002 *
Depth	1	36.17	<0.001 **

n/s=not significant
* p<0.05
** p<0.001

(iv) Trends in size structure

Season ($p<0.001$) had a significant effect on roman length with larger fish encountered over the winter months (Table 3.18). Reef profile ($p=0.008$) and rugosity ($p=0.002$) both had a significant influence on roman length with larger fish observed over high profile and high rugosity reefs. Roman length was also shown to decrease with increasing depth ($p<0.001$).

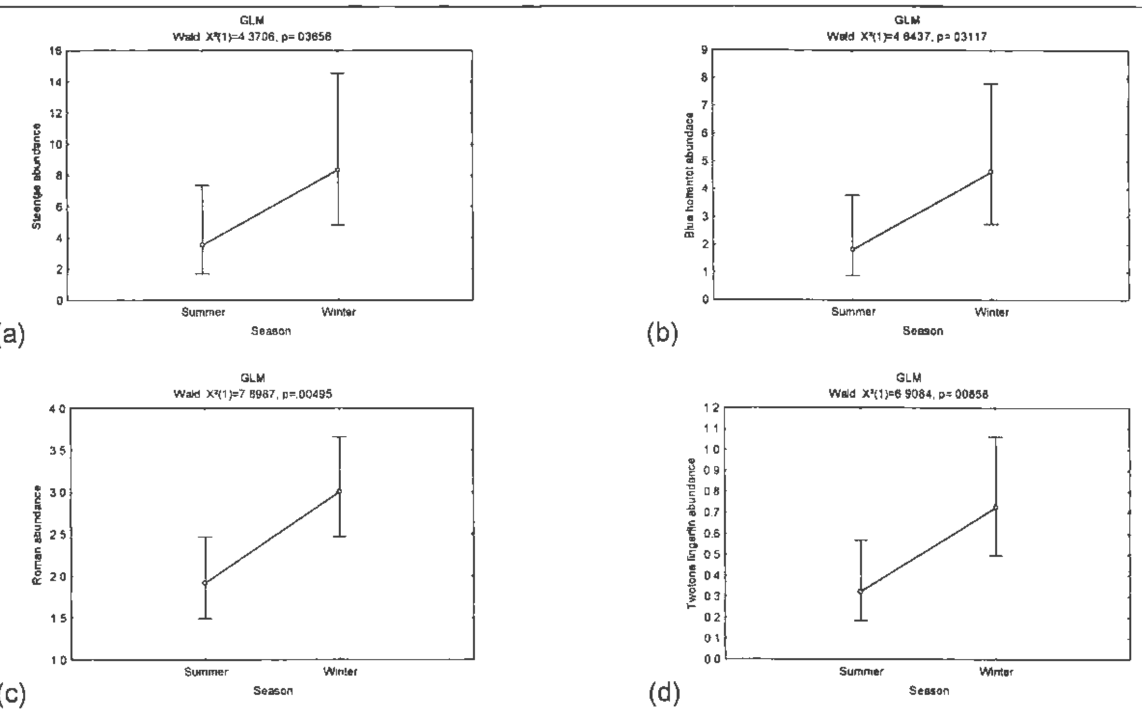


Figure 3.37. Influence of season on (a) steentjie, (b) blue hottentot, (c) roman and (d) twotone fingerfin abundance.

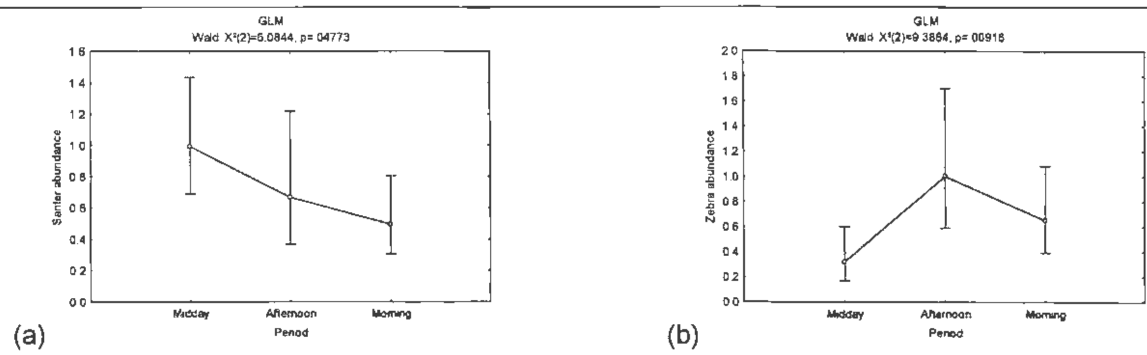


Figure 3.38. Influence of period on (a) santer and (b) zebra abundance.

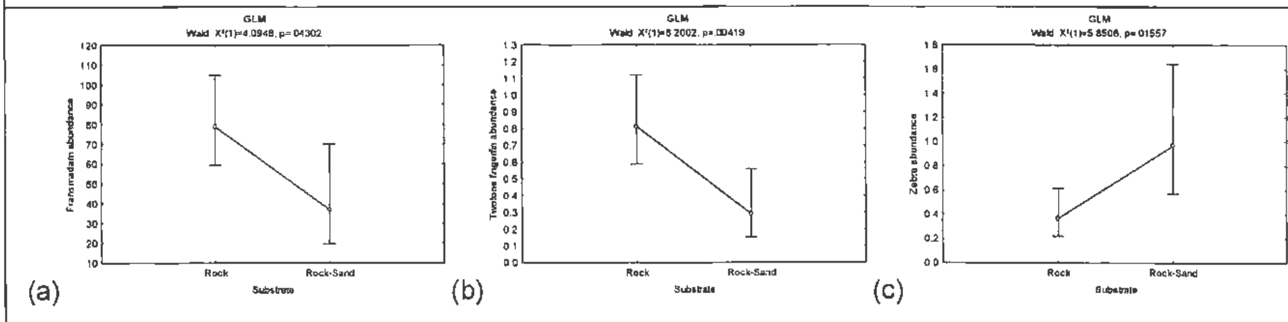


Figure 3.39. Influence of habitat type on (a) fransmadam, (b) twotone fingerfin and (c) zebra abundance.

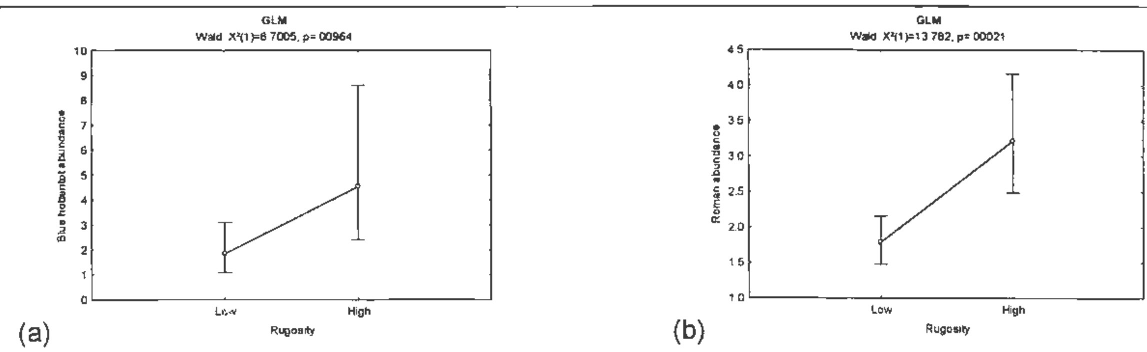


Figure 3.40. Influence of rugosity on (a) blue hottentot abundance and (b) roman abundance.

Summary of key findings – UVC

- Species composition differed between UVC and controlled angling survey methods
- The influence of factors on diversity was low (Table 3.19)
- The influence of factors on the relative abundance of individual species was variable, with depth, season and substrate being the most important factors (Table 3.20)
- Roman length was influenced by numerous factors (season, depth, reef profile and rugosity)

Table 3.19. Summary of key factors influencing diversity and community structure from UVC.

Species	Relative biomass						
	Subst.	Season	Period	Depth	Temp.	Rugosity	Profile
Species richness	×	✓	✓	×	✓	✓	✓
Pielou's Evenness	✓	×	×	×	×	✓	✓
Shannon-Wiener Diversity	×	✓	×	×	×	×	×
Taxonomic Diversity	×	×	×	×	×	×	×
Multivariate	✓	✓×	×	✓	✓	×	×
Importance score	40	50	20	20	40	40	40

Note: weak effects are indicated by both a tick and cross

Table 3.20. Summary of key factors influencing individual species abundance from UVC.

Species	Relative biomass							
	Subst.	Season	Period	Depth	Temp.	Vis.	Rugosity	Profile
Fransmadam	✓	×	×	✓	×	×	×	×
Steentjie	×	✓	×	✓	✓	×	×	×
Strepie	×	×	×	✓	×	×	×	×
Blue hottentot	×	✓	×	✓	×	×	✓	×
Roman	×	✓	×	×	×	×	✓	×
Blacktail	×	×	×	✓	×	×	×	×
Santer	×	×	✓	×	×	✓	×	×
Twotone fingerfin	✓	✓	×	×	×	×	×	✓
Zebra	✓	×	✓	✓	×	×	×	×
Importance score	33	44	22	67	11	11	22	11

3.4 Discussion

The main aim of the angling and diving survey presented in this chapter was to conduct a non-destructive baseline assessment of the reef linefish communities in order to determine spatial and temporal trends in community composition and relative abundances of dominant linefish species within Algoa Bay. This was achieved through the identification of the seven main reef areas of Algoa Bay, and through the use of non-destructive controlled angling surveys to determine community composition and relative abundance of linefish. Due to the non-destructive nature of the surveys no gut content analysis was conducted on the key fishery species. This would be a valuable contribution to future research in Algoa Bay as it would improve the understanding of the distribution of the different life history stages of key fisheries species in relation to the distribution and abundance of preferred prey items. This was, however, beyond the scope of the current study.

This chapter provides the first detailed fishery independent comparison of reef linefish communities across the full spatial range of Algoa Bay and contributes to the knowledge gained from earlier studies conducted in the western region of Algoa Bay (e.g. Buxton 1987; Beckley and Buxton 1989).

3.4.1 Spatial and temporal trends in reef linefish communities

(a) Spatial patterns

Multivariate analysis indicated a distinction in linefish communities with two groupings (Group 1 and Group 2) clearly discernable. The two inshore study areas within the western bight of Algoa Bay (Group 1: BB and StC) differed substantially from the other five reef complexes due to lower relative abundances of dominant reef associated sparidae species, including santer, fransmadam and roman, and the presence or higher relative abundances of non-reef dependent species such as elf, silver kob, red tjor-tjor, bluntnose spiny dogfish and white seacatfish. In addition, several reef dependent species from the Sparidae family were noticeably absent from these two sand-interspersed reef areas. Within the Group 2 community the two inshore study areas WC and CP grouped closely together, while the two offshore solid reef areas RB and BI were most similar to each other, suggesting an inshore-offshore similarity. These results indicate that there is a clear, habitat dependent structuring of reef fish communities in Algoa Bay.

Wave exposure and substratum characteristics are important factors influencing reef fish communities (Friedlander *et al.* 2003; Floeter *et al.* 2007; De Raedemaecker *et al.* 2010) and may play an important role in structuring reef linefish communities within Algoa Bay due to the local meteorological and oceanographic conditions. Driven by westerly winds, south-westerly swell is predominant in the region particularly over the winter months (Schumann *et al.* 2005). Study sites within the western region of Algoa Bay are more sheltered as a result of buffering by the Cape Recife headland which acts as a barrier to the dominant south-westerly swell. Contrarily study sites further offshore and to the east of Algoa Bay receive little or no protection, being exposed to heavy swell conditions. The difference in exposure could therefore influence reef linefish community structure within Algoa Bay.

Although the coastline's orientation results in some study areas being protected and others exposed, the notable absence of several reef associated species in Group 1 indicates that differing habitat characteristics play an overriding role in structuring reef linefish communities within Algoa Bay. As opposed to the other sheltered study area (Evans) the Group 1 reef complexes consisted of low profile reef, were smaller in size, and were more fragmented and interspersed by sand patches than the other reef complexes comprising Group 2. Longshore movement of large volumes of sand occurs through Algoa Bay in a north-easterly direction (Illenberger 1993; McLachlan *et al.* 1994). This sand movement and periodic storm events may lead to temporary sand inundation of smaller low profile reef complexes which would become temporarily unavailable to reef fish communities. This would reduce habitat, shelter and foraging areas for reef dependent species, increasing competition on these smaller reef complexes possibly contributing to the lower overall relative abundance of reef linefish in these areas. Sand inundation of reef substrata may also reduce benthic cover thereby affecting food availability for fish. Group 1 habitat characteristics are likely to be less suitable for highly resident reef dependent species which forage on reef macro-invertebrates or algae. Such conditions may favour generalists and opportunistic pioneers which are able to colonise changing or disturbed habitat more easily (Götz 2005), possibly further contributing to the differences in species composition.

White seacatfish, bluntnose spiny dogfish, red tjor-tjor and piggy were encountered more frequently on Group 1 reef complexes. The more sheltered nature of Group 1 reef complexes in the western region of Algoa Bay may also provide more suitable conditions for early juveniles of some species. Studies have indicated that numerous marine species utilise sheltered coastal embayments as nursery areas in several parts of the world (Smale 1984; Blaber *et al.* 1995; Conrath and Musick 2007; Bradbury *et al.* 2008) and certain areas of Algoa Bay may serve as important nursery areas for some species, possibly contributing to the differing community structure in these areas.

Due to the clear distinction of two reef linefish community types in Algoa Bay, each is discussed separately below in terms of the main species and factors influencing community structure.

(i) Group 1 communities

Depth was the only factor which had a meaningful effect on the community structure within Group 1. Substrate type, season and study area affects were very weak. Lower abundances of santer and higher abundances of white seacatfish and silver kob at deeper sites were the main contributors to the observed difference in community structure. Species abundance data indicated close association of the reef associated species, roman, santer and fransmadam to the left of the nMDS ordination. Contrarily white seacatfish and silver kob displayed similar spatial orientation on the right-hand side in the ordination. Marine fish illustrate strong substrate preferences (Macpherson 1994; Szedlmayer and Howe 1997; Sampson 2002). Santer, roman and fransmadam are more abundant over reef habitats (van der Elst 1995; Heemstra and Heemstra 2004), juvenile silver kob aggregate over sandy substrata behind the backline and in embayments (Griffiths 1997c), and white seacatfish and red tjor-tjor occur along reef fringes and over sandy substrata (van der Elst 1995; Heemstra and Heemstra 2004). These species habitat associations suggest that habitat preferences may be an overriding factor which may not have been detected due to the inaccuracies associated with distinguishing and assessing the quality of hard substrate using the colour echo display. Differentiation between gravel, reef and reef covered by a thin layer of sand is difficult to distinguish and may have influenced the interpretation of results. Future research should make use of remotely deployed cameras to evaluate habitat quality during controlled angling assessments in order to improve the interpretation of results.

Santer was numerically abundant in all study areas (Group 1 and 2 reef complexes); however, the mean length of santer in the BB area was considerably lower than in all other areas with only 19% of the sampled population above the size at 50% maturity, whereas at the other reef complexes 35%-65% were mature. The catch rate for adult santer was also considerably lower than at all other reef complexes, while the juvenile catch rate of santer was high. Santer is a heavily targeted species by the skiboat sector (Smale and Buxton 1985; Brouwer 1997; Donovan 2010)(Chapter 5) and the size related differences in catch rate in the BB reef complex indicate that fishing pressure is likely to be a significant pressure on reef fish communities in this area. The fishing pressure in this area is relatively high (Chapter 5) with many vessels stopping to fish this reef on return from other fishing locations further afield. Fishing from kayaks has recently become increasingly popular and the close proximity and sheltered nature of this reef complex make it a popular destination. Due to the small size of the reef complex it is likely to be relatively sensitive to fishing pressure as effort is concentrated over a small area. The influence of fishing pressure on reef fish communities has been illustrated over larger reef complexes along the south-east coast of South Africa based on differences in abundance and size across the border of a no-take MPA (Götz 2005; Götz *et al.* 2009b). Similarly studies in other regions have indicated a reduction in size (Friedlander and DeMartini 2002; Dulvy *et al.* 2004; Graham *et al.* 2005) and density (Friedlander and DeMartini 2002; Dulvy *et al.* 2004; Evans and Russ 2004) in heavily fished areas as well as a change in the trophic structure of reef fish communities as larger predators were replaced by herbivorous species in heavily fished areas (Friedlander and DeMartini 2002). The high abundance of juvenile fish suggests that this sheltered reef system provides good habitat for this species and even with the high levels of fishing pressure it serves as an important nursery area for reef linefish species within Algoa Bay. Indeed the greatest abundance of dageraad occurred in this area with all individuals being juveniles. Santer in the StC area were also smaller than the Group 2 reef complexes, but CPUE between juveniles and adults did not differ vastly. The smaller mean size in this area could also be due to sustained high levels of recreational (Chapter 5) and commercial (Chapter 6) fishing effort in this region of Algoa Bay leading to lower abundances and the removal of larger individuals as documented in other studies.

This study has shown the presence and high relative abundance of silver kob, an important fishery species, in the StC study area. Over one third of the silver kob caught in the StC area were below the size at 50% maturity indicating the potential importance of this area as a nursery ground for this species in Algoa Bay. Surveys conducted in 1980 similarly indicated high abundances silver kob in the shallow inshore regions of Algoa Bay, particularly near Jahleel Island, supporting the findings of this study (Smale 1984). Juvenile silver kob are known to recruit to shallow nursery areas immediately behind the surf zone, being particularly abundant over shallow soft substrates (Smale 1984; Griffiths 1997c). Previous studies have indicated that silver kob move offshore during winter months concentrating in the nearshore in summer (Griffiths 1997c). No seasonal differences in relative abundance of juvenile or adult silver kob were apparent within Algoa Bay in this study suggesting year round utilisation of the sheltered nearshore habitats in Algoa Bay. Few silver kob were captured in the BB study area. However, sampling effort was limited to fewer sites over a one-year period. The large temporal variability evident from the StC area coupled with the low sampling intensity in the BB area may lead to underestimation of the importance of this region for silver kob in this study and a more intensive assessment is required to verify the importance of this area to silver kob.

White seacatfish were the fourth most dominant species in the Group 1 communities and are an important fishery species for the subsistence sector, comprising between 20-23% of the catch in Eastern Cape estuaries, with a retention rate of approximately 88% (Potts *et al.* 2011). Although not heavily targeted by the skiboat fishery they are often retained when captured and are susceptible to fishing pressure due to their k-selected life history traits including longevity, late maturity, egg brooding and low fecundity (Hecht and Tilney 1989). White seacatfish showed considerable overlap with silver kob in the MDS ordination (Figure 3.9) suggesting similar habitat preferences, and are often caught in association with each other in the skiboat fishery (Hecht and Tilney 1989). Although not considered overexploited, higher catch rates in Algoa Bay than any other region or embayment along the south east coast (Smale 1984) highlight the importance of Algoa Bay for the conservation of this species. Large numbers of juveniles have been reported in Algoa Bay (Smale 1984) suggesting an important nursery area. Conversely, only 3.4% of the white seacatfish landed in the current study were below the length at 50% maturity, but this is likely due to gear selectivity towards larger fish. The recent increase in the allocation of subsistence permits in the Eastern Cape (A.Oosthuizen *pers. comm.*) is likely to place additional fishing pressure on estuaries in the region, potentially increasing the harvest of this species for which there are currently no fishery restrictions. Spatial closures would enhance formal protection for this species.

Two additional important fishery species were captured in the BB and StC areas. Elf comprised over 4% of the catch in both the BB and StC reef complexes. They are heavily targeted by the recreational sector and susceptible to overexploitation due to their predictable aggregation and migration patterns. Although adults may occur offshore, juveniles are more common in coastal embayments (Smale 1984; Mann 2000) and susceptible to exploitation in these areas, particularly as they are known to aggregate off points and structures. Few dageraad were caught in the BB area, but they represented the highest proportion of the total catch from any of the reef areas sampled. All fish were well below the size at 50% maturity suggesting either heavy exploitation by the fishery, or good recruitment of juveniles into the BB area.

The primary focus of this study was not to assess the nursery function of reefs, however, the methods were suitable for sampling sub-adults available to the fishery. These results have indicated the potential importance of the BB and StC areas as nursery grounds for santer, silver kob and dageraad. These findings suggest that further research on the distribution and recruitment of juvenile fish into Algoa Bay is required and would be valuable contribution to future management of the marine ichthyofauna. A research programme is currently investigating this further.

(ii) Group 2 communities

Although diversity indices (with the exception of the WC reef complex) were similar between the Group 2 and Group 1 reef complexes, the number of reef associated sparids was higher in the Group 2 reef areas and fewer soft sediment or bay associated species (e.g. piggy, silver kob, white seacatfish, elf) were encountered. Spatial differences in community structure and diversity between reef complexes in the Group 2 communities was the most influential factor, although the magnitude of the effect was relatively small. Temporal aspects were not significant and depth related differences in community composition were very weak.

Hierarchical clustering indicted a general separation within the Group 2 communities into inshore (Ev, WC and CP) and offshore (RB and BI) reef linefish communities. Although only small, the observed differences are likely to be attributable to habitat characteristics at each reef area sampled. RB was the only area in which diversity differed significantly from all other sites and differences in community assemblages were strongest. These trends correspond with the greatest habitat complexity and the close proximity of deep water (>30m) to the RB reef complex. The RB area is situated approximately 22km from shore and is surrounded by deep water (>80m) creating a high profile reef with numerous caves and ledges resulting in a high habitat complexity over a small spatial scale. Both habitat complexity and depth are important factors influencing reef ichthyofaunal communities, abundances and size structure (Bell 1983; Buxton and Smale 1989; Friedlander and Parrish 1998; Angel and Ojeda 2001; Friedlander *et al.* 2003; Götz 2005; Bennett 2007; Benfield *et al.* 2008; Howard *et al.* 2009; Götz *et al.* 2009b) and these factors are likely to have contributed to the higher diversity in the RB area, and the greater differences in community structure to other areas. Similarly, the BI area is situated some distance (10km) offshore and is characterised by large pinnacles and high profile reefs on the seaward side with the bathymetry dropping rapidly to depths >50m in close proximity. This is likely to account for the greater similarity between the RB and BI sites. Contrarily to the RB and BI areas no deep water reefs are present in close proximity to the CP and WC as they are situated close to the shoreline, possibly contributing to the greater similarity between these sites.

The WC reef complex differed most from all other Group 2 reef complexes due to lower diversity and the dominance of one species, santer. This was attributable to the high abundance of juvenile santer as the relative abundance of adult santer was similar across all Group 2 reef complexes, while juvenile CPUE in the WC area was three to six times higher than at the other Group 2 reef complexes. Approximately 65% of the santer landed were below size at 50% maturity. Skiboat fishing effort in this region of Algoa Bay is low (Chapter 5; Chapter 6) due to the long travel distances from vessel launch sites and it is unlikely that fishing pressure contributed to the skewed ratio of adults to juveniles in the WC area. Therefore, it is likely that WC serves as a recruitment area within Algoa Bay.

The differences in juvenile abundance between reef areas is not easily interpreted as little knowledge of the spawning behaviour, spawning areas and nursery habitats is available for this and many other sparids. Santer are known to spawn during summer along the Eastern Cape coastline (Coetzee 1978; Coetzee 1983) and juveniles frequent shallow protected coastal waters (van der Elst 1990). A marked difference between WC and other reef complexes is the predominance of flat reef with few high profile ridges. Santer are reported to show preference for flat low profile reef (Coetzee and Baird 1981b) although the reasons for this are unclear. Unfortunately, macro-benthic communities, which may provide important food sources for juvenile santer, could not be assessed in the WC area as repeated dive surveys were aborted due to poor diving conditions and low water visibility (<1m). Further studies are required in order to determine the primary factors contributing to the importance of this area to juvenile santer.

The controlled angling survey indicated an inverse correlation in abundance of santer and other dominant reef species on the Group 2 reef complexes. From the MDS ordination (Figure 3.25) it is apparent that in areas where santer were most abundant few other large sparids such as roman, scotsman and red stumpnose were present. The small sparid, fransmadam, and to some extent steentjie, also indicated a separation from santer dominated communities, while showing greater overlap with the less abundant larger sparids roman, scotsman and red stumpnose. The dominance of santer may have resulted from past fishing activities in Algoa Bay which removed other large sparids. Historical records indicate large red steenbras were captured frequently in Algoa Bay (Biden 1954), yet very few were captured in the current assessment. This highlights the importance of comparable baseline data for future monitoring and the potential influence of 'shifting baselines' in long-term temporal assessments of marine communities.

Alternatively these patterns may be related to species behaviour. Santer occur in dense shoals, are highly mobile and are aggressive feeders, and may be more susceptible to being caught than other more sedentary species. There is a paucity of behavioural information for most sparids which hinders the interpretation of such interactions. Few detailed movement and behavioural studies have been conducted on offshore sparids, with roman being the notable exception (Kerwath 2005; Kerwath *et al.* 2007a; Kerwath *et al.* 2007b). Conventional tagging studies have provided some insight into the long-term and broad scale movement patterns of some important sparids (Griffiths and Wilke 2002; Brouwer *et al.* 2003; Watt-Pringle 2009). There is little detailed knowledge of movement, behaviour or interactions of linefish species within reef complexes and further research into these aspects is required.

(b) Temporal trends

Community structure from controlled angling surveys within each study area differed significantly between years and seasons, but the magnitude of the effect was small (Global $R < 0.1$) indicating little difference in community assemblages. The relative abundance of certain species differed inter-annually, seasonally and within a sampling day indicating high natural variability in local populations. This was particularly noticeable for the shoaling species, silver kob and white seacatfish. Varying scales of temporal variability in communities and fish abundance have been investigated in previous studies (Thompson and Mapstone 2002; Willis *et al.* 2006; Masuda 2008), with a large proportion of variability attributed to short-term fluctuations.

No seasonal trends in santer abundance were observed, while roman abundance increased in winter. These findings support those of Götz *et al.* (2008) and Lechanteur (2002) for roman in the Goukamma and Castle Rock MPAs respectively, but contradict those of Buxton and Smale (1989) in the Tsitsikamma National Park and Cape Recife areas. Lower abundances in summer may be attributed to cold upwelling resulting in fish seeking shelter deep within the reef and becoming more inactive (Lechanteur and Griffiths 2002; Kerwath *et al.* 2007b). The peak spawning period for many temperate sparids in South Africa is reported to occur during summer months (Coetzee 1983; Buxton 1987; Buxton 1990; Brouwer *et al.* 2003; Brouwer and Griffiths 2005b) and reduced CPUE may be expected due to a reduction in feeding activity over this period. The current study detected lower roman abundance during both UVC and controlled angling surveys during summer. Differences in abundance cannot therefore be attributed directly to feeding behaviour alone. Reduced abundances over summer coincided with lower mean sizes of roman observed during UVC. Past studies have indicated that roman are highly resident (Kerwath *et al.* 2007a; Kerwath *et al.* 2007b) and movement of large adults from the study area during the spawning season is therefore unlikely. The observed differences in abundance of roman may be linked to the increased occurrence of cold water upwelling during the summer months (Schumann *et al.* 2005) leading to extended periods of inactivity and hiding (Kerwath *et al.* 2007b). The decrease in size during summer observed during UVC may be due to greater activity of females over this period as they have been shown to increase their home range size over the spawning period (Kerwath 2005) and are typically smaller as roman are protogynous hermaphrodites.

Results indicate changes in santer abundance and mean length during different sample periods of the day. Short-term temporal effects often account for the greatest variability although the cause is difficult to establish. Crepuscular peaks in activity linked to foraging and feeding behaviour have been reported for several marine species (Galzin 1987; Lowry and Suthers 1998; Thompson and Mapstone 2002; Dawson and Starr 2009) which may affect their short-term abundance and distributions within reef complexes. Avoiding dawn and dusk periods during sampling is therefore important in order to minimise bias associated with crepuscular movement. Stratification over different time periods of the day is important to minimise bias introduced by species specific behaviour (Willis *et al.* 2006).

This study has indicated high temporal variability in abundance estimates and the need for adequate replication and stratification of sampling effort to ensure statistical robustness is not compromised.

3.4.2 Other factors influencing reef linefish communities

An important aspect of this study was to evaluate the influence of key explanatory factors on the distribution and relative abundance of reef linefish communities in order to identify those which should be taken into consideration in the planning, design and analysis of future monitoring programmes in Algoa Bay. Several factors influence the distribution, abundance and size of species in the natural environment contributing to variability in the data. The relative abundance of linefish, as determined from controlled angling surveys, may be influenced by several factors which influence their availability to the sampling method. As the sampling method is dependent on fish feeding, factors which alter their feeding behaviour would influence their likelihood of capture and availability to the sampling method which is ultimately reflected as a change in relative abundance. Improved understanding of these factors is particularly important in long-term monitoring programmes as the design must take them into consideration. In order to improve the interpretation of long-term monitoring data key parameters which contribute to the natural variability need to be incorporated into the study design to minimise variability, making real trends become more apparent. Each factor recorded during the baseline survey is discussed briefly below in relation to the current results and findings of past studies.

(a) Depth

Results of both the controlled angling and UVC surveys indicated that reef fish community structure was influenced by depth, but the differences in fish assemblages between depth categories was small. Depth associated differences in the relative abundance and mean size of several species was recorded. No depth related differences were observed for either santer or roman abundance during UVC, while controlled angling surveys indicated decreasing abundance of adult, and increasing abundance of juvenile santer with depth and no differences for roman. Santer and roman length both decreased with increasing depth.

Studies conducted in tropical waters of the red sea (Brokovich *et al.* 2006), Hawaii (Friedlander and Parrish 1998) and the Caribbean (Williams *et al.* 2010) as well as temperate waters of Chile (Pérez-Matus *et al.* 2007), California (Martin and Lowe 2010), Australia (Bell 1983) and New Zealand (Brook 2002) have revealed similar depth related changes in ichthyofaunal communities. The change in fish community structure with depth is most likely due to changes in benthic composition and feeding biology of the reef associated fish species occurring in these areas. Depth and exposure related changes in algal and invertebrate communities have been reported by several authors (Branch and Branch 1983; Buxton 1987; Anderson and Stegenga 1989; Burger 1990; Garrabou *et al.* 2002; Götz *et al.* 2009c; Smale *et al.* 2010) which is likely to have influenced the distribution of fish feeding on them.

Depth related changes in the abundance of several species have been reported by several authors along the south coast of South Africa (Buxton and Smale 1989; Burger 1990; Lechanteur and Griffiths 2002; Götz 2005; Smith 2005b; Bennett 2007; Götz *et al.* 2009b). Some authors have reported higher fish abundance on shallow reefs (Lechanteur and Griffiths 2002; Götz 2005) while others have reported the opposite (Buxton and Smale 1989; Burger 1990). No prior studies have reported findings for the influence of depth on santer abundance and contradictory results exist for roman. Götz (2005) found higher abundances in shallow water (although limited to a minimum depth of 8m) while Buxton and Smale (1989) (11-25m) and Smith (2005b) (16-20m) reported greater abundances at deeper sites, and Bennett (2007) found roman abundance was greater at deeper sites during UVC but lower during angling surveys. This indicates high natural spatial and temporal variability in fish distribution which is likely to influence the ability to detect long-term changes in fish abundance if sampling is not appropriately replicated.

Similar to this study, roman length decreased with depth along the south coast (Götz 2005; Götz *et al.* 2009b). Contrarily Buxton (1984) reported a spatial separation of adult and juvenile roman on reefs in Algoa Bay (BB area in this study), with juveniles occurring in the shallow sub-tidal and adults showing a preference for deeper reefs (Buxton and Smale 1984; Buxton and Smale 1989). Few juvenile roman were caught during the controlled angling survey in this study, limiting the ability to distinguish preferences by age and size. The UVC survey at Bird Island, however, revealed that only 17% of the observed roman were juveniles and that larger adult fish comprised a larger proportion of the roman population in the shallow waters of Bird Island. Habitat characteristics and the associated algal and grazing faunal communities have been suggested to influence the distribution and size structure of roman (Götz 2005). The reef complex at BI, where UVC was conducted, is unique in having vast areas of shallow sub-tidal reef (<10m) with high abundances of algal beds, which receive large input of nutrients from the guano deposits on the adjacent islands, conditions which are uncommon along most of the south-east coast. These highly productive conditions in the shallow sub-tidal area may have influenced the distribution and size spectra of roman due to higher abundances of prey items. Similarly controlled angling indicated that santer were larger, and adults were more abundant in shallower water. This may also be related to ontogenetic changes in dietary preferences and feeding strategies with prey items being more abundant in the productive shallow waters around Bird Island.

It is apparent from this and other studies that depth influences the community structure and relative abundance of several species. Consequently, stratification over depth and incorporation of depth data into statistical analyses is an important aspect for future evaluation of reef fish communities.

(b) Substrate

Community structure differed between rock and rock-sand habitats but many species contributed to the observed differences and the magnitude of the effect was small suggesting little difference in community assemblages between the two habitat types. However, an influence of habitat type on reef fish communities has been reported in several other studies (Friedlander and Parrish 1998; Ault and Johnson 1998; Friedlander *et al.* 2003; Brokovich *et al.* 2006; Williams *et al.* 2010; La Mesa *et al.* 2010) indicating that it is an important factor determining the distribution and abundances of species. Distances between substrate categories (rock and rock/sand) during UVC surveys were short and may not have been sufficient to alter community structure significantly. Similarly difficulties in recording substrate type during controlled angling surveys may have confounded the results. Santer abundance was higher over rock than rock-sand habitats in the controlled angling survey in the BB and StC areas and it is well documented that resident reef fish are generally more abundant over hard substrata than areas of mixed rock-sand (Bennett 2007).

Habitat mapping of study areas using sidescan and multi-beam sonar will allow for the improved planning and interpretation of controlled angling results where *in situ* assessment of habitat type is not possible.

(c) Habitat complexity

Habitat complexity is a function of the vertical relief (profile) and the number of holes and crevices (rugosity) available as refuges to reef species. Community structure and diversity are closely associated with habitat complexity. More complex habitats generally support a greater diversity of species as a result of increased niche availability. Contrary to the findings of many studies (Roberts and Ormond 1987; Angel and Ojeda 2001; Friedlander *et al.* 2003; Almany 2004; Brokovich *et al.* 2006; Hunter and Sayer 2009) the results from this investigation indicated that neither profile nor rugosity influenced community structure significantly, similar to the findings of Götz (2005). Reef profile and rugosity did influence diversity measures significantly, with greater species numbers occurring over high relief and high rugosity reef areas, while evenness was greater over low relief and low profile reefs indicating the dominance of certain species in complex habitats.

Neither rugosity nor profile influenced santer abundance, but roman abundance and length was influenced significantly with higher densities and larger individuals in areas of greater rugosity, and larger individuals on high profile reefs. This suggests that both factors may be important for different life stanzas of reef species, which may not be detected through abundance estimates alone. Size differences in roman suggest dominance of larger adult males in areas of higher habitat quality. Reef profile has been shown to be an important factor for predicting the abundance of several species along the south coast of South Africa (Buxton and Smale 1989; Götz 2005; Smith 2005b; Götz *et al.* 2009b). Götz (2005) reported roman occurring in higher abundances over low rugosity areas, while findings by Smith (2005b) support the results of the current study and Bennett (2007) found that rugosity was not a significant factor in predicting roman abundance. This highlights the complexity of habitat effects on reef fish abundance and further research is required to investigate the specific habitat effects in more detail.

Bennett (2007) suggests that there is a strong correlation between rugosity and profile and that only one factor should be included in future monitoring and analyses. Rugosity is a measure of the reef complexity, providing an indication of the availability of suitable refuges for fish, while profile refers to the vertical relief of the reef and provides an indication of the presence and extent of pinnacles which may act as natural fish aggregating devices for shoaling species (Itano and Holland 2007). They were both considered important factors in the current study as they may influence the abundance of species differently. The lack of significance for these factors on abundance for many species was surprising. Reef profile should definitely be taken into account in the design and planning of monitoring studies. Stratification of sampling effort can be conducted over different reef profiles based on prior bathymetric mapping conducted within the study area. Improved technology such as sidescan and multi-beam surveys should be used in dedicated monitoring sites in order to obtain detailed information which will allow for improved design and stratification of the monitoring programme and interpretation of results.

(d) Visibility

Visibility (visual estimate of turbidity) may influence the abundance of fish counted during UVC surveys by either modifying a fish's behaviour or alternatively changing the efficiency with which a diver is able to detect and count fish during the survey. Modification of fish behaviour as a result of changes in visibility is likely to be species specific. Increased visibility may increase the susceptibility of reef associated fish species to predation by larger visual predators, resulting in them seeking refuge in caves and crevices, making them harder to detect during UVC counts. Alternatively turbid waters may lead to increased susceptibility to predation by non-visual predators such as sharks, which are highly adapted to low visibility conditions, causing fish to seek shelter under these conditions (Götz 2005). Some reef fish species have been reported to move away from divers (Lechanteur and Griffiths 2002), while others are attracted (Smith 2005b). Visibility may therefore alter a fishes behaviour based on the presence of divers, and although survey design cannot be structured around visibility it is an important factor which should be recorded during UVC surveys. Divers should also be well trained in order to adjust to different visibility conditions ensuring the census methods are adapted appropriately.

(e) Temperature

Bottom water temperature did not influence the overall CPUE, but santer abundance decreased with temperature indicating either lower feeding activity or movement away from the cold water during such events. Smith (2005b) reported a general decrease in abundance of all species below temperatures of 14°C. In the current study only a few dives (6%) (mean 17.8 ± 2.4°C) and few controlled angling sites (4%) (mean 17.2 ± 1.9°C) were conducted below this temperature, possibly accounting for the lack of temperature related differences detected during this study.

Temperature has a significant influence on reef fish abundance and distribution with rapid drops during upwelling responsible for fish mortality (Hanekom *et al.* 1989), and lower abundances of several species along the south coast of South Africa (Buxton and Smale 1989; Lechanteur and Griffiths 2002). Detailed telemetry studies have indicated that under such conditions roman seek refuge in caves and crevices deep within the reef with little movement until temperature increases (Kerwath *et al.* 2007b). Increased predator feeding activity has been linked to periods of inactivity of fish as a result of a drop in temperature (Kotrschal 1983). Fish may therefore be more susceptible to predation during cold water events and undertake a behavioural response by seeking refuge in caves and crevices to reduce their vulnerability. Such behaviour may account for the reduced abundance of reef fish associated with low water temperature reported along the south coast of South Africa by several authors (Buxton and Smale 1989; Götz 2005; Kerwath 2005; Bennett 2007).

The incongruity of the results between this and similar studies within the same biogeographic region, and conflicting results detected by different survey methods, highlight the complex interactions which influence reef fish communities. This results in high variability in abundance estimates and future studies must ensure temporal and spatial stratification across strata with adequate replication. Temperature should always be recorded, and detailed habitat information should be recorded during UVC surveys and obtained using remotely deployed cameras during controlled angling surveys.

3.4.3 Evaluation of survey methods

Controlled angling surveys are cost effective and simple to implement as they do not require specialised equipment or highly skilled personnel. They therefore allow for large sample sizes to be obtained over broad geographical areas in a relatively short space of time. In contrast, UVC surveys require appropriately trained divers (legislative requirement), are limited by the constraints of SCUBA (depth; dive time; number or repetitive dives), are more dependent on environmental conditions (visibility; currents; surge) and are more costly due to the specialised equipment required. Although most of the equipment required for UVC is a once-off capital expenditure, ensuring sufficient scientific divers are available during weather windows can be difficult, so trained commercial divers often need to be contracted to assist in field work being an additional expense to the operational costs of the monitoring programme. Monitoring programmes need to be easy and cost-effective to implement in order to be sustainable in the long-term. This often necessitates using less complicated sampling protocols which can be conducted by volunteers who are appropriately supervised by a trained scientist. Due to the range of reef areas to be sampled, the travel distances involved in accessing these sites, the limitations of SCUBA (legal requirements; depth; bottom time; number of dives per day), the poor diving conditions (visibility and surge) and the large sample sizes required, controlled angling was selected as the preferred sampling method for the broad scale baseline assessment of reef linefish communities across Algoa Bay. A major disadvantage is the inability to assess substratum type and habitat quality *in situ* at each sampling site (Bennett *et al.* 2009) in order to assess the potential influence of these factors on the catch rate. The assessment of habitat type is a clear limitation of the sampling method and in order to overcome this in future monitoring and improve interpretation of results, remotely operated or jump cameras should be used to obtain habitat information. Alternatively sidescan sonar and multibeam surveys could be used to produce accurate high resolution maps for areas in which long-term monitoring is to be conducted.

Based on the frequency of occurrence of species in the BI study area it was apparent that the results differed between the two methods (UVC and controlled angling). Although more angling than UVC sites were visited, both survey methods had the same spatial coverage and should therefore be representative of the ichthyofaunal communities in this area. The frequency of occurrence and relative percent difference indicates the bias of the controlled angling survey method towards larger and potentially more aggressive species. Santer, seventy-four and dageraad were captured more frequently than they were observed during UVC (Figure 3.41). Contrarily other large sparids such as roman, englishman, red steenbras, black musselcracker and red stumpnose, were observed during UVC more frequently than they were captured during controlled angling. These differences are likely to be due to behavioural responses of individual species to the different survey methods. Stimulation by bait is likely to induce a stronger feeding reaction in aggressive species, while the presence of divers during UVC deters others. Similarly the smaller species, fransmadam, steentjie, blacktail and blue hottentot were observed during UVC more frequently than they were captured during controlled angling. Although this may also largely be a result of behavioural interactions, the influence of hook size selectivity may also be significant. Nonetheless these observations reinforce the need for complementary survey methods when undertaking baseline surveys and highlight the importance of undertaking comparable studies for long-term monitoring.

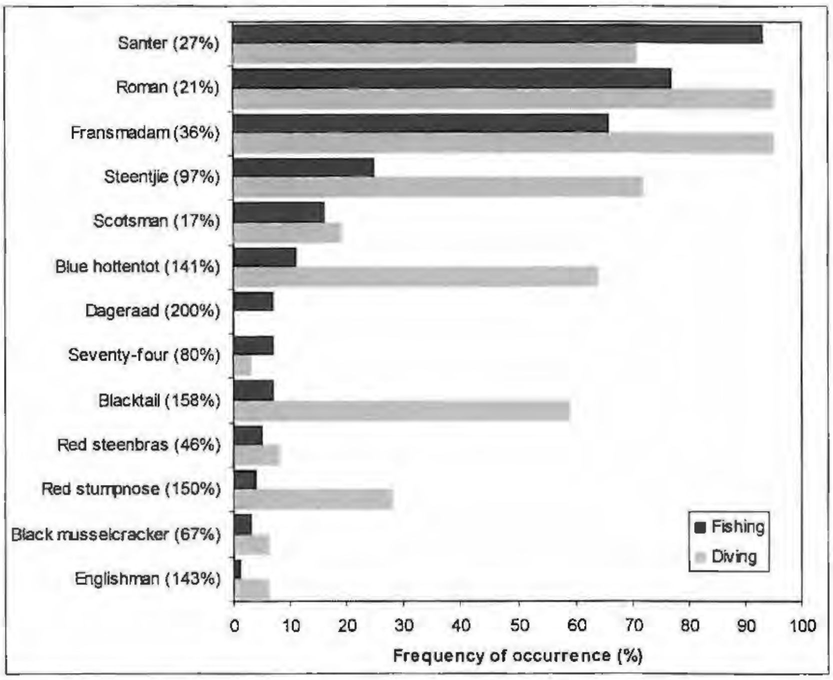


Figure 3.41. Frequency of observing or capturing species at controlled angling and UVC sites. Number in brackets indicates the relative percent difference.

3.4.4 Conclusions

Numerous, previously uncharted reef areas were identified in Algoa Bay and seven representative sites were selected for further investigation of reef fish communities. This revealed two distinguishable spatially separated reef fish community types and led to the identification of an aggregation area for juvenile kob which serves a nursery function. These findings highlight important features which must be taken into consideration during spatial planning for conservation. Spatial data layers were developed for each feature in order to incorporate them in future planning analyses (Table 3.21).

Numerous factors contributed to the high levels of variability in reef fish community structure, and the abundance and mean size of dominant species in the communities. Furthermore, contrasting results between different survey techniques has highlighted the complexities of designing an effective and statistically robust monitoring programme for evaluating potential long-term responses of reef fish communities to future spatial management initiatives. The data presented in this chapter provides valuable baseline information for the selection of sampling sites, stratification of effort and determination of sample sizes requirements in order to design effective monitoring programmes capable of evaluating future trends (Table 3.21).

Table 3.21. Contribution of chapter results to spatial planning and monitoring in Algoa Bay.

Chapter 7: Systematic conservation planning	Chapter 8: Monitoring and evaluation
1. Spatial layer of potential and confirmed reef 2. Spatial layer distinguishing two reef fish community types 3. Spatial layer defining kob nursery area	1. Identification of key sites for future monitoring of reef linefish communities 2. Identification of key factors influencing spatial and temporal variability in reef fish diversity, community structure, relative abundance and size required for stratification of future monitoring effort 3. Estimates of variability in relative abundance and mean size of selected fish species to determine sample size requirements for future monitoring

CHAPTER 4

ASSESSMENT OF DEMERSAL ICHTHYOFAUNAL COMMUNITIES ON THE TRAWLABLE GROUNDS IN ALGOA BAY

4.1 Introduction

Coastal embayments serve as important nursery, aggregation and foraging areas for many marine species (Lenanton 1982; Morton *et al.* 1993; Blaber *et al.* 1995; Fodrie 2006; Conrath and Musick 2007; Bradbury *et al.* 2008) yet they are subject to high levels of anthropogenic disturbances and have historically received little protection in South Africa. Recent research effort has largely focused on reef associated species and the role of MPAs in the protection of these species (Lechanteur and Griffiths 2002; Lechanteur 2004; Götz 2005; Smith 2005b; Mann *et al.* 2006; Bennett 2007), with less emphasis placed on the assessment of soft-sediment demersal communities. Hence little is known of the local community structure and distribution of species within these habitats and their potential importance for management and conservation. Identification of suitable areas for protection requires a sound understanding of the distribution of key habitats, aggregation areas, differences in community assemblages and the key factors which differentiate communities. Information on the habitat requirements for different life-history stages, and the ontogenetic changes in habitat preferences is also required for planning. Recent research efforts have focused on assessing long-term temporal trends in demersal communities along the west coast of southern Africa and the impacts of fishing activities on these communities (Atkinson *et al.* 2011a; Atkinson *et al.* 2011b) as well as spatio-temporal trends along the south coast (Yemane *et al.* 2008; Yemane *et al.* 2010). However, much of this information pertains to larger geographic areas and is of insufficient spatial resolution for local level planning initiatives. Assessing baseline information on the demersal ichthyofaunal assemblages over soft sediments and understanding their habitat preferences is therefore important for future spatial planning on a local level. Abiotic, environmental and anthropogenic factors structuring community assemblages need to be identified and understood in order to facilitate effective monitoring and interpretation of temporal community change through the establishment of causal links with environmental pressures.

Degradation of benthic habitats as a result of fisheries exploitation is one of the major anthropogenic pressures which impacts on marine ecosystems (Yemane *et al.* 2010). The soft-sediment and low-relief reef areas along the south-east coast of South Africa are important fishing grounds for the demersal trawl sector as they support diverse and abundant ichthyofaunal communities. Although the Cape hakes (shallow-water hake *Merluccius capensis* and deep-water hake *M. paradoxus*) and east coast sole (*Austroglossus pectoralis*) are the main target species (Payne 1989; Payne and Badenhorst 1989), the demersal trawl sector is a multi-species fishery due to the unselective nature of the gear. Research trawl surveys on the Agulhas Bank have recorded over 160 teleost and chondrichthyan species (Smale *et al.* 1993; Japp *et al.* 1994), while over 80 species have been reported in the landings of the commercial trawl sector (Booth and Hecht 1998; Walmsley *et al.* 2007a). Due to the unselective nature of the trawl gear many species are landed by the fishery and contribute to the economic value of the sector; however, eight species are consistently reported in large quantities in the commercial catches within the Eastern Cape and account for more than 98% of the landed catch (Booth and Hecht 1998).

Bycatch of non-target species over the trawl grounds is a major problem as species of low economic value, and damaged and undersized fish are discarded at sea (Walmsley *et al.* 2007a). Bycatch has historically not been recorded and little was known of the composition or biomass of the discards and the potential impact of trawling on non-targeted species. A recent study along the south coast, however, indicated that 5% of the inshore hake-directed catch, and 19% of the sole-directed catch was discarded at sea (Walmsley *et al.* 2007a). The total discarded bycatch along the south coast was estimated to be in the region of 8 000t per annum. Although dominated by small hake, several species currently regarded as overexploited were also present (Walmsley *et al.* 2007a). The unselective nature of the trawl gear and limited knowledge of the community assemblages and distributions over the soft sediments has hindered effective management, which may reduce the landings of bycatch species. This has led to concerns regarding the sustainability of both target and non-target species. Possible means to reduce bycatch include an increase in the minimum mesh size or spatial or temporal closures of known spawning or nursery areas (Walmsley *et al.* 2007b).

Implementation of seasonal and/or spatial closures in areas where juveniles are abundant could reduce the pressure on sensitive life history stages. Furthermore spatial closures such as MPAs over trawlable grounds will afford protection to non-targeted species occurring in these habitats thereby contributing to an ecosystem approach to fisheries (EAF). In order to identify key areas for protection information on the composition and size distribution of species over the trawlable grounds is required. The majority of MPAs within the Agulhas Bioregion incorporate large areas of reef complexes with little soft or trawlable habitat protected, and the need to increase the representivity of these habitats in the existing South African MPA network has recently been highlighted (Clark and Lombard 2007). Although the demersal trawl sector is subject to additional spatial restriction with no trawling permitted within some portions of coastal embayments on the south coast, these restrictions are not well enforced and do not apply to the recreational and commercial linefisheries, chokka-squid jig fishery and the demersal long-line fishery, limiting their protective value for species targeted by these sectors. Establishment of permanent closed areas over the trawlable grounds will also contribute to long-term monitoring of changes in community assemblages in response to global climate change and allow comparisons with areas open to exploitation.

This chapter aims to provide a baseline assessment of spatial and temporal trends in demersal ichthyofaunal communities using research trawl data to identify key factors influencing their distribution on the trawlable habitats in Algoa Bay. This baseline information is required for and will contribute to marine spatial planning in Algoa Bay (Chapter 7) and the development of monitoring protocols for long-term evaluation (Chapter 8). The main objectives of this chapter were:

1. to determine key factors influencing demersal ichthyofaunal community structure and diversity in Algoa Bay; and
2. to assess the spatial and temporal trends, and factors influencing the distribution of dominant species in order to contribute to future spatial planning and monitoring in Algoa Bay.

4.2 Methods

In order to assess demersal fish communities on the trawlable grounds in Algoa Bay research trawl data were obtained from the Fisheries branch of the Department of Agriculture, Forestry and Fisheries (formally Marine and Coastal Management branch of the Department of Environmental Affairs and Tourism). Research cruises are conducted along the east coast of South Africa aboard the F.R.S. *Africana* with data available for the period 1986 to 2008. Data used in this analysis was limited to the biannual cruises conducted by the F.R.S. *Africana* during spring (September/October) and autumn (April/May). Biological data recorded during trawl surveys included species composition (by weight and numbers) and length frequency.

Each seasonal demersal trawl survey along the south coast aims to achieve a target of 100 trawl stations which are pseudo-randomly stratified across four depth strata (0-50m; 51-100m; 101-200m and 201-500m) with the number of trawl stations per stratum proportional to the area (Badenhorst and Smale 1991; Yemane *et al.* 2008). Trawl stations are selected using a 5'x5' grid (Sampson 2002; Yemane *et al.* 2008) and trawling is limited to flat areas with known high profile reefs avoided. Trawling was conducted during daylight hours for a period of 30 minutes where possible, but was shortened in instances when rough ground was encountered. In all cases data were standardised to account for different trawl durations. Towing speed (3.5 knots) and the mouth width of the net (26m) are assumed to be constant (Yemane *et al.* 2010), with the exception of recent changes in gear configuration (discussed below).

Prior to September 2003 the trawl gear consisted of a two panel 55m German otter trawl with a mouth opening of 26m and vertical height of between two and three meters, a 75mm mesh cod end fitted with a 35mm mesh liner, a rope-wrapped chain footrope and 1.5t WV otter boards (Barange *et al.* 1998; Yemane *et al.* 2008) (Hereafter referred to as "old"). Prior to the September 2003 survey the trawl configuration was changed (hereafter referred to as "new") and was used inter-changeably with the old gear during subsequent cruises (Table 4.1). The new trawl configuration consisted of a four panel 55m German otter trawl with a mouth opening of 20 to 29m and a three to four meter vertical height, and the footrope was constructed from rubber discs (Atkinson *et al.* 2011b). The new trawl gear resulted in increased sampling of the water column, reduced herding of fish due to narrower door spread, and reduced sampling of flatfish and batoids due to the absence of the chain footrope (Atkinson *et al.* 2011b).

All fish landed were identified to species level, with catches greater than five kg weighed to the nearest half kg, while small species catches were weighed to the nearest gram. For commercially important species length-frequencies were measured to the nearest centimeter below.

Table 4.1. Seasonal demersal trawl surveys conducted by the F.R.S *Africana* along the south coast of South Africa indicating the number of trawls within Algoa Bay per cruise and the trawl configuration used during the survey.

Year	Autumn (Apr/May)		Spring (Sept)	
	No. of stations	Trawl gear (old/new)	No. of stations	Trawl gear (old/new)
1986	-	-	5	Old
1987	-	-	3	Old
1988	3	Old	-	-
1989	4	Old	-	-
1990	-	Old	5	Old
1991	-	Old	5	Old
1992	5	Old	7	Old
1993	4	Old	8	Old
1994	-	Old	7	Old
1995	3	Old	5	Old
1996	4	Old	-	-
1997	3	Old	-	-
1998	-	-	-	-
1999	5	Old	-	-
2000	-	-	-	-
2001	-	-	2	Old
2002	-	-	-	-
2003	4	Old	4	New
2004	3	New	5	New
2005	5	New	-	-
2006	3	Old	5	Old
2007	4	New	2	New
2008	6	New	4	New

4.2.1 Data analysis

The swept area for each trawl was calculated in square nautical miles using the following equation as end coordinates were not available for all trawl stations:

$$sa = d \times s \times \frac{ws}{1852} \qquad \text{Equation 4.1}$$

Where *sa* is the swept area in square nautical miles, *d* is the duration in hours, *s* is the speed in knots, *ws* is the wing spread of the net in meters and 1852 is the number of meters in a nautical mile (Mackett 1973; Sparre and Venema 1998).

Biomass per species was standardised to catch per unit area (CPUA) to account for different trawl durations using the following equation:

$$CPUA_{ij} = \frac{cw_{ij}}{sa_j} \qquad \text{Equation 4.2}$$

Where *CPUA* is the density in kg.nm^{-2} of species *i* during trawl *j*, cw_{ij} is the catch weight in kg of species *i* during trawl *j* and sa_j is the swept area during trawl *j* (Mackett 1973; Sparre and Venema 1998).

Algoa Bay was divided into three similarly sized sectors (west, central and east) and two depth categories (inshore (0-50m) and offshore (51-100m)) (Figure 4.1). Trawls deeper than 100m were excluded from the analysis. Surficial sediment texture maps for Algoa Bay were obtained from past studies (Bremner 1978; Bremner and du Plessis 1982; Sampson 2002). Five categories of sediment have been distinguished in Algoa Bay, namely, gravel, sand, sandy-mud, muddy-sand and mud. Individual trawl stations were assigned a substrate type by intersecting the trawl locations with the surficial texture layer in ArcView 3.2. Mud areas were limited to two small patches in Algoa Bay and no trawls were conducted in these areas, allowing a comparison of a maximum of four substrate types. Categorical factors used in further analyses therefore included area (west, central, east), depth (<51m, 51-100m), season (autumn or spring), gear type (old, new), substrate (gravel, sand, sandy-mud, muddy-sand) and year.

(a) Diversity and Community analyses

The analysis was limited to teleosts, chondrichthyans and cephalopods as these were the primary groups landed by the trawl gear which are targeted by recreational and commercial fisheries within Algoa Bay. In order to minimise the noise resulting from chance occurrence, species which were recorded in less than 5% of the trawl stations were removed from the dataset (Ungaro *et al.* 1999; Massutí and Moranta 2003). Density data were fourth root transformed (Clarke and Green 1988; Ungaro *et al.* 1999; Catalán *et al.* 2006) to down weigh the influence of the dominant species on the resemblance measure prior to calculation of Bray-Curtis similarity matrices. Diversity and multivariate analyses of trawl density data were conducted in a similar manner as described in Chapter 3. The influence of factors on community structure was investigated using nMDS ordination and cluster dendograms, while ANOSIM was used to test for significant differences. The SIMPER routine was used to determine species that characterise a given group and species that contributed most to the dissimilarity between groups. Species richness, Shannon-Wiener Diversity and Pielou's Evenness indices were calculated as in Chapter 3. However, average taxonomic distinctness (AvTD), based on presence/absence data were determined rather than taxonomic diversity, and variability in taxonomic distinctness (VarTD) was also calculated for each trawl station. AvTD quantifies diversity based on the relatedness of species and distances between species through the taxonomic tree (Clarke and Warwick 2001a) and can therefore be calculated on presence/absence data using species lists, being independent of abundance or density data (Clarke and Warwick 2001a). VarTD indicates the variability in branch length between all species pairs within a sample, providing an indication of the variability and unevenness of the taxonomic tree at each station and is independent of sample size, number of species and value of AvTD within a sample (Tolimieri and Anderson 2010). Both these indices are calculated on presence/absence data, have been shown to be effective and sensitive in detecting shifts in communities, and lack dependence on the mean value and sample size (Clarke and Warwick 2001b; Warwick *et al.* 2002) and were therefore appropriate for the trawl dataset used in the analysis. Average taxonomic distinctness (Δ^+) is calculated as:

$$\Delta^+ = \left[\sum\sum_{i<j} \omega_{ij}\right] / \left[S(S-1)/2\right] \qquad \text{Equation 4.3}$$

Where ω_{ij} is the distinctness weight given to the path length linking species *i* and *j* and *S* is the number of species.

Variability in taxonomic distinctness (Λ^+) is calculated using the following formula (Clarke and Warwick 2001b; Warwick *et al.* 2002):

$$\Lambda^+ = \left[\sum\sum_{i<j}(\omega_{ij} - \Delta^+)^2\right] / \left[S(S-1)/2\right] \qquad \text{Equation 4.4}$$

Where ω_{ij} is the distinctness weight given to the path length linking species *i* and *j*, Δ^+ is the average taxonomic distinctness of the sample and *S* is the number of species.

The diversity indices were checked for normality using a Kolmogorov-Smirnov Test and the homogeneity of variances with Levene's Test, if assumptions were met a students t-test or one-way ANOVA was conducted. If assumptions were not met a non-parametric Mann-Whitney U test for paired comparisons or a Kruskal-Wallis ANOVA for comparison of multiple groups was undertaken. Where significant differences between multiple groups occurred, post hoc testing was conducted by pair wise Mann-Whitney U tests with a Bonferroni adjusted level of significance as described in Chapter 3.

Temporal stability of the demersal ichthyofaunal communities in Algoa Bay was assessed by ranking of the average annual species density using Kendall's coefficient of concordance (*W*) and obtaining their importance intervals (*Rij*). The higher the coefficient value (maximum of 1), the greater the stability of the community over time (Kendall 1962).

(b) Species relative density and size analysis

Analyses were conducted for 12 of the commercially important and abundant species captured during trawl surveys.

Standardised density data were not normally distributed and included several zero values for some but not all species. Trawl gear is by nature unselective resulting in the capture of numerous species while zero catches for many commercially important species are common. Long-term temporal trends in relative density were therefore investigated using the Delta-X approach, which is widely recognised as the most appropriate method for dealing with CPUE data where large proportions of zero catches are recorded (Stefánsson 1996; Punt *et al.* 2000; Ellender *et al.* 2010; Donovan 2010). The Delta-X approach is a two-step approach to modelling density data, whereby the probability of capture (*Pc*), or the presence or absence of a species in the total catch (1 or 0) is modelled separately from the positive catch (or in this case CPUA) ($CPUE_{Pos}$), and the two models are then combined to give a standardised estimate of density taking into account the zero-inflation in the dataset.

Due to the small overall sample size and the prevalence of positive or negative probabilities during some survey years, *Pc* was not modelled using a GLM and nominal weighted means of *Pc* were used to adjust the positive catch rates.

Small sample sizes precluded the inclusion of area and substrate type as explanatory factors in the GLM for positive catch rates. Factors taken into consideration for standardising positive catches therefore included season, depth and year, where season was either the spring or autumn survey period, depth was either inshore (<50m) or offshore (51-100m), and year was the year in which the survey was conducted (1986-2008). The positive catch rates approximated a gamma distribution and were modelled using a GLM with gamma distribution and the log-link function as described in Chapter 3. The model took the form:

$$log(CPUE_{Pos}) = \beta_0 + \beta_1(season) + \beta_2(depth) + \beta_3(year) + \varepsilon \quad \text{Equation 4.5}$$

Where $\beta_{0\text{-}i}$ are the estimated parameters of the explanatory variables and ε the normal error (McCullagh and Nelder 1995).

In order to obtain the standardised density the *Pc* and $CPUE_{Pos}$ were combined in the following manner (Stefánsson 1996):

$$\text{Standardised CPUA} = Pc \times CPUE_{Pos} \quad \text{Equation 4.6}$$

The Delta-X approach does not allow for the significance of specific explanatory variables on the overall CPUA (combined effects of *Pc* and positive catch) to be determined as it is conducted in a two-step process. As a result statistical analyses were conducted using the whole dataset for all explanatory variables under investigation (season, depth, gear, area and substrate) using non-parametric Mann-Whitney U tests for pairwise comparisons and Kruskal-Wallis ANOVAs for comparison of multiple groups (Petrakis *et al.* 2001). The influence of the change of gear type on density was investigated first. Where a significant effect was detected, only data from the old trawl gear were used to investigate the influence of additional explanatory variables on species density in order to eliminate potential bias due to gear selectivity. Post hoc testing of multiple comparisons among levels were conducted after Bonferroni correction as described in Chapter 3. In order to evaluate trends and stability in the density of individual species, the coefficient of variation (CV) of the density was calculated as outlined in Chapter 3 and plotted to illustrate temporal trends.

The influence of explanatory variables on the mean length of individual species was investigated using GLMs. Length is a continuous response variable which is typically left skewed and was therefore modelled using a GLM with a gamma error distribution and the log-link function as described in Chapter 3. The factors season, depth, year and area were used to model the length of individual species. All factors were as described above, while area was either the west, central or eastern sector of Algoa Bay (Figure 4.1). The GLM took the form:

$$Length = \beta_0 + \beta_1(season) + \beta_2(area) + \beta_3(depth) + \beta_4(year) + \varepsilon \quad \text{Equation 4.7}$$

Where $\beta_{0\text{-}i}$ are the estimated parameters of the explanatory variables and ε the error (McCullagh and Nelder 1995).

In order to visualise trends in density and length frequency of dominant species spatially, trawl data over the entire period were grouped per 5' grid cell and average annual density and length frequency distribution patterns were plotted spatially. The size at 50% maturity for each species investigated was obtained from published literature (Table 4.2) and used to display the distribution of juveniles and adults spatially (selected to the nearest 5cm class).

Statistical analyses were conducted in STATISTICA 9.0 and R while GIS analysis was conducted using ArcView 3.2 and ArcMap 9.2.

Table 4.2. Size at 50% maturity for each species investigated.

Species	Scientific name	Size at 50% maturity	Reference
Horse mackerel	*Trachurus trachurus capensis*	320 mm TL	(Hecht 1990; Kerstan and Leslie 1994)
Shallow-water hake	*Merluccius capensis*	Males 360 mm TL; Females 480 mm TL	(Botha 1986)
Panga	*Pterogymnus laniarius*	286mm TL	(Booth and Hecht 1997)
St Joseph	*Callorhinchus capensis*	Males 435mm FL; Females 496mm FL	(Freer and Griffiths 1993a)
Lesser gurnard	*Chelidonichthys queketti*	195mm TL	(Booth 1997)
White seacatfish	*Galeichthys feliceps*	Males 315mm FL; Females 295 mm FL	(Mann 2000)
Cape gurnard	*Chelidonichthys capensis*	Males 300mm TL; Females 340mm TL	(Hecht 1977; McPhail 1998)
Kob	*Argyrosomus sp.* (based on *A. inodorus)*	Male 290mm TL; Females 310mm TL	(Griffiths 1997c)
East coast sole	*Austroglossus pectoralis*	270-305mm TL	(Le Clus *et al.* 1994)
Carpenter	*Argyrozona argyrozona*	Males 222 FL; Females 206 mm FL	(Brouwer and Griffiths 2005b)
Kingklip	*Genypterus capensis*	520 mm TL	(Payne 1985)
Chokka-squid	*Loligo reynaudi*	Males 188-203mm; Females 173-181mm	(Olyott *et al.* 2006)

4.3 Results

4.3.1 Demersal ichthyofaunal communities from trawl surveys

A total of 123 trawl stations, 56 autumn and 67 spring, were conducted within Algoa Bay between 1986 and 2008 (Figure 4.1; Table 4.1) and recorded 35 species of chondrichthyans from 17 families, 95 species of osteichthyes from 46 families, 10 species of cephalopods from three families, and one agnathan species (Appendix 2).

Overall 22 species were common across the study area, being encountered at more than 50% of the trawl stations, and included ten chondrichthyans, 11 osteichthyes and one cephalopod species (Appendix 2). The Sparidae family dominated the catch accounting for 19% of the catch, followed by Carangidae (18%), Merlucciidae (8%), Clupeidae (8%) and Triglidae (7%), with these five families cumulatively accounting for 60% of the catch weight. The catch was dominated by few species with the top five and ten most abundant species accounting for 53% and 74% of the catch by weight respectively. The top five species included horse mackerel (18%), red tjor-tjor (12%), shallow-water hake (9%), panga (7%), and sardine (7%).

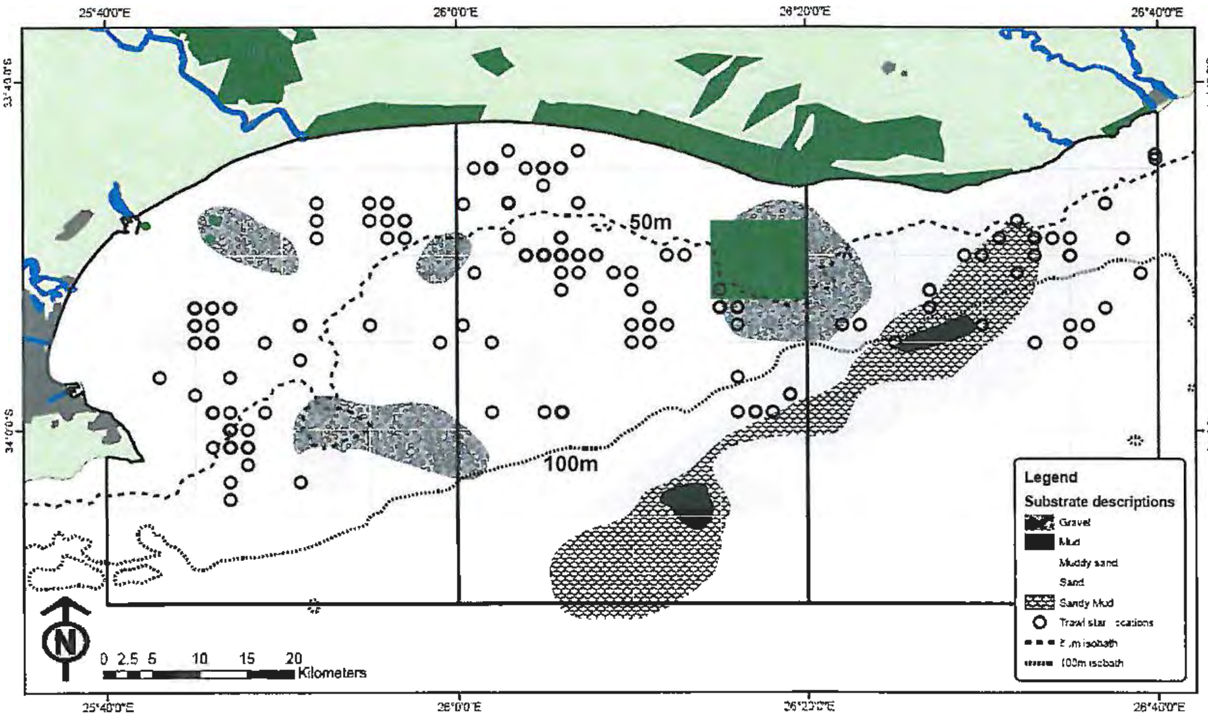


Figure 4.1. Spatial distribution of substrata habitats, division of the study area into three sectors and two depth categories and the location of research trawls within Algoa Bay. Existing Addo Elephant National Park areas are indicated in dark green.

4.3.2 Univariate measures of diversity

Area did not influence any diversity indices significantly (Table 4.3). Seasonal effects were apparent in Species Richness (p=0.007) and VarTD (p=0.028) with greater diversity occurring in spring in each case, while VarTD (p=0.019) was greater in autumn(Table 4.3). Substrate type influenced only species richness (p=0.12) with greater species over sand than sandy mud. Depth did not influence any indices significantly (Table 4.3), while gear type (change in gear configuration) influenced species richness significantly (p=0.045) with greater number of species sampled in the old gear compared to the new.

Annual trends in diversity indices are shown in Figure 4.2. Species Richness, Shannon Weiner and Pielou's Evenness show similar trends with an increase in diversity during the 1990s followed by a subsequent decline in the early 2000s. No clear trends are apparent for AvTD and VarTD.

Table 4.3. Influence of explanatory factors on diversity indices. Cells highlighted in green and orange indicate significant differences at $p<0.05$ and $p<0.001$ respectively.

Factor	Species richness	Shannon-Wiener	Pielou's Evenness	Average Taxonomic Distinctness	Variation in Taxonomic Distinctness
Area	F(2,123)=2.9 p=0.056 n/s	H(2, n= 123) =3.19 p =0.203 n/s	H(2, n=123)=1.52 p=0.467 n/s	F(2, 120)=2.54 p=0.0824 n/s	F(2, 120)=0.43 p=0.646 n/s
Season	t=2.7;df=121 p=0.007* Spring > Autumn	MWU (n=123) p=0.371 n/s	MWU (n=123) p=0.884 n/s	t=-2.40; df=121 p=0.019* Autumn>Spring	t=2.22; df=121 p=0.028* Spring>Autumn
Substrate	F(3,119)=3.8 p=0.012* Sand>Sandy mud	H(3, n= 123) =7.77 p =0.051 n/s	H(3, n=123)=3.57 p =0.310 n/s	F(3, 119)=2.0312 p=0.113 n/s	F(3, 119)=0.36 p=0.776 n/s
Depth	t=-0.16; df=121 p=0.877 n/s	MWU (n=123) p=0.110 n/s	MWU (n=123) p=0.114 n/s	t=-0.76; df=121 p=0.446 n/s	t=1.91; df=121 p=0.056 n/s
Gear	t=2.02; df=121 p=0.045* old>new	MWU (n=123) p=0.492 n/s	MWU (n=123) p=0.968 n/s	t=-0.44; df=121 p=0.656 n/s	t=1.49; df=121 p=0.137 n/s

n/s=not significant
* $p<0.05$
** $p<0.001$

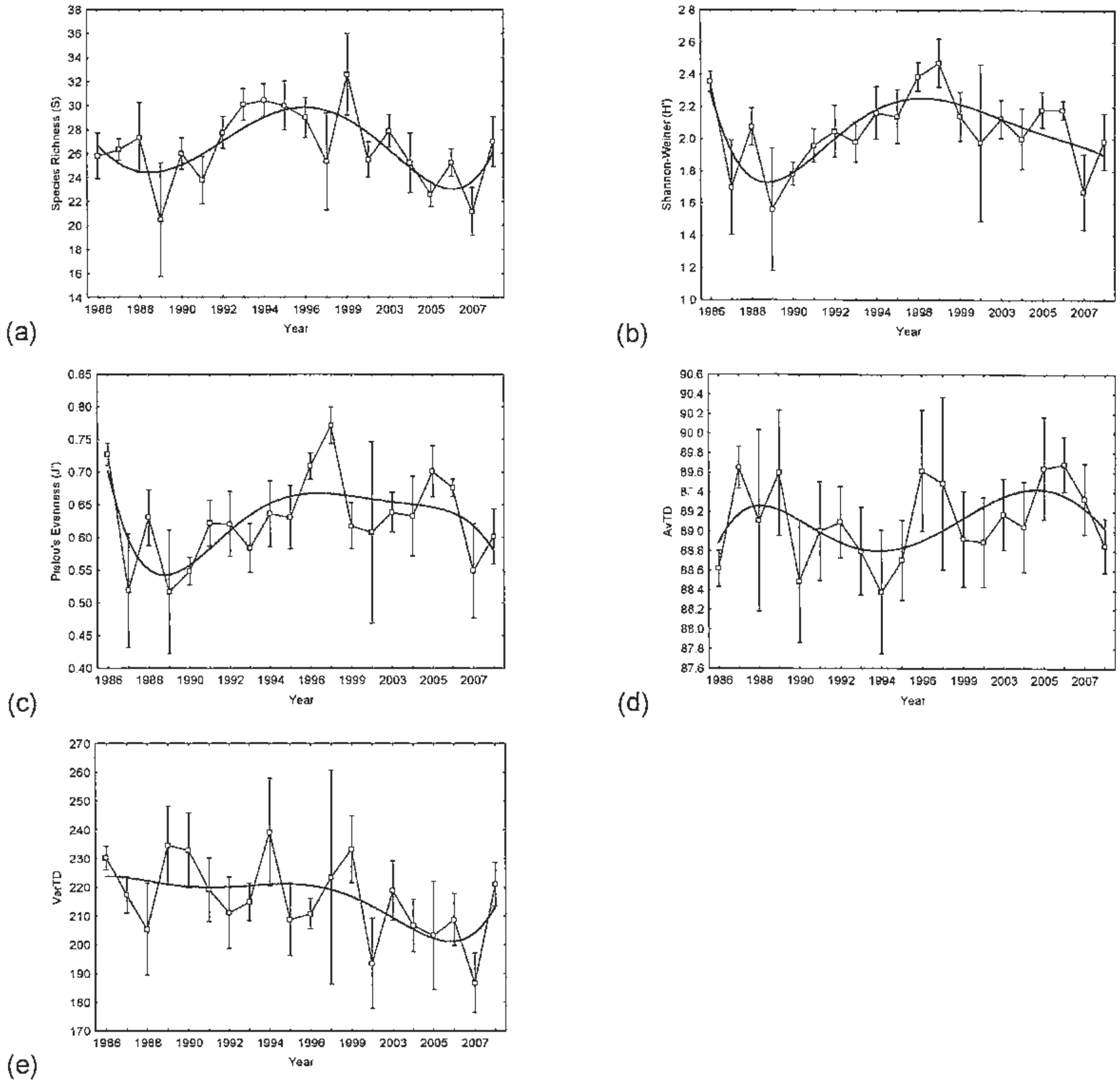


Figure 4.2. Annual trends in univariate indices of diversity (a) species richness (S), (b) Shannon-Wiener Diversity (H'), (c) Pielou's Evenness, (J'), (d) average taxonomic distinctness and (e) variation in taxonomic distinctness. General temporal trends are illustrated by 5th order polynomial regressions.

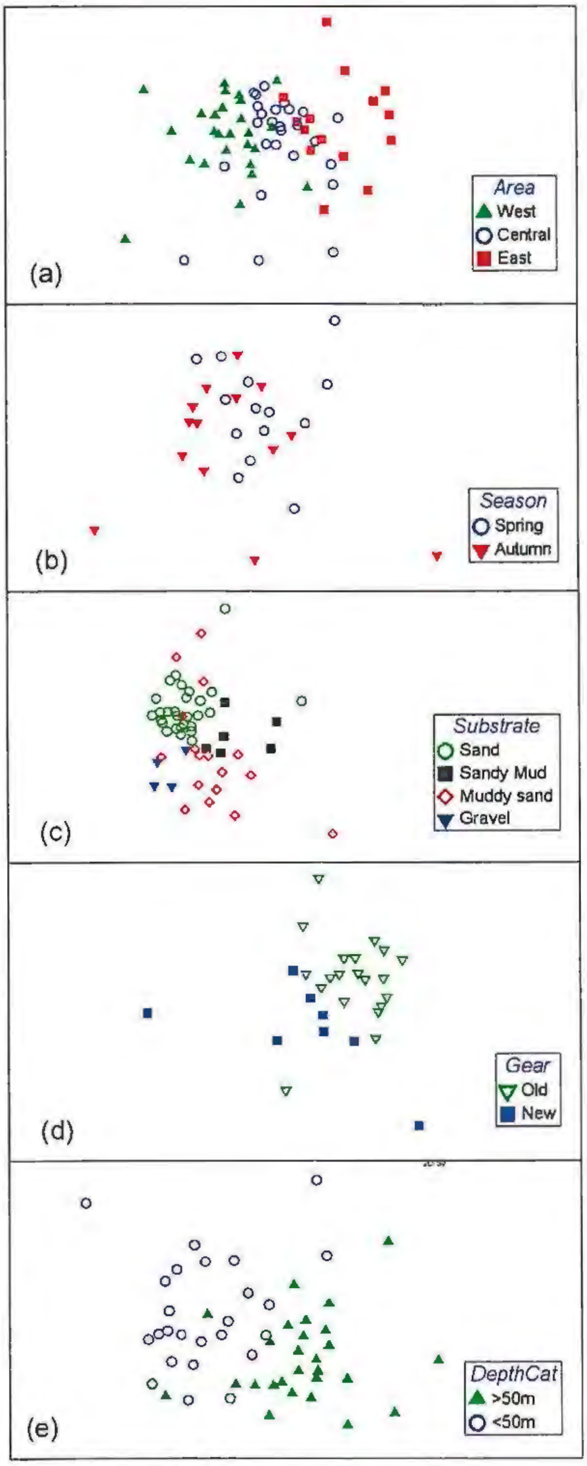


Figure 4.3. nMDS ordination of annual seasonal means by (a) area (stress 0.19), (b) season (stress 0.19) (c) substrate type (stress 0.18), (d) gear (stress 0.19), and (e) depth (stress 0.18).

4.3.3 Multivariate analysis

A spatial transition in demersal ichthyofaunal fish communities from east to west across Algoa Bay was evident in the nMDS ordination (Figure 4.3a) and ANOSIM tests confirmed significant spatial separation (Global R 0.247; p=0.001) with differences between all groups (Table 4.4).

nMDS ordination indicated no distinct seasonal separation with considerable overlap in communities between spring and autumn (Figure 4.3b); however, ANOSIM results were significant although the effect size was small (Global R 0.105; p=0.001) (Table 4.4).

Substrate type had a significant influence on community structure (Global R 0.213, p=0.001) with pairwise tests confirming significant differences between sand and both muddy-sand and sandy-mud substratum.

Differences in community structure as a result of old and new trawl gear configurations were evident with spatial separation in the nMDS ordination, which was confirmed through the results of ANOSIM tests (Global R 0.254; p=0.001) (Table 4.4).

The influence of depth was evident with two communities clearly distinguishable in the plot area and ANOSIM results indicating that depth had the strongest influence on differences in community structure (Global R 0.329; p=0.001).

The nMDS ordination with a temporal trajectory overlay for all seasons and years combined indicated strong clustering for the majority of sites with little differentiation between autumn and spring samples (Figure 4.4a). Despite some seasonal samples being distinguished from the main cluster of samples no clear directional temporal changes in community structure were apparent. Similarly, despite the new gear samples grouping closely together, five of the new gear samples grouped within 71% similarity to the old gear samples. Similar patterns were evident on separation of the seasonal samples (Figure 4.4b and c) with a combination of old and new gear samples grouping within 71% similarity, while few old and new gear samples were of lower similarity.

Table 4.4. Results of ANOSIM tests of significance of factors influencing community structure.

Factor	Global R	p value	Significant pairwise comparisons (p<0.05)
Area	0.247	0.001 **	West ≠ Central; West ≠ East; East ≠ Central
Season	0.105	0.001 **	Spring ≠ Autumn
Substrate	0.213	0.001 **	Sand ≠ Muddy-sand, Sandy-mud
Gear	0.254	0.001 **	Old ≠ New
Depth	0.329	0.001 **	Shallow ≠ Deep

n/s=not significant
* p<0.05
** p<0.01

Using data from the old trawl configuration only, Kendall's coefficient of concordance indicated relatively high stability in the rank order of species density between years for autumn (W=0.75; X^2, r = 384.6, p<0.05) and spring (W=0.76 X^2, r = 484.2, p<0.05) surveys respectively. However, on inclusion of both old and new gear types there was a decrease in community stability (autumn W=0.70; X^2, r = 779.7, p<0.05; spring W=0.73; X^2, r = 652.7, p<0.05).

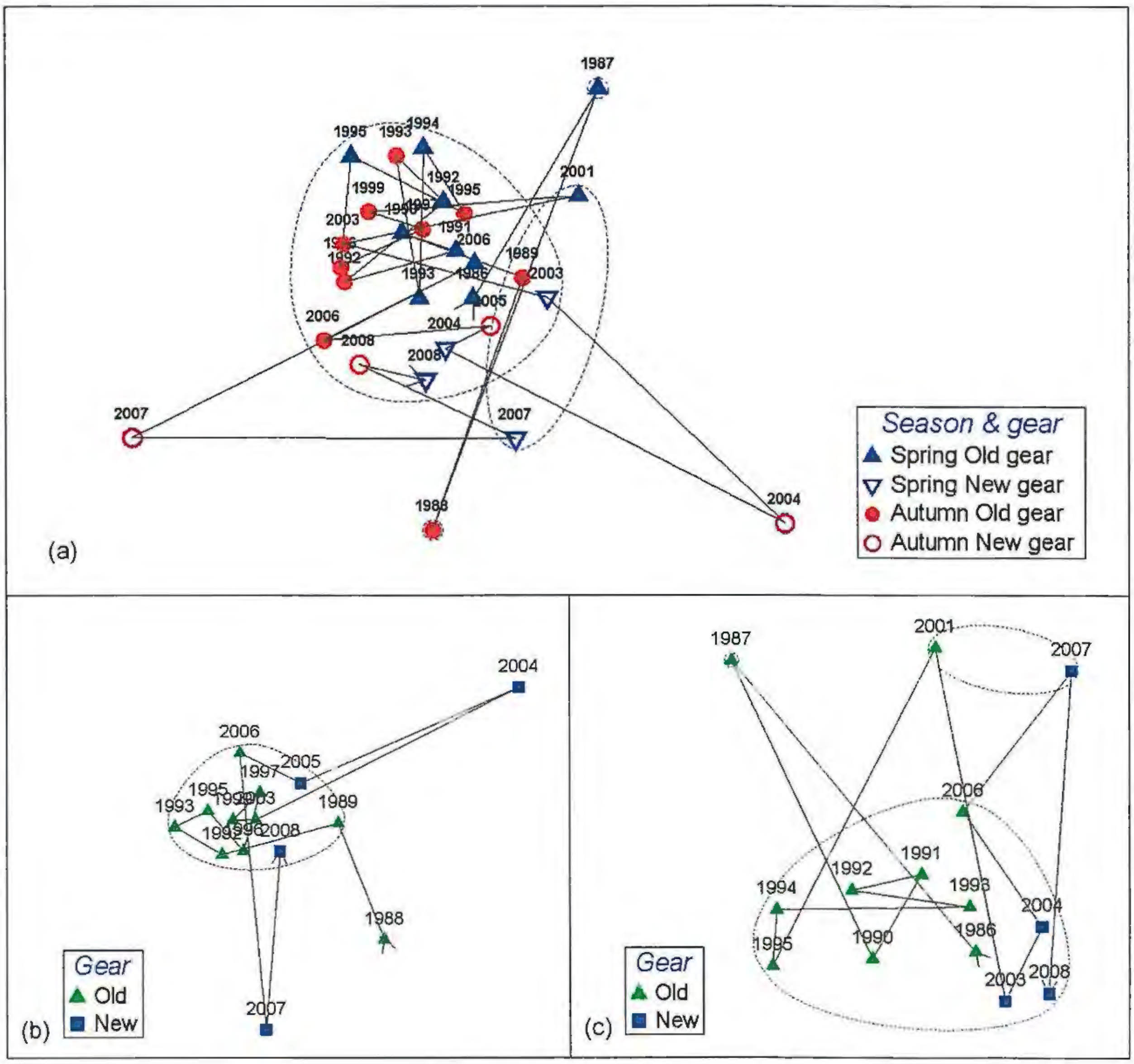


Figure 4.4. (a) Ordination of annual seasonal density data for demersal communities within Algoa Bay (Stress 0.19), and ordinations of (b) autumn surveys and (c) spring surveys indicating similarity between old and new gear configurations. Dashed lines indicates groupings of 71% similarity.

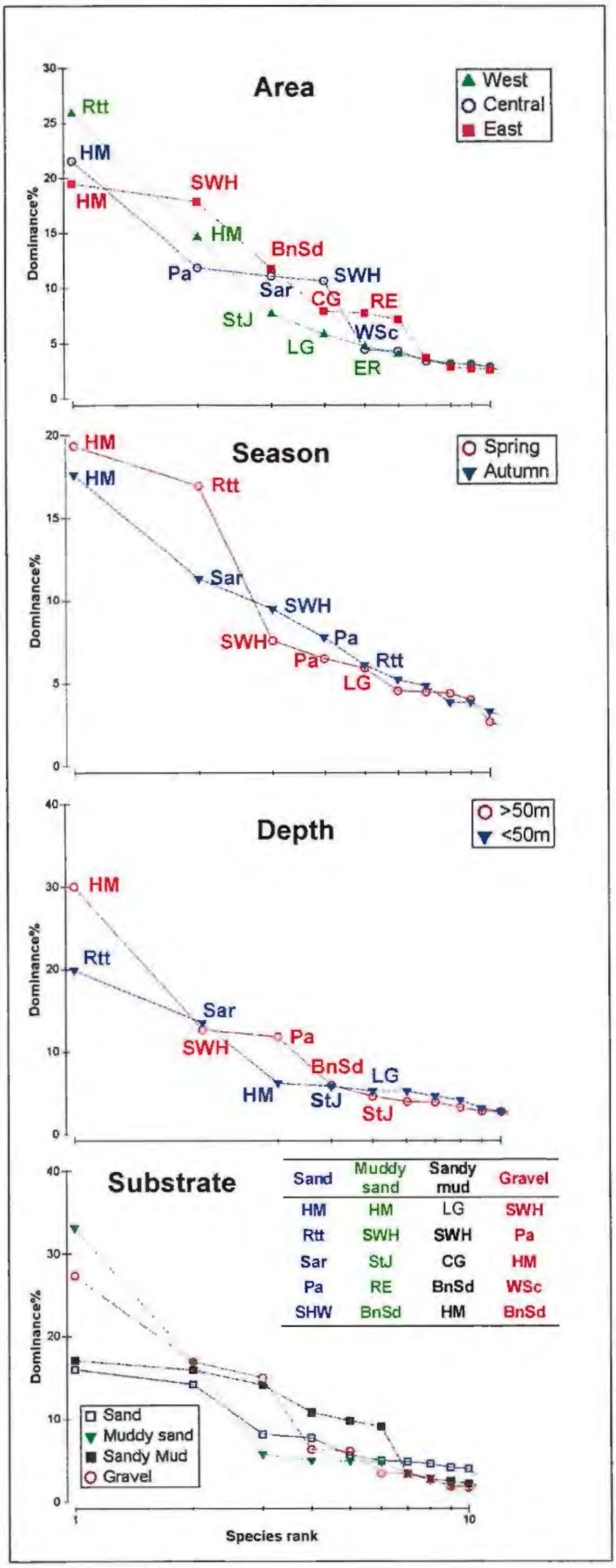


Figure 4.5. Influence of sectors, season, depth and substrate on ichthyofaunal community structure over the trawl grounds in Algoa Bay. (HM=horse mackerel; Rtt=red tjor-tjor; SWH=shallow-water hake; Pa=panga; LG=lesser gurnard; StJ=St Joseph; Sar=sardine; ER=eagle ray; WSc=white seacatfish; BnSd=bluntnose spiny dogfish; RE=redeye round herring).

Red tjor-tjor dominated the catch by weight (26%) in the western sector of Algoa Bay followed by horse mackerel (15%), St Joseph (8%), lesser gurnard (6%) and eagle ray (5%) (Figure 4.5). Horse mackerel dominated in both the central and eastern sectors of Algoa Bay representing 22% and 20% respectively, while panga accounted for 12% and 8%, and shallow-water hake 11% and 18% in the central and eastern sectors, respectively.

Horse mackerel was the dominant species in both autumn (18%) and spring (19%) representing similar proportions of the catch weight (Figure 4.5). Sardine (11%), shallow-water hake (10%), panga (8%) and red tjor-tjor (6%) were the next most dominant species in autumn. The dominant species in spring included red tjor-tjor (17%), shallow-water hake (8%), panga (7%) and lesser gurnard (6%).

Horse mackerel (30%), shallow-water hake (13%) and panga (12%) dominated in deeper water cumulatively accounting for 55% of the catch, while red tjor-tjor (20%), sardine (13%) and horse mackerel (6%) were the three most dominant species in the shallow waters of Algoa Bay (Figure 4.5).

The three most dominant species over sandy substratum were horse mackerel (16%), red tjor-tjor (14%) and sardine (8%) (Figure 4.5). Horse mackerel (33%), shallow-water hake (17%) and St Joseph (6%) were the most dominant over muddy-sand, while lesser gurnard (17%), shallow-water hake (16%) and Cape gurnard (14%) dominated the catch over sandy-mud. Shallow-water hake (27%), panga (17%) and horse mackerel (15%) were dominant over gravel substrates.

4.3.4 Trends in species density and mean length

(a) Horse mackerel

Horse mackerel density was influenced significantly by depth ($p<0.001$) and area ($p=0.032$) with higher densities in deeper offshore waters than the shallower inshore waters (Table 4.5). Although area was a significant factor, spatial differences between areas were not detected by pairwise comparisons (Table 4.5). However, the central region contributed more to the average annual catch and lower proportions of horse mackerel catch were evident in the shallow western and central regions of the bay (Figure 4.6). Temporal trends in horse mackerel catch indicated a significant peak in 1993 with a progressive decline in density (Figure 4.7). The proportion of total catch was also highly variable peaking in 1993 at 43% with a subsequent peak of 41% in 2004 with no clear trends discernable (Figure 4.7). The CV of density varied considerable between years, but a general increase in variability was apparent with time (Figure 4.7).

Depth ($p<0.001$), area ($p<0.001$) and season ($p<0.001$) all influenced the length of horse mackerel significantly (Table 4.6). The mean length of horse mackerel was greater in deeper water, and decreased from east to west across the bay (Figure 4.6). In addition the proportion of mature fish in the catch was greater in deeper water and in the central and eastern sectors of Algoa Bay (Figure 4.6). Larger horse mackerel were present in Algoa Bay during spring than autumn. Annual trends in the mean length of horse mackerel in Algoa Bay indicate a progressive decline from 2000 onwards (Figure 4.7).

(b) Shallow-water hake

The relative density of shallow-water hake was influenced significantly by depth ($p<0.001$), area ($p<0.001$) and substrate ($p<0.01$) (Table 4.5). Greater densities occurred in deeper water and in the central and eastern sectors of the bay with only small catches in the western sector (Figure 4.6). Hake showed a preference for muddy-sand and gravel over sandy substrata (Table 4.5). Annual trends in density indicate high variability with no distinct trend apparent; however, an increase in density is evident from 2005 to 2008 (Figure 4.7). The proportion of shallow-water hake in catches within Algoa Bay indicates a general increase over the time period considered, with a recent peak in 2008 at 22% of the average annual catch. Inter-annual variability in the CV of the catches was high with no clear increasing or decreasing trend evident (Figure 4.7).

Shallow-water hake length in Algoa Bay was influenced by depth ($p<0.001$), area ($p<0.001$) and season ($p<0.001$) (Table 4.6). Larger fish were captured in deeper water, during autumn and in the western region of Algoa Bay (Table 4.6; Figure 4.6). However, very few fish above the size at 50% maturity were captured (Figure 4.6). Long-term temporal trends in the mean length of shallow-water hake shows a decline from 1996 to 2001, which was followed by a subsequent increase in size. However, no clear long-term trend is apparent (Figure 4.7).

Table 4.5. Influence of trawl gear, depth, area, substrate and season on the density of important species within Algoa Bay from 123 research trawl stations. Cells highlighted in green and orange represent indicate significant differences at p<0.05 and p<0.001 respectively.

Species	Gear	Depth	Area	Substrate	Season
Horse mackerel *Trachurus trachurus capensis*	MWU (n=123) p=0.299 n/s	MWU (n=123) p<0.001 Deep>Shallow	H(2, n=123)=6.97 p=0.032 Not detected	H(3, n=123)=5.01 p=0.171 n/s	MWU (n=123) p=0.204 n/s
Shallow-water hake *Merluccius capensis*	MWU (n=123) p=0.591 n/s	MWU p<0.001 Deep > Shallow	H(2, n=123)=22.61 p<0.001 Central & East > West	H(3, n=123)=16.08 p=0.001 Muddy-sand & Gravel > Sand	MWU (n=123) p=0.073 n/s
Panga *Pterogymnus laniarius*	MWU (n=123) p=0.560 n/s	MWU (n=123) p<0.001 Deep > Shallow	H(2, n=123)=14.11 p<0.001 Central & East > West	H(3, n=123)=7.02 p=0.071 n/s	MWU (n=123) p=0.285 n/s
St Joseph *Callorhinchus capensis*	MWU (n=123) p=0.003 Old>New	MWU (n=90) p=0.012 Shallow > Deep	H(2, n=90)=27.23 p<0.001 West > Central > East	H(3, n=90)=3.40 p=0.334 n/s	MWU (n=90) p=0.618 n/s
Lesser gurnard *Chelidonichthys queketti*	MWU (n=123) p<0.001 Old>New	MWU (n=90) p=0.047 Deep>Shallow	H(2, n=90)=6.20 p=0.045 West > Central & East	H(3, n=90)=5.47 p=0.141 n/s	MWU (n=90) p=0.154 n/s
Cape gurnard *Chelidonichthys capensis*	MWU (n=123) p=0.016 Old>New	MWU (n=90) p=0.112 n/s	H(2, n=90)=6.16 p=0.048 Not detected	H(3, n=90)=6.99 p=0.072 n/s	MWU (n=90) p=0.355 n/s
White seacatfish *Galeichthys feliceps*	MWU (n=123) p=0.603 n/s	MWU (n=123) P<0.001 Shallow > Deep	H(2, n=123)=7.71 p=0.021 West > East	H(3, n=123)=14.42 p=0.002 Gravel & Sand > Muddy-sand	MWU (n=123) p=0.286 n/s
East coast sole *Austroglossus pectoralis*	MWU (n=123) p<0.001 Old>New	MWU (n=90) p=0.651 n/s	H(2, n=90)=3.63 p=0.163 n/s	H(3, n=90)=3.88 p=0.274 n/s	MWU (n=90) p=0.239 n/s
Chokka-squid *Loligo reynaudi*	MWU (n=123) p=0.079 n/s	MWU (n=123) p=0.029 Shallow>Deep	H(2, n=123)=16.85 p<0.001 West>Central & East	H(3, n=123)=13.58 p=0.004 No detected	MWU (n=123) p=0.082 n/s
Kob *Argyrosomus sp.*	MWU (n=123) p=0.860 n/s	MWU (n=123) p=0.042 Deep > Shallow	H(2, n=123)=10.12 p=0.006 Central > West	H(3, n=123)=6.55 p=0.087 n/s	MWU (n=123) p<0.001 Spring > Autumn
Kingklip *Genypterus capensis*	MWU (n=123) p=0.062 n/s	MWU (n=123) p<0.001 Deep > Shallow	H(2, n=123)=25.37 p<0.001 East > West	H(3, n=123)=26.20 p<0.001 Muddy-sand > Sand	MWU (n=123) p=0.359 n/s
Carpenter *Argyrozona argyrozona*	MWU (n=123) p=0.352 n/s	MWU (n=123) p=0.026 Deep > Shallow	H(2, n=123)=4.26 p=0.119 n/s	H(3, n=123))=1.28 p=0.734 n/s	MWU (n=123) p=0.931 n/s

(c) Panga

The density of panga was influenced significantly by depth (p<0.001) and area (p<0.001) (Table 4.5). Greater catches of panga were made in deeper water and in the central and eastern sectors of the bay than shallow areas and the western sector (Figure 4.6). Long-term temporal trends indicate an increase in relative density from 1991 to 1999 which was followed by a progressive decrease (Figure 4.7). The proportion of panga in the annual catch was also highly variable, peaking in 1999 at 24% but declining to 6% of the average annual catch in 2008 (Figure 4.7). The CV of panga density indicates high variability with no clear temporal trend apparent (Figure 4.7).

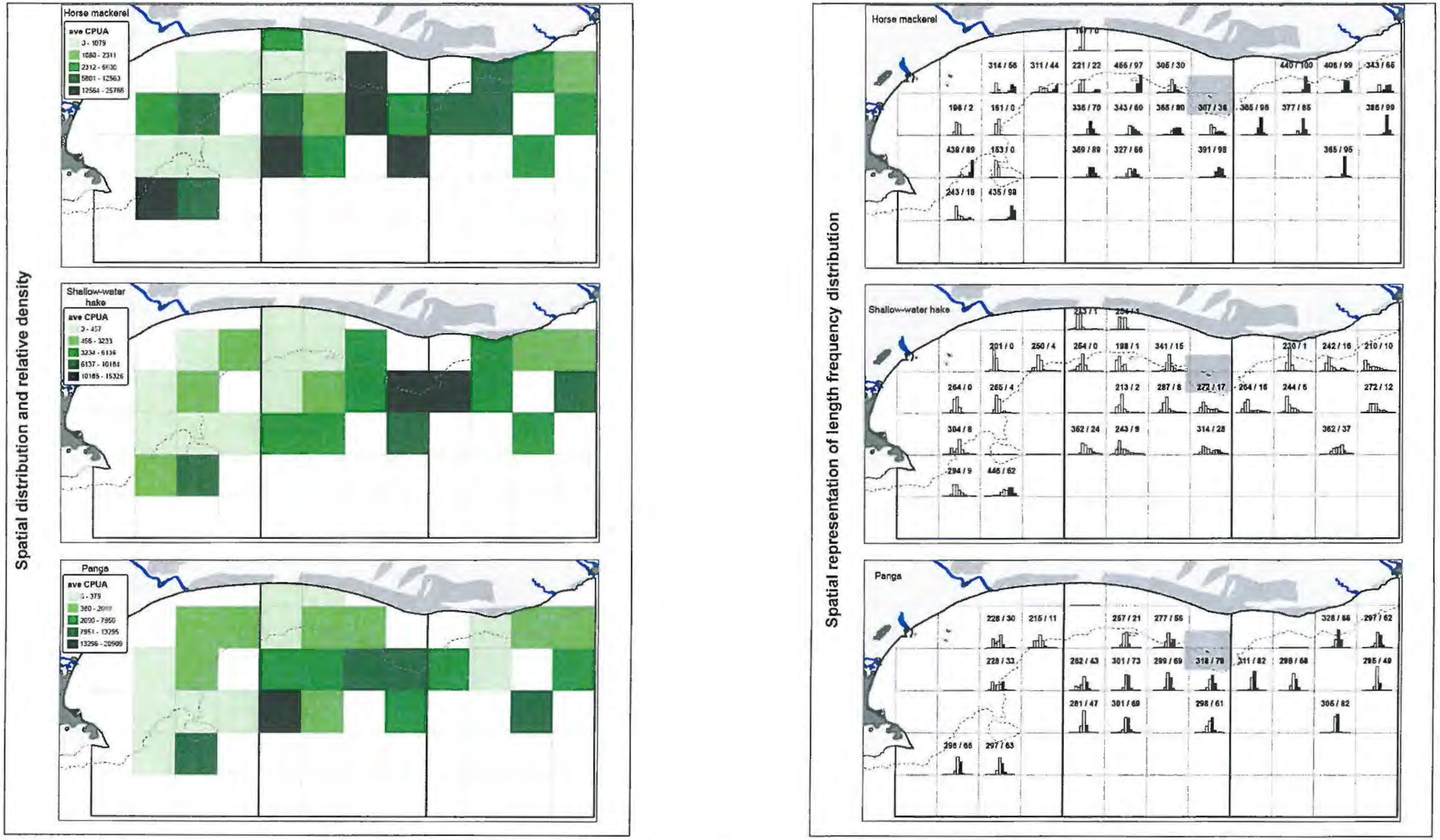


Figure 4.6. Spatial representation of the average annual density (kg.nm^{-2}) (left) and length frequency (right) of horse mackerel (top left), shallow-water hake (middle left) and panga (bottom left) per 5' grid cell in Algoa Bay. Numbers above length frequency histograms indicate mean length (TL mm) / % mature, white bars indicate size below 50% maturity and black bars above. Existing Addo Elephant National Park areas are indicated in darker grey. Dashed line is the 50m isobath.

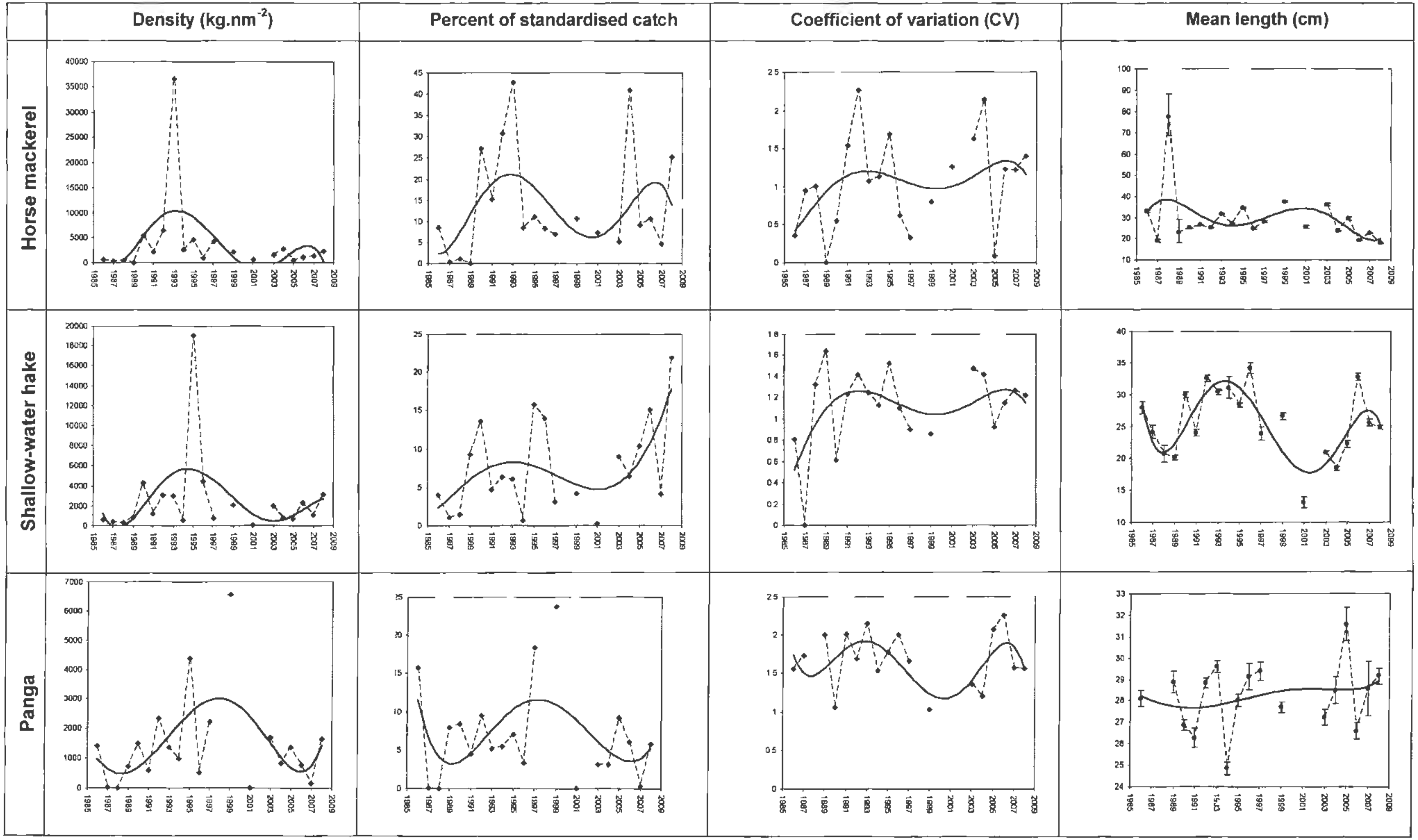


Figure 4.7. Annual trends in the density, percentage of catch composition, CV of density and mean length (±95% confidence intervals) of horse mackerel (top), shallow-water hake (middle) and panga (bottom). General temporal trends are illustrated by 5[th] order polynomial regressions (solid line).

Table 4.6. Influence of factors on the mean length of the most dominant and commercially important species using GLM (year, depth, area and season). Cells highlighted in green and orange represent significant differences at $p<0.05$ and $p<0.001$ respectively.

Species	Year	Season	Depth	Area
Horse mackerel	Wald $X^2(19)=53914.6$ $p<0.001$ Figure 4.7	Wald $X^2(1)=5389.7$ $p<0.001$ Spring > Autumn	Wald $X^2(1)=4597.2$ $p<0.001$ Deep > Shallow	Wald $X^2(2)=8738.0$ $p<0.001$ East > Central > West
Shallow-water hake	Wald $X^2(19)=7858.0$ $p<0.001$ Figure 4.7	Wald $X^2(1)=226.8$ $p<0.001$ Autumn>Spring	Wald $X^2(1)=200.3$ $p<0.001$ Deep > Shallow	Wald $X^2(2)=205.5$ $p<0.001$ West > Central > East
Panga	Wald $X^2(16)=1335.0$ $p<0.001$ Figure 4.7	Wald $X^2(1)=34.0$ $p<0.001$ Spring > Autumn	Wald $X^2(1)=18.2$ $p<0.001$ Deep > Shallow	Wald $X^2(2)=1498.2$ $p<0.001$ East & Central > West
St Joseph	Insufficient data	MWU $p<0.001$ Spring > Autumn	MWU $p<0.001$ Deep > Shallow	$H(2)=51.9$ $p<0.001$ East I > West
Lesser gurnard	Wald $X^2(14)=2478.4$ $p<0.001$ Figure 4.9	Wald $X^2(1)=56.4$ $p<0.001$ Spring > Autumn	Wald $X^2(1)=661.7$ $p<0.001$ Deep > Shallow	Wald $X^2(2)=957.8$ $p<0.001$ East > Central > West
Cape gurnard	Wald $X^2(15)=912.8$ $p<0.001$ Figure 4.9	Wald $X^2(1)=107.7$ $p<0.001$ Spring > Autumn	Wald $X^2(1)=272.5$ $p<0.001$ Deep > Shallow	Wald $X^2(2)=1486.8$ $p<0.001$ East > Central > West
White seacatfish	Insufficient data	Insufficient data	MWU $p<0.001$ Deep > Shallow	$H(2)=30.6$ $p<0.001$ East & Central > West
Chokka-squid	Wald $X^2(19)=2934.6$ $p<0.001$ Figure 4.11	Wald $X^2(1)=139.0$ $p<0.001$ Spring > Autumn	Wald $X^2(1)=87.5$ $p<0.001$ Shallow > Deep	Wald $X^2(2)=104.6$ $p<0.001$ West > Central & East
East coast sole	Wald $X^2(19)=969.2$ $p<0.001$ Figure 4.11	Wald $X^2(1)=12.3$ $p<0.001$ Spring > Autumn	Wald $X^2(1)=593.4$ $p<0.001$ Deep > Shallow	Wald $X^2(2)=354.5$ $p<0.001$ Central & East > West
Kob	Wald $X^2(17)=424.1$ $p<0.001$ Figure 4.13	Wald $X^2(1)=63.3$ $p<0.001$ Autumn >Spring	Wald $X^2(1)=52.1$ $p<0.001$ Deep > Shallow	Wald $X^2(2)=234.1$ $p<0.001$ East > Central > West
Kingklip	Wald $X^2(14)=224.9$ $p<0.001$ Figure 4.13	Wald $X^2(1)=148.4$ $p<0.001$ Spring > Autumn	Wald $X^2(1)=23.2$ $p<0.001$ Deep > Shallow	Wald $X^2(2)=17.3$ $p<0.001$ East > Central
Carpenter	Wald $X^2(14)=116.7$ $p<0.001$ Figure 4.13	Wald $X^2(1)=14.3$ $p<0.001$ Autumn > Spring	Wald $X^2(1)=79.5$ $p<0.001$ Deep > Shallow	Wald $X^2(2)=4.9$ $p=0.087$ n/s

Depth ($p<0.001$), area ($p<0.001$) and season ($p<0.001$) were all significant explanatory factors for the mean length of panga (Table 4.6). Mean length of panga was greater in deeper water, in the east and central regions of Algoa Bay than the west, with greater proportions of the landed catch in the central and eastern sectors being above the size of 50% maturity, while panga within the inshore area and western sector tended to be smaller and immature (Figure 4.6). There was considerable variability between years and no clear long-term directional change in mean length of panga was apparent (Figure 4.7).

(d) St Joseph

The density of St Joseph was influenced significantly by gear ($p=0.003$), depth ($p=0.012$) and area ($p<0.001$) (Table 4.5). Catches were greater with the old trawl gear than the new, and greater densities of St Joseph occurred in the shallower water and decreased from west to east across Algoa Bay (Figure 4.8). No clear long-term trend was apparent in relative density (Figure 4.9). The percentage contribution to the total catch was relatively stable, but the CV in St Joseph density indicated an increase from 2001 onwards (Figure 4.9).

Few years of length frequency data were available for St Joseph. Depth ($p<0.001$), area ($p<0.001$) and season ($p<0.001$) all influenced the mean length of St Joseph significantly (Table 4.6). Larger fish were caught in the deeper and eastern and central regions of Algoa Bay with smaller fish in the western and inshore regions (Figure 4.8).

(e) Lesser gurnard

Lesser gurnard density was influenced significantly by gear type ($p<0.001$), depth ($p=0.047$) and area ($p=0.045$) (Table 4.5). Densities were higher using the old trawl gear and in deeper waters of Algoa Bay (Table 4.5). Although densities appear similar across Algoa Bay (Figure 4.8) they were higher in the western region than the eastern and central regions (Table 4.5). Long-term temporal trends in density indicate a significant declining trend from 1995 onwards, while the contribution to the average annual catch remained relatively stable only indicating a decline from 2006 (Figure 4.9). Large temporal variability was apparent with an increase in the CV of lesser gurnard density from 2001 to onwards (Figure 4.9).

The size of lesser gurnard was influenced significantly by depth ($p<0.001$), area ($p<0.001$) and season ($p<0.001$) (Table 4.6). Larger lesser gurnard were captured in deeper waters in Algoa Bay (Table 4.6). Fish size decreased from east to west across Algoa Bay (Table 4.6). Larger fish were captured during autumn than spring (Table 4.6). Long-term trends in mean length indicate a progressive increase in mean length from 1993 to 1999. However, subsequent to this the variability in mean length increased considerably and indicated a general declining trend (Figure 4.9).

(f) Cape gurnard

Trawl gear ($p=0.016$) and area ($p=0.046$) were the only factors which had a significant effect on Cape gurnard density (Table 4.5). Catches were higher with the old trawl gear and although pairwise tests between areas did not detect significant differences, higher densities were evident in the eastern region where two grids contributed significantly (17.2 and 22.8%) to the average annual catch (Figure 4.8). Long-term trends indicate an increase in the density of Cape gurnard from 1989 to 1999 after which large variability in density occurred, with a general decline discernable (Figure 4.9). The CV of the Cape gurnard density was highly variable between years with an increase evident from 2001 onwards.

Depth ($p<0.002$), area ($p<0.001$) and season ($p<0.001$) influenced the size of Cape gurnard significantly (Table 4.6). Larger fish occurred in deeper water and in the eastern region and central region, where a larger proportion of the population were above size at 50% maturity, than the western region of Algoa Bay (Figure 4.8). Long-term trends in Cape gurnard size indicate a general declining trend in mean length (Figure 4.9).

(g) White seacatfish

The density of white seacatfish was influenced significantly by depth ($p<0.001$), area ($p=0.021$) and substrate ($p=0.002$) (Table.5). Density was greater in shallower than deeper water and greater in the western than the eastern sector of the bay (Table 4.5). The inshore areas of the central sector contributed most to the annual catch (Figure 4.10). Preference was shown for gravel and sand over muddy-sand substrate (Table 4.5). Long-term trends indicated no clear trends in density or the contribution to the average annual catch (Figure 4.11). However an increase in the variability in the density of white seacatfish is apparent.

The size of white seacatfish was influenced significantly by depth ($p<0.001$) and area ($p<0.001$) with larger fish occurring in deeper water, in the central and eastern sectors of Algoa Bay (Table 4.6).

(h) East coast sole

Trawl gear was the only factor which had a significant effect on the catch of east coast sole ($p<0.001$) with greater densities using the old gear configuration compared to the new configuration (Table 4.5). Although spatial trends were not significant, a greater proportion of catch was landed in the central region (Figure 4.10). Temporal trends indicated a decline in the density and the percentage contribution to the total catch from 1999 onwards, with no clear trends in CV (Figure 4.11).

Depth ($p<0.001$), area ($p<0.029$) and season ($p<0.001$) were all significant predictors of mean length for east coast sole (Table 4.6). Fish were larger in deeper water and in the central and eastern regions than the western region (Table 4.5; Figure 4.10). A greater proportion of juvenile fish occurred in the shallow and western sectors of Algoa Bay (Figure 4.10). Annual trends in size were also evident with an increase in the mean length from 1989 to 1996 followed by a steady decline to 2008 (Figure 4.11).

(i) Chokka-squid

The density of chokka-squid was influenced significantly by depth ($p=0.029$), area ($p<0.001$) and substrate ($p=0.004$), with higher density in shallower water, in the western sector of the bay and over sandy substrata (Table 4.5). Chokka-squid density was greatest inside of the Riy Banks reef complex which accounted for 39% of the average annual catch (Figure 4.10). In addition the area off the Sundays River mouth contributed 13% to the Algoa Bay catch. No clear temporal trends in density or associated variability were apparent (Figure 4.11).

Chokka-squid size was influenced by depth ($p<0.001$), area ($p<0.001$) and season ($p<0.001$) (Table 4.6) with larger chokka-squid occurring inshore, in the western sector and during spring (Table 4.5). A general decrease in mean length was apparent over time (Figure 4.11).

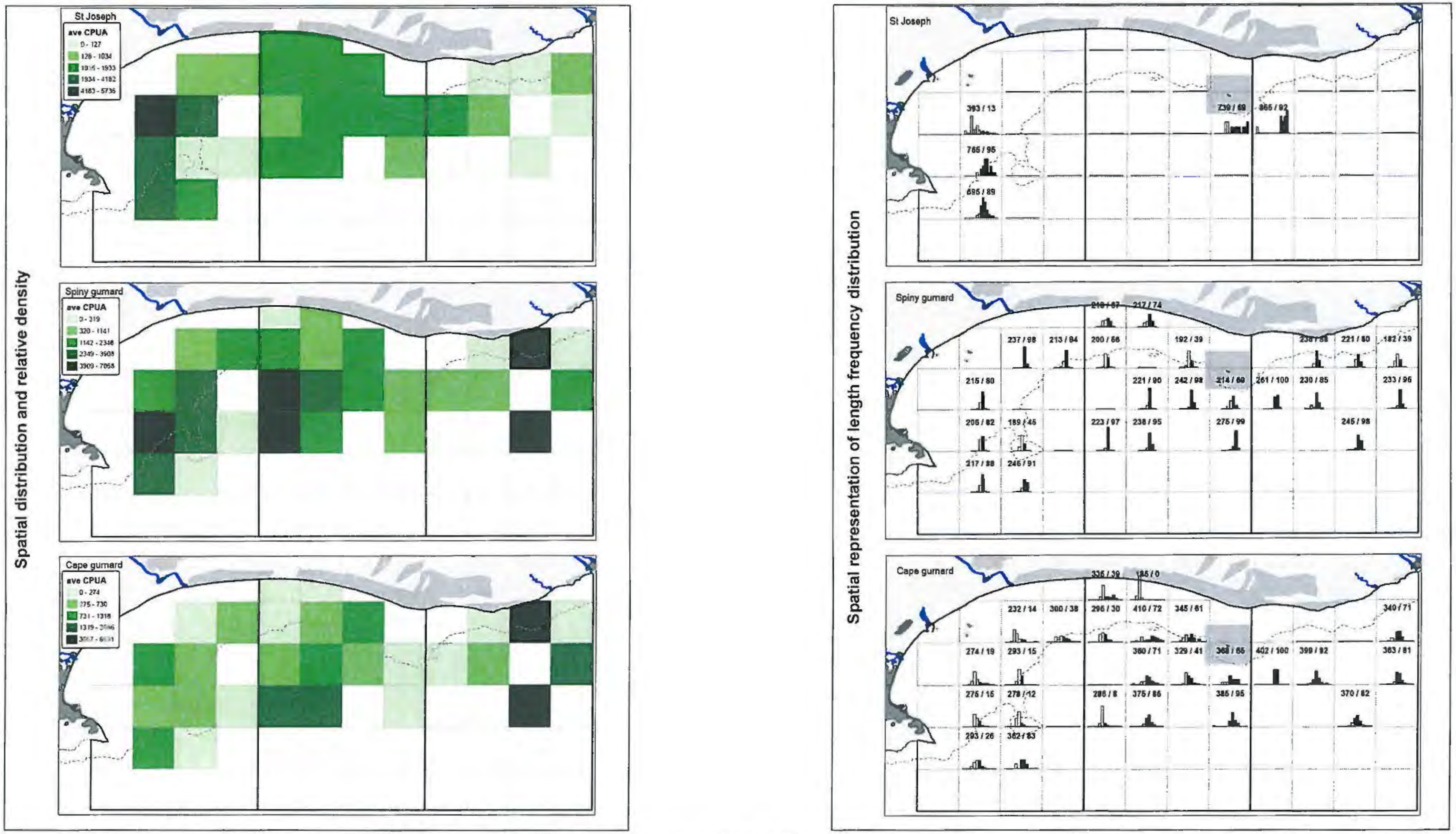


Figure 4.8. Spatial representation of the average annual density (left) and length frequency (right) of St Joseph (top left), lesser gurnard (middle left) and Cape gurnard (bottom left) per 5' grid cell in Algoa Bay. Numbers above length frequency histograms indicate mean length (TL mm) / % mature, white bars indicate size below 50% maturity and black bars above. Existing Addo Elephant National Park areas are indicated in darker grey. Dashed line is the 50m isobath.

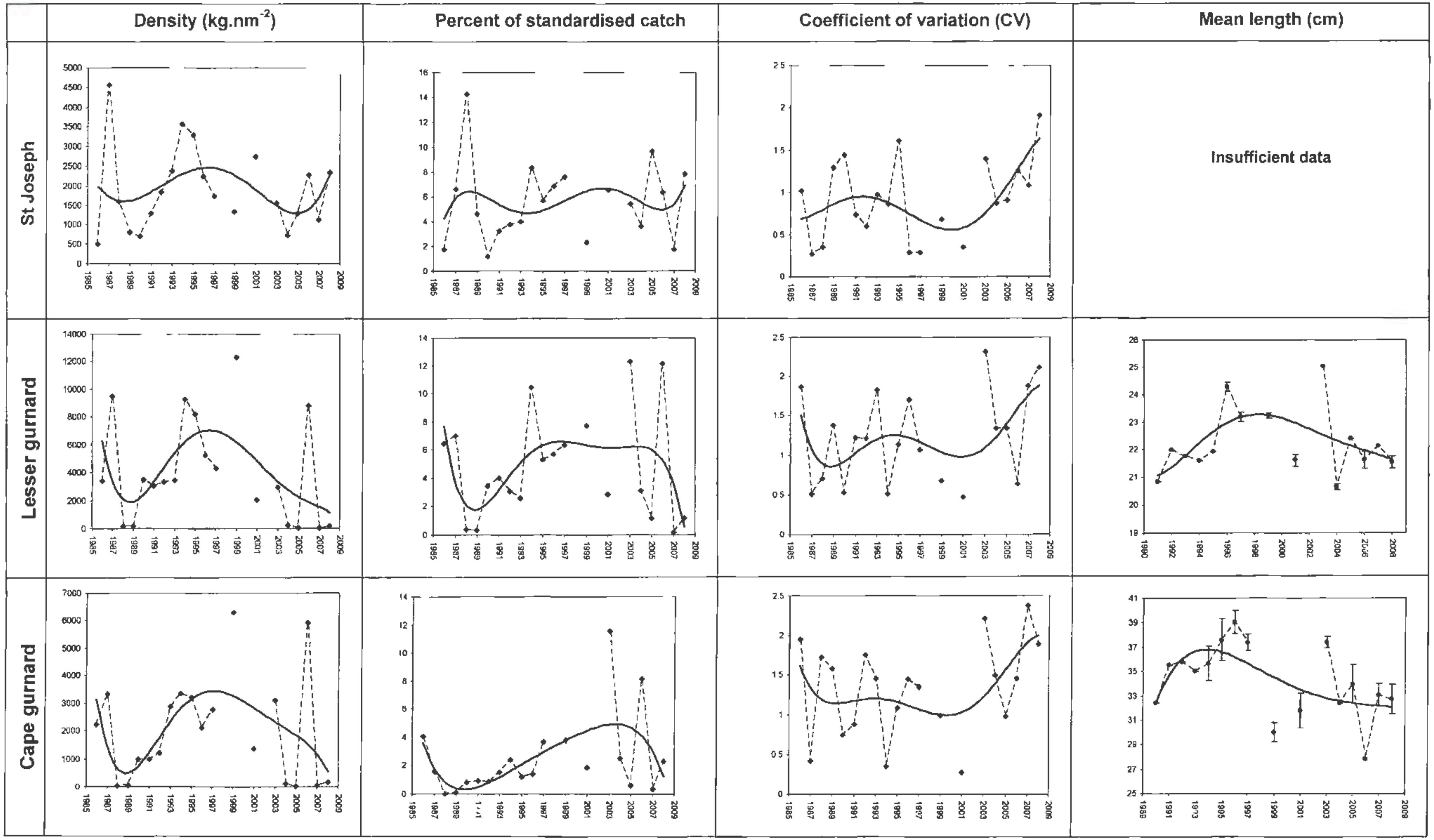


Figure 4.9. Annual trends in the density, percentage of catch composition and mean length (±95% confidence intervals) of St Joseph (top), lesser gurnard (middle) and Cape gurnard (bottom). General temporal trends are illustrated by 5th order polynomial regressions (solid line).

(j) Kob

Depth (p=0.042), area (p=0.006) and season (p<0.001) influenced the density of kob significantly (Table 4.5). Density was greater in deeper water, and greater in the central than the western sector of Algoa Bay, as well as greater during spring than autumn. Spatially the contribution to overall catch was greatest seaward of the Bird Island (Figure 4.12). No clear annual trends in kob density or the contribution to the annual catch was evident; however the CV, increased over time (Figure 4.13).

The size of kob was influenced significantly by depth (p<0.001), area (p<0.001) and season (p<0.001) (Table 4.6). Fish were larger in deeper water, and size increased from west to east with a greater proportion of fish above the 50% size at maturity in the eastern sector (Figure 4.12). The mean length of kob remained fairly low up until 2001 after which an increase in mean length occurred (Figure 4.13).

(k) Kingklip

Kingklip density was influenced significantly by depth (p<0.001), area (p<0.001) and substrate (p<0.001) (Table 4.5). Relative density was higher in deeper water, in the eastern than central or western regions and over muddy-sand than sand (Table 4.5; Figure 4.12). Few other areas in Algoa Bay contributed to kingklip catches. Annual trends indicate high variability in density between years with peaks in 1995, 2004 and 2008 (Figure 4.13).

Little annual data were available for kingklip size; however, depth (p<0.001), area (p<0.001) and season (p<0.001) were significant predictors of the mean size of fish with larger fish occurring in deeper water, in the eastern region and during spring than autumn (Table 4.6). The mean size of kingklip has remained relatively stable over time (Figure 4.13).

(l) Carpenter

The density of carpenter was influenced by depth (p=0.026) only, with greater catches landed from the deeper waters of Algoa Bay (Table 4.5; Figure 4.12). Density was greatest in deeper water between Riy Banks and Bird Island (Figure 4.12). Temporal trends indicate a significant peak in density in 1995 after which a progressive decline was apparent (Figure 4.13). The proportion of carpenter in the total catch has dropped considerably from the 1990s to present day (Figure 4.13).

The size of carpenter was influenced by depth (p<0.001) and season (p<0.001) (Table 4.6). Carpenter were larger in deeper water and during autumn than spring (Table 4.6; Figure 4.12). A large proportion of fish occurring in the west and shallow regions of Algoa Bay were below the size at 50% maturity (Figure 4.12) and a general decrease in the mean size was observed over time (Figure 4.13).

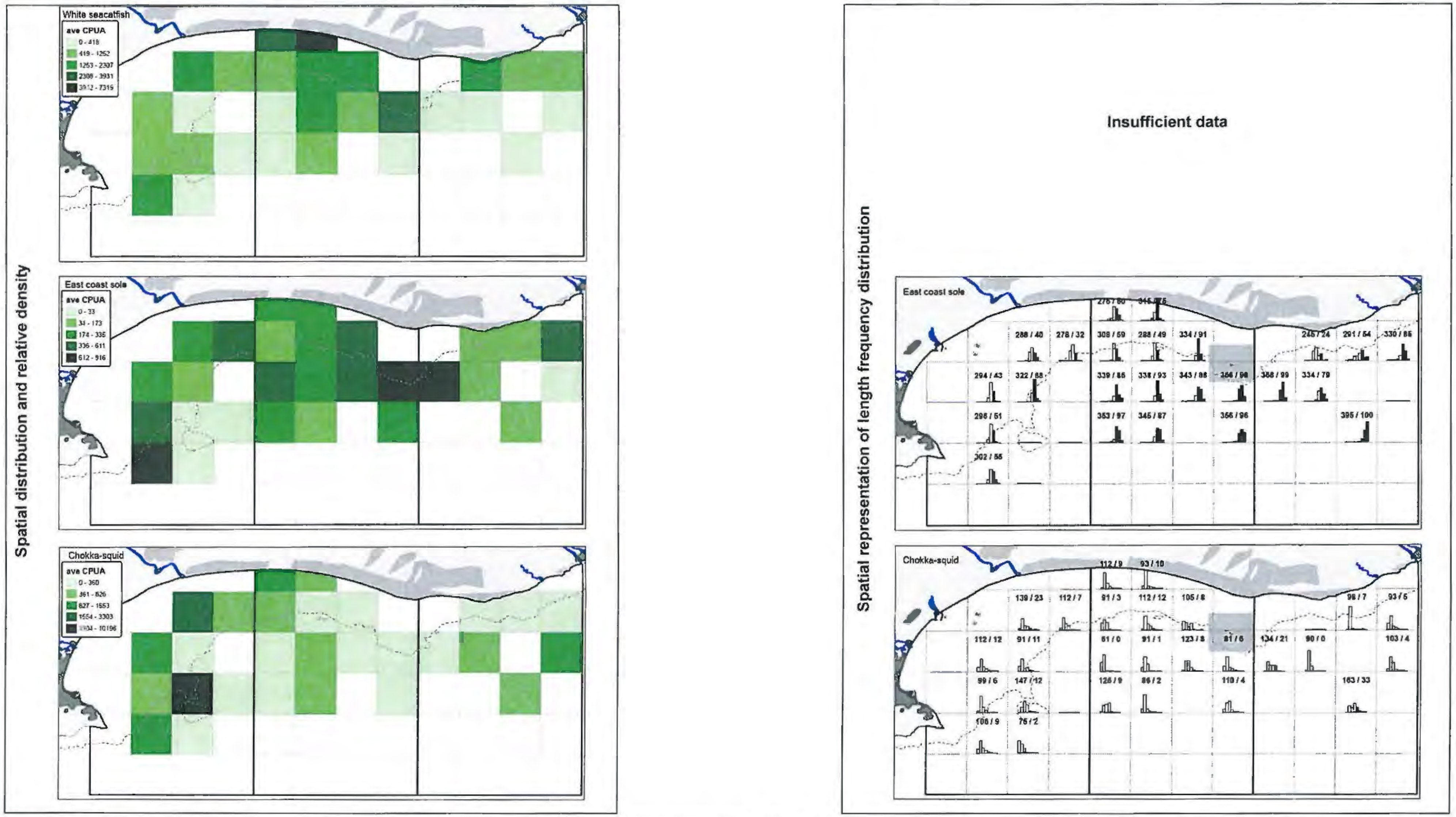


Figure 4.10. Spatial representation of the average annual density (left) and length frequency (right) of white seacatfish (top left), east coast sole (middle left) and chokka-squid (bottom left) per 5' grid cell in Algoa Bay. Numbers above length frequency histograms indicate mean length (TL mm) / % mature, white bars indicate size below 50% maturity and black bars above. Existing Addo Elephant National Park areas are indicated in darker grey. Dashed line is the 50m isobath.

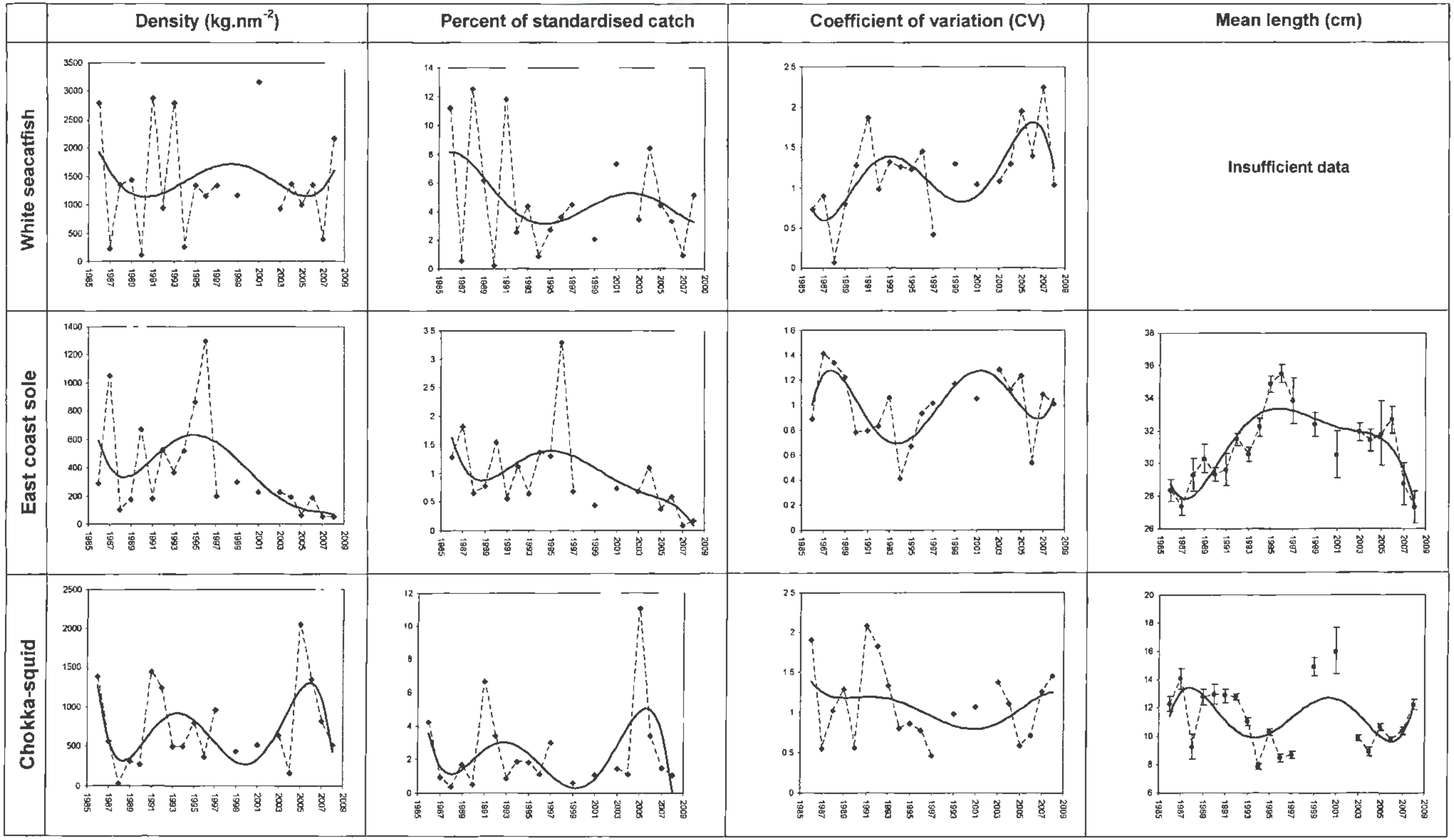


Figure 4.11. Annual trends in the density, percentage of catch composition and mean length (±95% confidence intervals) of white seacatfish (top), east coast sole (middle) and chokka-squid (bottom). General temporal trends are illustrated by 5[th] order polynomial regressions (solid line).

Summary of key findings

- Changes in trawl gear configuration did not affect diversity indices but influenced community structure significantly (Table 4.7)
- All of the key factors showed a significant effect on some aspect of biomass or mean length for the species assessed (Tables 4.7 and 4.8)
- Location and depth had the most pronounced effect on community structure and the relative abundance and mean length of individual species (Table 4.7 and 4.8)
- Inter-annual variability in relative abundance and mean length was high

Table 4.7. Summary of key factors influencing diversity indices and multivariate statistics.

Metric	Gear	Depth	Area	Substrate	Season
Species richness	×	×	×	×	✓
Shannon-Wiener Diversity	×	×	×	✓	×
Pielou's Evenness	×	×	×	✓	×
AvTD	×	×	×	×	×
VarTD	×	✓	×	×	✓
Multivariate	✓	✓	✓	✓	✓
Importance	17	33	0	50	50

Table 4.8. Summary of key factors influencing individual species abundance and size.

Species	Relative density					Mean length		
	Gear	Depth	Area	Subst.	Season	Depth	Area	Season
Horse mackerel	×	✓	✓	×	×	✓	✓	✓
Shallow-water hake	×	✓	✓	✓	×	✓	✓	✓
Panga	×	✓	✓	×	×	✓	✓	✓
St Joseph	✓	✓	✓	×	×	✓	✓	✓
Lesser gurnard	✓	✓	✓	×	×	✓	✓	✓
Cape gurnard	✓	×	✓	×	×	✓	✓	✓
White seacatfish	×	✓	✓	✓	×	✓	✓	✓
East coast sole	✓	×	×	×	×	✓	✓	✓
Chokka-squid	×	✓	✓	✓	×	✓	✓	✓
Kob	×	✓	✓	×	✓	✓	✓	✓
Kingklip	×	✓	✓	✓	×	✓	✓	✓
Carpenter	×	✓	×	×	×	✓	×	✓
Importance	33	83	83	33	8	100	92	100

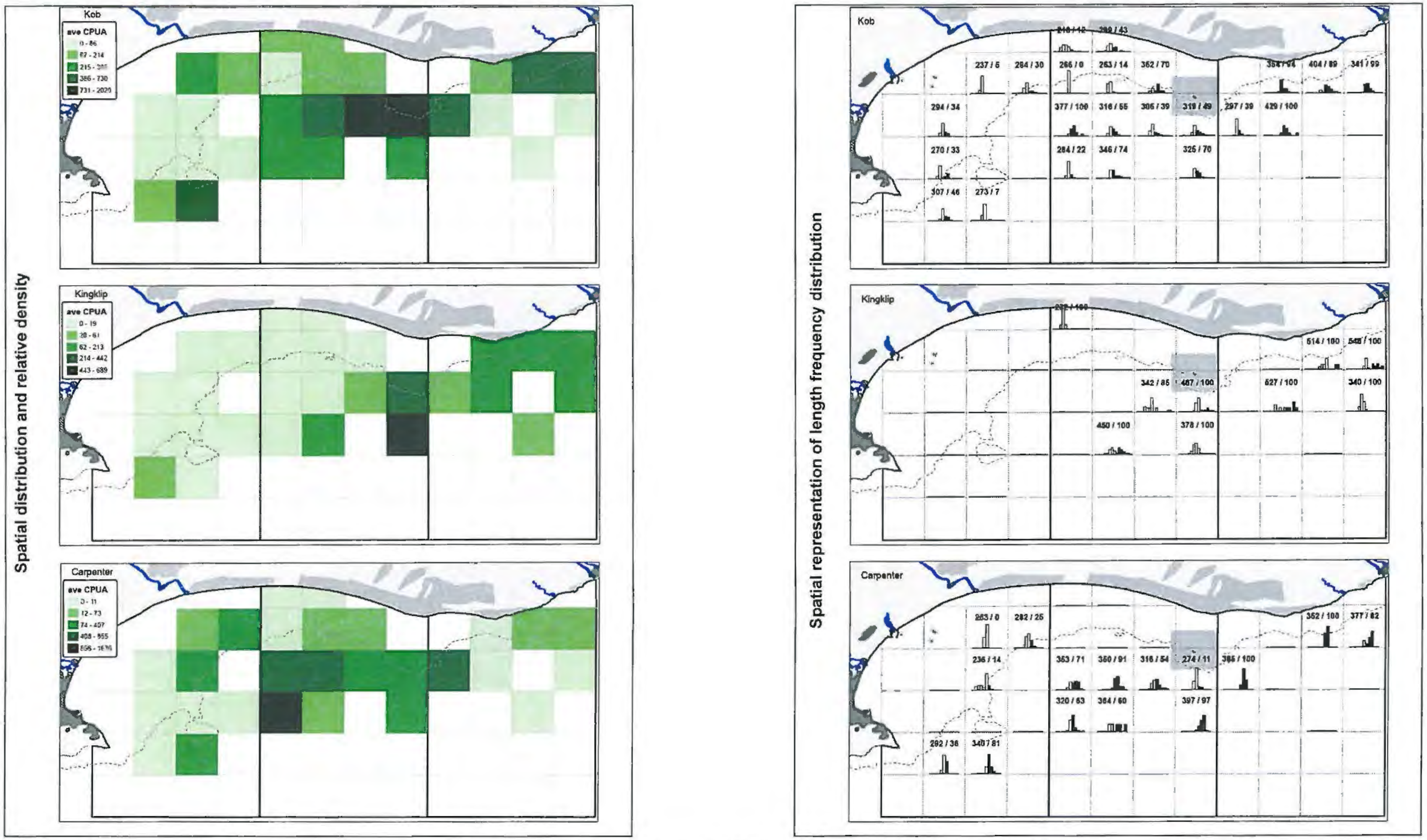


Figure 4.12. Spatial representation of the average annual density (left) and length frequency (right) of kob (top left), kingklip (middle left) and carpenter (bottom left) per 5' grid cell in Algoa Bay. Numbers above length frequency histograms indicate mean length (TL mm) / % mature, white bars indicate size below 50% maturity and black bars above. Existing Addo Elephant National Park areas are indicated in darker grey. Dashed line is the 50m isobath.

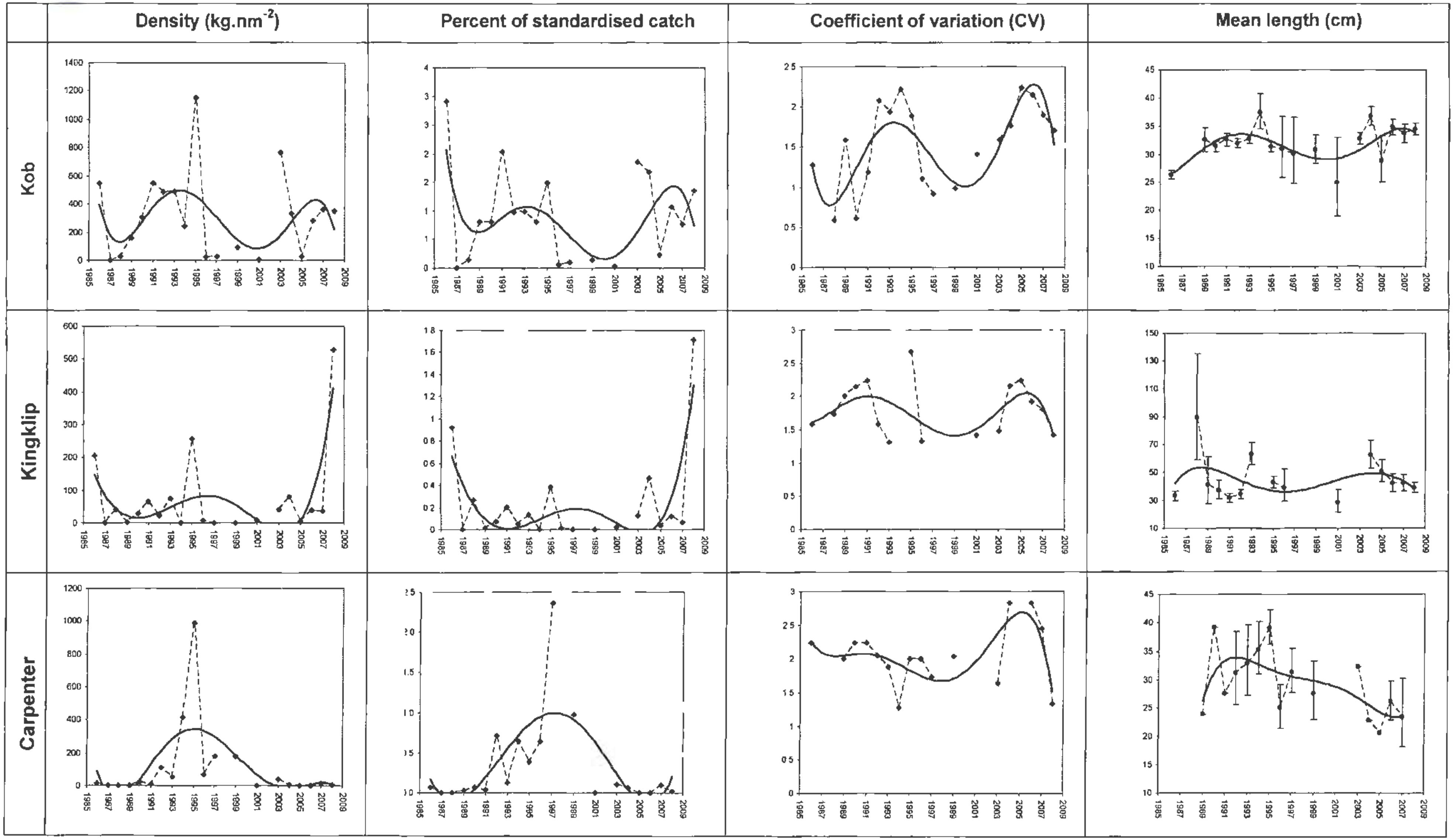


Figure 4.13. Annual trends in the density, percentage of catch composition and mean length (±95% confidence intervals) of kob (top), kingklip (middle) and carpenter (bottom). General temporal trends are illustrated by 5th order polynomial regressions (solid line).

4.4 Discussion

This chapter is based on an analysis of fisheries independent research trawl data from biomass surveys which are conducted biannually along the south coast of South Africa. Despite being independent of the biases common with fisheries dependent data, the survey design has limitations in the context of this study due to the localised nature of the study area considered and the broad national objectives on which the survey design is based. The biomass surveys are designed to sample the south-east coast of South Africa in a representative manner with the selection of sampling localities based on a pseudo-random design across depth strata. Hence the number of sampling stations selected per depth stratum is proportional to the area of each stratum (Yemane *et al.* 2010). Algoa Bay only comprises a small portion of the two shallowest depth strata (0-50m and 51-100m) within the survey area resulting in inconsistencies in the number of sites sampled within Algoa Bay each season and year. Furthermore, substrate composition is not considered in the selection of sampling locations which would be beneficial when conducting surveys on localised scales. The survey design is therefore not optimal for localised assessments such as this study on Algoa Bay. In particular there is a high level of spatial correlation between the factors of interest to the study (depth; area; substrate) and the observed effects could be as a result of one of these factors, or a combination of many. Nonetheless, despite these limitations these surveys provide invaluable information on the demersal ichthyofaunal composition available to the trawl gear which would otherwise be too costly to obtain through a dedicated trawl survey designed specifically to assess and monitor communities within Algoa Bay. The limitations of the data used in this analysis are therefore recognised and acknowledged; however, the paucity of information available on offshore demersal ichthyofauna, particularly for soft benthic habitats where trawl surveys are the only practical means for assessing demersal communities, necessitates the use of such data for local level planning initiatives in South Africa.

Although dedicated inshore trawling surveys have been conducted in Algoa Bay in the past (Buxton *et al.* 1984; Wallace *et al.* 1984a; Beckley 1984a; Wallace *et al.* 1984b), they have largely been restricted to the nearshore (<50m) with the objective of assessing habitat use by estuarine associated species, and were not designed for spatial assessment of demersal communities across Algoa Bay, particularly in deeper waters. The objective of this component of the study was therefore to utilise available sources of data for describing the composition of demersal ichthyofaunal communities sampled by the trawl gear over the trawlable areas of Algoa Bay, as well as identifying key explanatory factors which influence diversity, community structure and the distribution of economically important species. Identifying and understanding the importance and role of different factors structuring these communities will allow for improved planning and management of resources locally within Algoa Bay in the future.

Research trawl data indicated diverse ichthyofaunal assemblages in Algoa Bay, but species representation was lower than on the Agulhas Bank, with 141 and 219 species recorded respectively (Japp *et al.* 1994). This may in part be attributable to the small size of the Algoa Bay study area which only includes shallow depth strata to the 100m isobath, in comparison to that of the large spatial extent of the south coast which includes depths up to 500m. It may also be due to other factors including oceanographic conditions and primary and secondary productivity amongst others. Similarly to the Agulhas Bank where 16 species accounted for 71% of the biomass (Japp *et al.* 1994), the trawled ichthyofaunal communities in Algoa Bay were dominated by few species with ten species accounting for 75% of the biomass. Despite horse mackerel comprising a lower proportion of the biomass within Algoa Bay (18%) than on the Agulhas Bank (36%), it was the dominant species in both areas. Hake constituted the third highest proportion of the biomass in Algoa Bay (9%) accounting for a considerably lower proportion of the overall biomass than on the Agulhas Bank (19%). This can be attributed to its preferences for deeper water (Fairweather 2001) and the shallow depths sampled in this study. The proportion of panga biomass in Algoa Bay and on the Agulhas Bank was, however, similar (7 and 8% respectively) (Japp *et al.* 1994). Only 18% of the species were recorded in at least half of the trawl stations indicating the low prevalence of most species. Nonetheless, several of the less common species recorded are commercially and biologically important, supporting several commercial and recreational fisheries in Algoa Bay as well as playing important roles in the ecosystem. Many species recorded in Algoa Bay during the trawl surveys are heavily exploited throughout their distributional range in South African waters and altemative management initiatives, including ongoing output controls for some species, and spatial and temporal restrictions have been advocated as a viable means to regulate fishing pressure and reduce mortality.

4.4.1 Key factors influencing demersal community structure

(a) Spatial patterns

Commonly used diversity indices have been shown to be sensitive to spatial effects along the south coast of South Africa (Yemane *et al.* 2010) as well as other regions of the world (McClatchie *et al.* 1997; Catalán *et al.* 2006). However, no spatial effect was observed in the current study. The lack of importance of the area effect is likely due to the small spatial scale over which the study was conducted and the occurrence of similar species throughout Algoa Bay, albeit at varying densities and size classes. Nonetheless spatial differences in species distributions and density were insufficient for distinction of different levels of diversity and communities based on the univariate indices alone.

Multivariate analysis, however, indicated significant differences between the areas considered in Algoa Bay, with greatest differences between the two most distant sites, suggesting a gradient of change in trawled demersal ichthyofauna from west to east across Algoa Bay. Similar gradual spatial changes in trawled community assemblages have been documented to occur along the west coast of South Africa (Atkinson *et al.* 2011b). In the context of the current study, and the small spatial scale considered, the bay environment and the differing habitats it creates is most likely to be a significant contributing factor to spatial differences in the community structure observed. Ichthyofaunal communities may be influenced by the level of exposure to currents and wave action with certain species or life history stages preferring the sheltered bight of embayments in contrast to exposed coastal areas. The western bight of Algoa Bay is protected from the predominant south-westerly swell and is therefore more sheltered than the central and eastern sectors. Such protected coastal embayments serve as important nursery and spawning areas for numerous marine species (Ansari *et al.* 1995). Species preferences for sheltered habitats, particularly during periods of the year associated with spawning, recruitment or migration may therefore account for the spatial differences in community patterns observed in the current study and highlights the importance of fine scale spatial assessment for local planning and conservation initiatives. An investigation of the gut content of the dominant species, and differences between life history stages within each species, would contribute to improved understanding of the spatial use of Algoa Bay with regards to feeding and foraging behaviour and is an important aspect for future research in the area.

Density of St Joseph, white seacatfish, the lesser and Cape gurnards, and chokka-squid was greater in the western region of Algoa Bay, indicating preferences for a more sheltered environment. Contrarily the density of horse mackerel, shallow-water hake, panga, kob and kingklip was greater in either the central or eastern sectors which are more exposed to the prevalent oceanographic conditions. Although the western sector of Algoa Bay has a larger proportion of shallow water habitat which could have contributed to the observed spatial effects on community structure, a two-way ANOSIM revealed that both spatial aspects and depth structured community assemblages in Algoa Bay. This supports the possibility of a spatial community gradient of trawled communities based on the level of protection/exposure across Algoa Bay.

Furthermore with the exception of shallow-water hake, carpenter and chokka-squid all species were of a smaller mean length in the western than the central or eastern regions, and a large proportion of the population was below the size at 50% maturity. This suggests that the shallow protected western areas of Algoa Bay potentially play an important role as a nursery habitat for the juveniles of numerous demersal species.

(b) Temporal patterns

(i) Season

Species richness and VarTD were influenced significantly by short-term seasonal changes in the trawled demersal communities, with both greater over spring, suggesting that an inshore migration of less common and typically deepwater species may occur during this period of the year. However, multivariate analysis indicated that although the trawled demersal ichthyofaunal community structure was influenced significantly by seasonal effects, the magnitude of the effect was weak, and less than that for all other factors investigated. Four species were dominant (horse mackerel, red tjor-tjor, shallow-water hake and panga) during both seasons accounting for similar proportions of the total catch in Algoa Bay, representing 54% and 48% during spring and autumn, respectively. Smale *et al.* (1993) noted that seasonal differences in community structure were difficult to determine using trawl survey data from the Agulhas Bank. The authors did, however, suggest a possible inshore movement of fish in autumn/winter which they attributed to temperature fluctuations. This is contrary to the findings in the current study with greater diversity observed in spring. Many marine species undertake seasonal migrations (Smale 1985; Griffiths 1996b; Heemstra and Heemstra 2004) and juveniles have been shown to recruit into inshore nursery areas at certain times of the year in India (Ansari *et al.* 1995). Such movement patterns may potentially affect the relative density of species differently, thereby leading to detection of seasonal differences in community structure. Despite these factors, short-term temporal changes in community structure were not readily evident in Algoa Bay. Limited seasonal differences in the community structure of trawled demersal ichthyofauna has also been reported along the west coast of South Africa (Roel 1987; Atkinson *et al.* 2011b) as well as in the North Aegean Sea (Labropoulou and Papaconstantinou 2000).

Kob was the only species for which the relative density differed significantly by season, with higher densities in Algoa Bay during spring. It has been suggested that kob spawn in inshore waters in spring (Smale 1985; Griffiths 1997c) and the higher densities observed over this period may therefore be linked to an inshore movement of spawning adults. However, mean size of kob was lower during the spring survey period suggesting that recruitment of juveniles into inshore embayments may account for the greater catch over this period. The mean length of panga, lesser and Cape gurnard, east coast sole and carpenter was also lower during spring months, supporting the possible evidence for recruitment of juveniles into Algoa Bay over this period.

(ii) Long-term temporal trends

Diversity indices showed variable long-term temporal trends. Increasing diversity and declining dominance has been observed in demersal ichthyofaunal (Yemane *et al.* 2010) and linefish (Yemane *et al.* 2004) communities in South Africa, and demersal communities elsewhere in tropical ecosystems (Bianchi *et al.* 2000) and the North Sea (Greenstreet and Hall 1996). These patterns were attributed to the effects of differing levels of exploitation on individual species resulting in declining dominance of species which were previously abundant but heavily targeted by fisheries.

Despite changes in trawl gear configuration (discussed below) multivariate analysis did not differentiate distinct temporal groupings in trawled demersal communities with only a few, non sequential survey years falling outside of a 71% level of similarity. This is despite the high inter-annual variability observed in individual species density, and the proportional contribution to the total catch. Relative stability in trawled community structure in the long-term is further supported by high concordance values for the rank order of species abundance for seasonal surveys (75% autumn; 76% spring).

(c) Depth

Previous studies have shown that depth is one of the major factors influencing ichthyofaunal diversity (Tolimieri and Anderson 2010; Yemane *et al.* 2010). Along the south coast of South Africa diversity typically decreased with increasing depth from the shoreline (Yemane *et al.* 2010). Within the current study, however, no diversity indices were influenced significantly across the limited depth range in Algoa Bay.

Multivariate analysis revealed that depth had the greatest influence on the trawled demersal fish community structure. Similarly depth has been shown to be one of the main factors influencing the trawled demersal ichthyofaunal community assemblages and species distributions along the south-east (Badenhorst and Smale 1991; Smale and Badenhorst 1991; Smale *et al.* 1993) and west coasts of South Africa (Roel 1987; Atkinson *et al.* 2011b) as well as other regions of the world (Flemming 1981; Rainer 1984; Blaber *et al.* 1994; Ungaro *et al.* 1999; Labropoulou and Papaconstantinou 2000; Martin *et al.* 2010; Tolimieri and Anderson 2010). The depth range assessed in this study was narrow (100m) compared to previous assessments in South Africa which have investigated the influence of depth along the south and west coasts to the 500m isobath (Smale *et al.* 1993; Japp *et al.* 1994; Atkinson *et al.* 2011b). Despite the narrow depth range two, demersal community groups were discernable from the trawl data within Algoa Bay based on a division at the 50m isobath. The shallow water community (<50m) was characterised by similarities in the presence and abundances of red tjor-tjor, St Joseph, eagle ray and white seacatfish, which are commonly encountered in nearshore coastal areas and embayments (Smale and Badenhorst 1991; Smale *et al.* 1993; Freer and Griffiths 1993b; Heemstra and Heemstra 2004). Although horse mackerel was more abundant offshore, where it dominated the catch, it also accounted for the third greatest biomass in the shallow regions of Algoa Bay illustrating its dominance throughout the bay. Three commercially important species, horse mackerel, shallow-water hake and panga accounted for 55% of the catch in the deeper zone (51-100m), dominating the demersal communities.

Overall the relative density of ten of the 12 commercially important species was influenced by depth. Seven of these species, horse mackerel, shallow-water hake, panga, kob, carpenter, lesser gurnard and kingklip, showed preferences for deeper water habitats. All of these species were also larger in deeper water indicating a possible offshore movement with growth, or recruitment of juveniles into the shallower waters. St Joseph and white seacatfish density was greater in the shallower waters of the bay yet mean size was smaller, further indicating inshore recruitment of juveniles of these species into the bay and the potential importance of the shallow water sheltered bay environments as a nursery habitat. The distribution and availability of food for these species may also differ with depth, and differences in abundances of the varying ichthyofaunal life history stages may be influenced by dietary preferences related to the availability of the preferred prey items. Further investigation into the gut content of the different life history stages occurring across the depth ranges of Algoa Bay would provide valuable insight into the potential influence of foraging and feeding behaviour on the observed patterns and is an important aspect for future research.

The distribution of horse mackerel and shallow-water hake has previously been shown to be strongly influenced by depth with higher densities (Badenhorst and Smale 1991) and larger fish (Kerstan and Leslie 1994; Burmeister 2001) found in deeper waters along the south coast. This was also found in the current study with few horse mackerel (12%) above the size at 50% maturity in shallow water, and similarly few (12%) shallow-water hake were sexually mature within Algoa Bay. Badenhorst and Smale (1991) proposed that the inshore (<50m) habitats along the south coast are used as nursery areas by both species and it has been suggested that juvenile horse mackerel recruit into bays along the south coast (Barange *et al.* 1998). These suggestions are supported by the findings of the current study as smaller fish of both species were more abundant in the shallow waters of Algoa Bay.

Density of the linefish species panga, kob and carpenter was also greater in the deeper waters of Algoa Bay. Higher densities of juvenile kob have previously been reported within embayments (Smale and Badenhorst 1991) suggesting the inshore recruitment of juveniles which utilise shallow protected nursery grounds and subsequently move offshore to deeper waters with growth and increasing age (Smale 1984; Beckley 1984a; Griffiths 1997c). Spawning has been suggested to occur in depths less than 50m and over the spring months (Smale 1985; Griffiths 1997c) possibly accounting for higher abundances observed within Algoa Bay over this period. Algoa Bay has also been proposed as a nursery area for carpenter due to the high abundances of juvenile fish, which move offshore and westwards with age and maturity (Griffiths and Wilke 2002; Brouwer *et al.* 2003; Brouwer and Griffiths 2005b). The density of St Joseph and white seacatfish was greater in shallow water largely due to high abundances of juvenile fish in these areas. These trends indicate the potential importance of the shallow Algoa Bay habitats as nursery areas for several species.

(d) Change in trawl gear configuration

Surprisingly the change in trawl gear configuration did not change the communities sampled by the gear sufficiently to influence any of the diversity indices significantly. However, multivariate analysis indicated significant differences in the structure of the community sampled by the different gears and single species analyses indicated reduced catches of St Joseph, lesser and Cape gurnards as well as east coast sole. The change in footrope configuration from a heavy chain footrope in the old gear to rubber discs in the new gear has reduced the sampling efficiency for flatfish and batoids, and a reduction in the door spread leads to reduced herding of shoaling species, while a greater volume of water is sampled due to increased vertical height of the net (Atkinson *et al.* 2011b). As a result of these modifications, changes in structure of the sampled community would therefore be anticipated as observed in this study, and flatfish and shoaling species are likely to contribute the most to the differences. However, numerous species made small contributions to the observed differences, with horse mackerel, red tjor-tjor and panga having the most pronounced effect. Further evidence for the influence of gear type on the sampled community structure is illustrated through a reduction in long-term community stability when both old and new gear types were included in the analyses. Community stability in autumn decreased by 5% while in spring it decreased by 3%, confirming the effect of gear type on sampling efficiency of the demersal communities. Future surveys should take into account the differing effects of the trawl gear, which should be alternated between surveys in order to improve the calibration between net types.

(e) Substrate type

Community structure and species distributions are strongly influenced by substrate type (Le Clus *et al.* 1994; Macpherson 1994; Le Clus *et al.* 1996; Fairweather 2001; Sampson 2002). However, in the current study substrate type had a significant effect on the species richness only. Multivariate statistics also revealed that ichthyofaunal communities within Algoa Bay were influenced by substrate composition. The lack of a strong (low Global R) distinction between substrate types is likely due to the transition from one community to another across substrate borders and the course resolution of the substrate data in relation to the spatial scale of the assessment. No *in situ* substrate data were collected during the trawl surveys and historical substrate maps which were developed from geological surveys conducted in the 1960s and 1970 were used to infer substrate type at each trawl station. As a result inaccuracies may arise due to earlier spatial interpolation of the sediment types. In addition, trawl stations may cover multiple substrate types further confounding the interpretation of community structure based on substrate. Nevertheless, this study provides some evidence for community assemblage differences as a result of substratum characteristics on a local scale and has important implications for future management of the demersal ichthyofauna.

4.4.2 Conclusions

This and earlier studies (Badenhorst and Smale 1991; Smale and Badenhorst 1991; Brouwer and Griffiths 2005b) have demonstrated the importance and use of Algoa Bay by both adults and juveniles of several demersal species. Certain species showed clear trends of increasing size and density with increasing depth and from west to east across Algoa Bay. Juveniles of several species which are conventionally considered deep water species, were often more abundant in the sheltered, shallow western section of Algoa Bay, suggesting that this is an important nursery area. Depth was also shown to be an important factor influencing the distribution of species and the structure of communities. These are important findings from a spatial management perspective. In addition, this analysis has provided distributional data for key migratory linefish species which are heavily targeted by the recreational and commercial skiboat sectors, and for which stocks are considered depleted. This chapter has therefore provided valuable spatial data for key conservation features which need to be taken into consideration, and can contribute to development of a spatial management plan for conservation in Algoa Bay (Table 4.9).

It is also evident from this study that there is high spatial and temporal variability in demersal ichthyofaunal communities sampled by the trawl gear in Algoa Bay, and that spatial and temporal trends differ between species. These trends are driven by numerous factors, with depth being one of the key factors influencing the community structure and distribution of species. This has important implications in the design of a sampling strategy for long-term monitoring of these communities. The data presented in this chapter provides a baseline against which future monitoring can be evaluated. The results from this chapter contribute to the overall planning and monitoring objectives of the study as outlined in Table 4.9 below.

Table 4.9. Contribution of chapter results to spatial planning and monitoring in Algoa Bay.

Chapter 7: Systematic conservation planning	Chapter 8: Monitoring and evaluation
1. Spatial layer of inshore shallow water demersal nursery area 2. Spatial layer of silver kob distribution for use as a surrogate species in spatial planning 3. Spatial layer of geelbek distribution for use as a surrogate species in spatial planning	1. Long-term temporal data for future comparative assessments, including variability and determination of required sampling effort 2. Identification of key factors influencing spatial and temporal distribution of demersal ichthyofauna required for stratification of future monitoring effort 3. Identification of dominant species for use as a potential indicators

CHAPTER 5

COASTAL RECREATIONAL LINEFISHERIES OF ALGOA BAY

5.1 Introduction

5.1.1 Global recreational fisheries

Recreational angling is a popular outdoor leisure activity in many developed countries and the social and economic value of the sector is now widely recognised (Post *et al.* 2002). In some cases the contribution of recreational fisheries to local or regional economies outweighs that of commercial fisheries (Post *et al.* 2002). Although growth in the recreational sector in recent years has been considerable in many countries, it is generally unquantifiable due to an absence of historical data for comparison. Surveys in the United States of America have indicated a 20% increase in the number of coastal recreational fishing trips over a four-year period (Sutinen and Johnston 2003) suggesting that the potential worldwide growth of the recreational sector has been, and is likely to be considerable in the future.

Recreational fisheries are characterised by high levels of user participation, and although individual anglers may not have measurable effects on the resources they target, the cumulative impact of the sector is of concern for the sustainable utilisation of the target resources. Annual yields from recreational fisheries are considerable in many countries, being greater than that of commercial fishery sectors in some instances (Schroeder and Love 2002; Atkinson and Clark 2005). It has also been shown that recreational sectors can have measurable effects on marine resources (Buxton and Clarke 1991; Bennett 1993; Schroeder and Love 2002; Cooke and Cowx 2004). Cooke and Cowx (2004) estimated the annual recreational catch to be 47 billion fish worldwide with a retention rate of approximately one-third, highlighting the potential contribution recreational fisheries may have on the declining stock status of many fishery species.

Recreational anglers focus their effort in areas likely to yield the greatest returns, which coincide with critical and sensitive habitats where fish aggregations occur during spawning, migration of feeding events (Jackson *et al.* 2001; Cooke and Cowx 2004) making them particularly susceptible to fishing pressure. Although regulations may limit the number and size of fish retained, non-compliance (Sullivan 2002; Wilberg 2009) and post-release mortality (Muoneke and Childress 1994; Alós 2009; Henderson 2009) are high in the recreational sector. Additionally recreational anglers target species which exhibit life-history characteristics such as slow growth, late maturity, longevity and sex change, making them particularly vulnerable to fishing pressure and overexploitation (Buxton 1992). Fisheries regulations in the commercial sector often prohibit catch of such species, or have regulations which are highly restrictive and usually well enforced. Effort in the recreational sector is, however, unlimited, and the bag and size restrictions have often failed to limit the catch of individual anglers (Bennett 1993; Attwood and Bennett 1995). For species perceived to be vulnerable, recreational angling has shown to contribute to up to 23% of the landings in the United States of America (Coleman *et al.* 2004).

No direct means for limiting the number of participants in the recreational sector are available. However, seasonal and spatial closures provide a meaningful way to reduce fishing effort (Pereira and Hansen 2003), particularly when imposed over key aggregation periods (spawning aggregations; migration periods etc.) or aggregation areas (nursery, feeding, spawning areas etc.). Baseline spatial and temporal information of the fishery characteristics (catch and effort) is generally lacking, is time consuming and costly to collect, yet is essential if recreational fisheries activities are to be taken into consideration in the future planning and design of no-take MPAs. This allows recreational fisheries activities to be accommodated in the planning process, yet effective management measures developed to reduce recreational pressure in critical areas in which the target resources are particularly vulnerable. In addition baseline information is required as a benchmark against which ongoing research and monitoring studies can be compared to determine whether the implemented management interventions achieve the desired long-term objectives.

5.1.2 The South African recreational linefishery

The South African marine linefishery is a multi-species fishery with commercial, recreational and subsistence sectors. The recreational sector has both shore (estuarine and coastal rock and surf angling) and boat-based (estuarine and offshore) components which target a wide diversity of species (Brouwer *et al.* 1997; Mann *et al.* 1997; Sauer *et al.* 1997). In terms of the targeted species and habitats there is overlap between shore and boat-based components, as well as with the commercial and subsistence sectors (Brouwer and Buxton 2002; King 2005).

Recreational angling is one of the most popular sport and outdoor activities in South Africa with an estimated 2.5 million participants (all components) and direct economic impact of ZAR15.9 billion in 2007 (Liebold and van Zyl 2008). The increasing annual participation in the recreational fishery (Clarke and Buxton 1989; Coetzee *et al.* 1989; Brouwer *et al.* 1997) and the concomitant decline in the catches of many targeted species (van der Elst and de Freitas 1988; Hecht and Tilney 1989; Brouwer *et al.* 1997; Brouwer 1997; Brouwer and Buxton 2002) has been documented in several studies. Recent participation in marine fisheries has been estimated at 850 000 recreational shore anglers and over 100 000 recreational skiboat anglers (Liebold and van Zyl 2008). First efforts to manage the recreational sector were implemented in 1985 with the introduction of seasonal restrictions, and daily bag and minimum size restrictions (Bennett *et al.* 1994). These regulations remain in place and have been subject to periodic review. A formal national permit system was introduced in 1999 (Griffiths and Lamberth 2002).

Past assessments of the recreational fishery in South Africa have included estimates of total catch and effort (Brouwer *et al.* 1997; Brouwer 1997; Brouwer and Buxton 2002); analysis of catch returns and enforcement and monitoring patrol data (Pradervand *et al.* 2003; Pradervand and Hiseman 2008; Pradervand and van der Elst 2008) as well as angling competition data (Coetzee *et al.* 1989; Pradervand *et al.* 2007). The most comprehensive of these studies was the National Survey initiated in 1994 to evaluate the catches, socio-economic status and perceived attitudes of the participants in the recreational and commercial linefishery sectors (Cockcroft *et al.* 1999). Results from this study indicated that recreational anglers accounted for 96% of the participation in the linefishery.

Recreational rock and surf shore angling accounted for 92% of participation, with the skiboat and spearfishery sectors accounting for 3% and 1% of the effort respectively (Cockcroft *et al.* 1999). Although dominating the participation, the recreational shore fishery landed only 11% of the catch, while the skiboat sector accounted for 21%, and the spearfishery for 1%. The remaining 67% of the catch was attributed to the commercial linefish sector. Brouwer *et al.* (1997) estimated the annual recreational shore fishery catch at approximately 4.5 million fish or 3 000 tons.

The ratio of commercial to recreational catch is likely to have changed significantly in recent years due to the State of Emergency declared in the linefishery in 2000, and the resultant reductions in commercial fishing effort (DEAT 2000). However, participation in the recreational fishery has remained unlimited although catch regulations have been amended based on the results of recent scientific studies. A major challenge in the linefishery was identified as the need for improved data gathering from all fishing sectors (Cockcroft *et al.* 1999). A national monitoring system was therefore designed and implemented; however, it focused on the commercial sector with evaluation of the recreational sectors largely being limited to *ad hoc* research projects.

Despite the potential negative effects of the recreational fishing sector on the marine environment, the socio-economic value and contribution to local economies is considerable and needs to be taken into consideration in future management. An improved understanding of the fishery dynamics and future growth is therefore required if management is to be successful in maintaining a sustainable fishery which continues to contribute to the local economy.

The overall aim of this chapter was to assess recreational fisheries within Algoa Bay to determine key factors influencing spatial and temporal trends. This baseline information is required for and will contribute to marine spatial planning in Algoa Bay (Chapter 7) and the development of monitoring protocols for long-term evaluation (Chapter 8). The specific main objectives of this chapter were:

1. to determine the factors influencing the spatial and temporal dynamics of effort and catch per unit effort (CPUE) for the recreational shore and skiboat fisheries in Algoa Bay;
2. to determine the catch composition and estimate annual effort and harvest of the recreational shore and skiboat sectors; and
3. to develop spatial indices of recreational fishing effort to aid future spatial planning and monitoring in Algoa Bay.

5.2 Methods

5.2.1 Study area

With the exception of the Port Elizabeth and Coega harbours where public access is controlled, shore fishing is permitted along the entire coastline of Algoa Bay provided the anglers have the required fishing licenses in terms of the Marine Living Resources Act (Act 18 of 1998). Although large stretches of the coastline fall within the Woody Cape and Colchester sections of the Addo Elephant National Park, and are therefore managed by SANParks, fishing is permitted along the coast provided anglers have an access permit to enter a parks area.

Boat access to the offshore marine environment occurs via three formally recognised launch sites, namely the Port Elizabeth harbour, where recreational vessels launch from the Port Elizabeth Deep Sea Angling Club (PEDSAC), and the Boknes and Kenton beach launch sites (Figure 5.1). Although the Kings Beach and Hobie Beach launch sites in Port Elizabeth are legally permitted sites, they are mainly used to launch jetskis and sailing craft, and few recreational fishing vessels are launched at these sites (Beach Manager's Office *pers. comm.*). Some recreational fishing vessels launch through the Swartkops and Sundays estuaries, but these are not formally permitted launch sites for sea access and the levels of utilisation are low. Skiboat fishing occurs throughout Algoa Bay, only being prohibited in the Bird Island MPA, situated approximately 10km off the Woody Cape headland, and with 500m of the St Croix, Brenton and Jahleel islands (Figure 5.1).

5.2.2 Survey design

The assessment of recreational fisheries within Algoa Bay was undertaken using a combination of roving creel, access point and aerial surveys.

(a) Roving creel surveys

Roving creel surveys are commonly employed for obtaining information on the catch and effort in dispersed fisheries (Pollock *et al.* 1994), and have been widely used in the past to assess recreational shore fisheries in South Africa (Brouwer *et al.* 1997; Brouwer 1997; Mann *et al.* 2003; Ellender *et al.* 2010) and elsewhere (Lockwood *et al.* 1999; Smallwood *et al.* 2006; Rangel and Erzini 2007). A detailed survey designed to obtain catch, effort and demographic information for the recreational shore fishery was undertaken between Coega and Boknes estuaries by means of roving creel surveys (Figure 5.1). This stretch of coastline forms the core planning area for the AENP MPA and baseline information on recreational fishing activity was required by SANParks for the design of no-take areas (See Appendix 1) and evaluation of the potential losses to the recreational shore fishery. Detailed monthly site-based catch and effort surveys were therefore conducted along this stretch of coastline with aerial surveys used to estimate and assess the spatial distribution of recreational shore and skiboat fishing effort in the broader study area from Cape Recife to Bushmans River Mouth (Figure 5.1).

For roving creel surveys the coastline was divided into five zones based on the distribution of access points and distances which could be traversed within a four-hour period using a four-wheel drive vehicle or by foot where vehicular access was restricted (Figure 5.1). With the exception of the Hougham Park (HP) survey zone, which was temporarily closed to public access during the study period due to construction activity, four monthly creel surveys were conducted within each zone between April 2006 and March 2009 and were stratified by day type, survey period and tidal phase. Day type consisted of work and non-work days (weekends and public holidays) and sampling effort was split equally between the two. Each sampling day was divided into three survey periods, morning (sunrise to 10:30am), midday (10:30am-14:30pm) and afternoon (14:30 to sunset). Starting time within each survey period, and travel direction within each survey zone[3] were randomly selected. No sampling was conducted at night; however, an indication of night fishing effort was obtained from anglers encountered in the late afternoon who said that they would be fishing late into the night, or alternatively from anglers encountered during the early morning, who indicated that they had been fishing during the night. In addition any signs of overnight fishing were recorded.

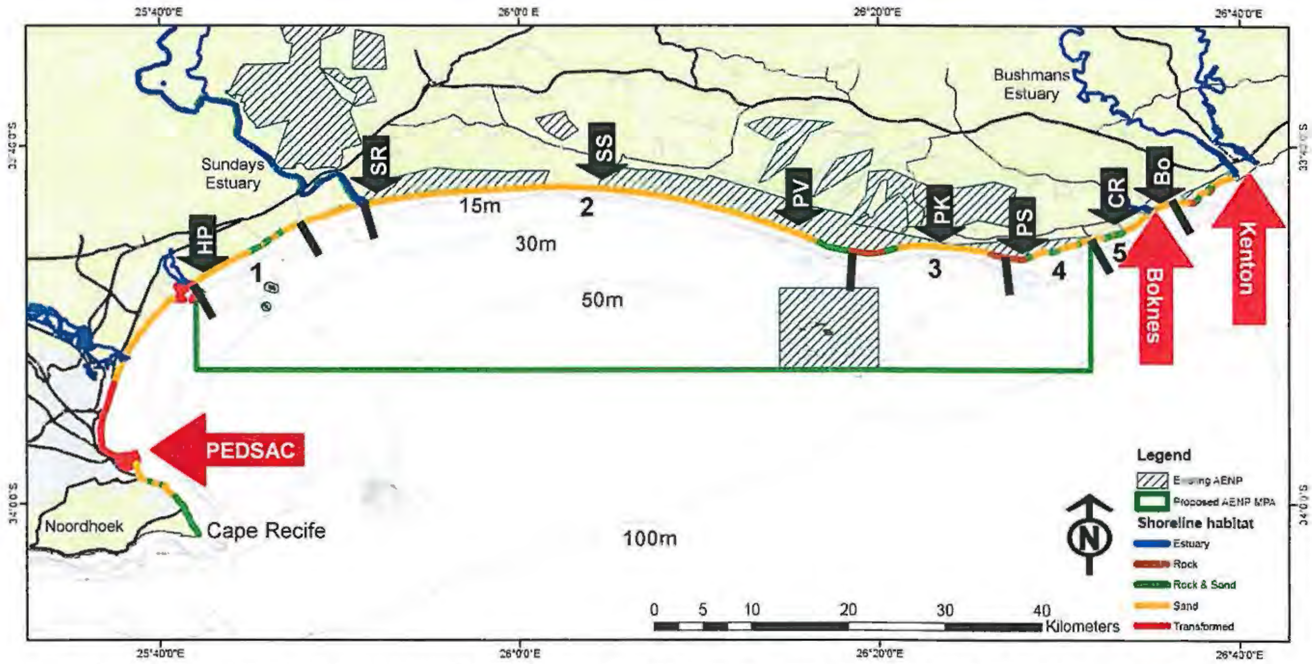


Figure 5.1. Distribution of main beach access points (black arrows) and vessel launch sites (red arrows) in Algoa Bay. Numbers 1-5 indicate the roving creel survey zones. HP= Hougham Park, SR=Sundays River, SS=Sundays Surf, PV=Perdevlei, PK=Perdekloof, PS=Pump Station, CR=Cannon Rocks, Bo=Boknes.

All people observed during the survey were counted, their activities noted and their positions recorded using a handheld GPS. Where possible all people engaged in fishing or bait collecting activities were approached and individually interviewed for catch and effort information. In certain instances when large groups of anglers were encountered one angler was asked to represent the group and his/her demographic information was obtained and catch and effort data for the whole group were recorded. During peak holiday periods when it was not possible to interview all anglers within a survey period, all people were counted and their positions and activities recorded and a sub-sample of anglers were interviewed for catch and effort information.

[3] Travel direction for Zone 2 could not be randomly selected due to limited accessibility

During each interview demographic, fishing effort and catch information was requested from the anglers (Appendix 3). Demographic information included name, age, sex, race, home language, place of residence and current occupation[4]. All anglers were asked if they had the required fishing licenses and to present them for verification purposes. Although many anglers responded that they had the required licenses they either could not, or refused to produce them. As the surveyor was not empowered as a fishery inspector there was no legal requirement on their behalf to do so. However, failure to produce a license was assumed to mean that they either did not have a valid license, or did not have the license on their person as legally required and they were therefore contravening the recreational regulations. Effort information was obtained from each interviewee and included the number of fishing rods in use, time that fishing commenced, time of the interview and the expected time that they would stop fishing. Anglers were requested to produce all retained fish which were identified and measured to the nearest millimeter fork length. Anglers were also requested to provide information on the bait types used and the species and number of additional fish which had been captured but released. A sub-sample of anglers were interviewed with a long questionnaire designed to obtain socio-economic information and their views on the current management and status of marine resources (Appendix 4).

(b) Access point surveys

The limited number of formally recognised boat launching sites for skiboat anglers in Algoa Bay warranted the use of access point surveys for assessing catch and effort in recreational skiboat fishery. Access point surveys involve on-site contact with anglers on return from their fishing trips and unlike roving creel surveys, catch and effort data can be obtained for the complete fishing trip (Pollock *et al.* 1994). Three access points were identified for the quantification of recreational effort and catch and included PEDSAC, Boknes and Kenton. All three sites are legally registered for vessels launching to sea and commonly used by recreational anglers. Two types of access point surveys were conducted; the first was designed to obtain detailed information on launching effort at each site, and the second survey was designed to obtain catch and effort information.

(i) Access point launching effort surveys

The design of the surveys differed between sites due to the local characteristics of each site.

PEDSAC

All vessels departing from PEDSAC are required to sign out in a logbook as part of the harbour safety requirements. Information recorded in the logbook includes vessel identification, skipper name, crew number, time of departure and time of return as well as the site name of the destination. Detailed daily launch records were therefore available for all vessels launching from this site and were obtained for a three-year period (June 2006-May 2009). Although the vessel identification, crew number and times of departure and return are considered accurate as it is a legal requirement for all members, the spatial data is questionable as anglers are hesitant to provide information on their preferred fishing areas. Spatial information was therefore verified during angler interviews and aerial surveys.

[4] Not all data was analysed in this study but was collected as it forms an important baseline for future comparative purposes

Upon review of the PEDSAC logbook data it was apparent that there were missing datasheets, as for long periods of time no vessels launched from the facilities despite suitable weather conditions (checked with local weather station data). As a result average monthly effort was determined by assessing the number of days for which the datasheets covered for each month and identifying periods over which data were missing. Where greater than three consecutive days of logbook data were missing it was assumed that logbook datasheets had been misplaced. Monthly effort was therefore estimated by determining the number of work and non-work days represented in the logbook (recorded days) and upscaling the effort by the ratio of recorded days to actual days per month for each day type (work or non-work). Where no records were missing total monthly effort was determined as the sum of all entries. Monthly effort estimates were determined by averaging the monthly data over the three-year period.

Boknes and Kenton

No launch registers are maintained at either the Boknes or Kenton launch sites and no formal club facilities are available on-site. Effort counts were conducted at the Boknes and Kenton beach launch sites three to four times per week over a 12-month period. During each survey the number of launch vehicles and vessel trailers parked at the launch sites were counted and details recorded[5]. Counts were stratified over work and non-work (weekend and public holidays) days with the time of the count being randomly selected between 07:00 and 13:00. This coincided with the peak fishing time in the region (Brouwer 1997) as strong winds and poor sea conditions often limit fishing effort in the latter half of the day.

<u>(ii) Access point catch surveys</u>

Catch interviews were undertaken at all three boat launch sites primarily over weekends and holiday periods between 2007 and 2009 when the weather conditions were favourable for offshore fishing and angler activity was therefore highest. Interviews were conducted with all skippers of recreational fishing vessels returning from sea on that sampling day. On the first encounter with a skipper demographic and catch information was obtained. Demographic information requested on first contact included skipper's name, age, race, occupation and town of permanent residence, and gender and race composition of crew members (Appendix 5). Catch information obtained for all interviews included number of anglers, time of departure and time of return (Appendix 6). In addition information on the boat's fishing locations, depths and targeted species at each location was requested as well as hook sizes and bait types used. If permission was obtained from the skipper, all retained fish on the vessel were identified and counted, and where time permitted all fish were measured to the nearest millimeter fork length. If the catch was large a random sub-sample was taken for measurement.

[5] As part of marine safety regulations all trailers are required to be marked with the owner's name, contact number and vessel name.

(c) Aerial surveys

A small fixed wing aeroplane was used to conduct aerial surveys during which counts of shore-based recreational beach users and offshore boating activity were conducted between Noordhoek (west of Cape Recife, Figure 5.1) and the Bushmans River Mouth. During each survey a low level coastal flight path (300-400ft) over the surf zone was used to count shore-based recreational activity (all fishing and non-fishing activities), while offshore boating activity was documented during the return flight, which was conducted at a slightly higher altitude (500-600ft). A hand-held GPS was set to record the flight path at 5-second intervals and the time of all observations recorded by the surveyors was documented to allow synchronisation with the flight path recorded by the GPS unit. A digital camera was used to capture information on high use areas with photographs analysed on completion of the survey. The digital camera and GPS clocks were synchronised to ensure the spatial accuracy of data. Activities were identified as accurately as possible and later grouped into two categories, fishing or non-consumptive use.

Flights were conducted on an *ad hoc* basis during the study period depending on pilot and plane availability, and weather conditions, with visibility and wind strength being the main considerations. Aerial surveys were primarily used to assess the spatial distribution of coastal use and were therefore conducted over weekends and holiday periods when activity was greatest. Due to the lower frequency and *ad hoc* nature of aerial surveys, the relative proportion of effort per coastal segment was compared to the high resolution shore activity data obtained during the roving creel survey where study areas overlapped.

5.2.3 Data analysis

(a) Roving creel survey

Estimates of fishing effort and catch rate are required in order to quantify the total harvest of a fishery (Pollock *et al.* 1994). However, catch rates in recreational fisheries are typically low with a large proportion of the captured fish being returned as they are either under the minimum legal size limit (MLS) or considered inedible (sharks and rays). As interviews were conducted while anglers were engaged in fishing activities (incomplete fishing trip) the total number of fish caught per angler was also generally low. However, due to low catch rates shore anglers tend to have good recollection of the species, numbers, and in some instances estimated sizes, of fish released prior to the interview. It was therefore possible to differentiate between total catch ($CPUE^T$) (all fish hooked and landed) and retained catch ($CPUE^R$) (all fish not released) for the recreational shore fishery and to calculate both CPUE components for each angler.

Due to the low catch rates and high release rates in recreational fisheries, CPUE data are often zero-inflated and requires non-standard analyses to accommodate the large proportion of zero values. One such approach is to analyse zero-inflated data using a two-step approach involving a binary (zero/non-zero) response and the conditionally distributed non-zero catches (Stefánsson 1996; O'Neill and Faddy 2003; Ellender *et al.* 2010). This allows different factors potentially influencing the two processes to be taken into consideration during the analysis. This is known as the Delta-X approach and has been used in fisheries assessments to improve confidence around the estimates of CPUE

(Maunder and Punt 2004; Fletcher *et al.* 2005; Ellender *et al.* 2010). In this approach Delta refers to the binary process in which an event may or may not occur, while the X refers to the distribution of the positive values following an event occurring. In this instance Delta is the probability of an angler catching a fish, the probability of capture (*Pc*), which is a binary variable where a value of 0 indicates no fish were caught, and 1 indicates that one or more fish were caught during the outing. X refers to the CPUE of the positive catches ($CPUE_{Pos}$) only, where anglers caught one or more fish. $CPUE_{Pos}$ is usually normally distributed following a natural logarithm transformation. $CPUE_{Pos}$ was calculated for each angler or angler group as the number of fish caught per angler-hour (fish.angler-hour^{-1}) using the following equation:

$$CPUE^{c} = \frac{n}{h} \qquad \text{Equation 5.1}$$

Where CPUE is the positive catch rate in number of fish.angler-hour^{-1} where *c* is either the total (*T*) or retained (*R*) catch component, *n* is the total or retained number of fish caught by the angler and *h* is the angling effort of the fishing trip at the time of interview.

Adjusted CPUE for any time period or access point was calculated by integrating the *Pc* and $CPUE_{Pos}$ using the delta lognormal approach in the form:

$$\overline{CPUE}^{c} = Pc^{c} \times exp(log\,CPUE^{c}_{Pos} + \frac{\sigma^{2}}{2}) \qquad \text{Equation 5.2}$$

Where *c* is either *T* or *R* for total or retained catch respectively *Pc* is the probability of the angler having caught a fish, *exp* is the antilog of the natural logarithm, $CPUE_{Pos}$ is the positive catch and σ^2 is the variance of $logCPUE_{Pos}$.

In order to avoid potential biases as a result of extreme catch rates which may have occurred by chance at the start of the fishing trip, all interviews where the fishing duration to time of interview was less than 30 minutes were discarded (Pollock *et al.* 1994).

GLMs were used to determine the influence of external factors on fishing effort (number of anglers and fishing duration) and CPUE (probability of capture and $CPUE_{Pos}$) and the importance of such factors in estimating annual effort and harvest in the fishery. AIC was used to assist in identifying the optimal combination of factors in each model. Interactions were not included due to the number of factors being investigated and complexities of each model. The sample size for each model was large and the number of factors included in the final model never exceeded one third of the sample size (Crawley 1993).

(i) Angler number

The effect of accessibility of fishing locations on angler number was tested using a non-parametric Kruskal-Wallis ANOVA. Each access point to the coastline in the study area was categorised into one of four categories according to ease of accessibility to the general public based on the proximity to urban settlements, and the nature of access roads to the site. Access points providing easy access were situated within a 2km radius of a residential area, moderately accessible sites were situated within a 10km radius and had clearly demarcated roads, sites difficult to access were situated >10km from a main tarred road, and access points classified as very difficult required either an off-road vehicle (ORV) to access the coastline or a walking distance of greater than 3km to access the coastline at the nearest point.

The effect of spatial, temporal and environmental effects on the number of anglers (*An*) was assessed using a GLM with a Poisson distribution as it is based on count data. The log-link function is commonly used for Poisson models and dispersion was taken into account, both of which are described in Chapter 3. Explanatory variables thought to influence *An* included season, day type, sample period, access point, wind and tidal phase. The year was divided into four seasons of three months each, with summer commencing on first of December each year. Day type was divided into work and non-work days, with weekends and public holidays being grouped as non-work days. Sample period was the time of day at which the count was undertaken, being classified as morning, midday or afternoon as described above. Vehicular access to the coastline by anglers within the study area was limited, and seven main access points commonly used by angers were identified, namely Hougham Park (HP), Sundays River (SR), Sundays Surf (SS), Perdevlei (PV), Perdekloof (PK), Pump Station (PS), Cannon Rocks (CR) and Boknes (Bo). Mean wind speed over the sample period was calculated from data obtained from the South African Weather Services (SAWS) Bird Island weather station and used as a continuous predictor for weather conditions likely to influence angler number during any particular period. Tidal phase was the moon phase at which the survey was conducted being either neap or spring tides. To model the influence of these parameters on *An* the following GLM was applied:

$$log(An) = \beta_0 + \beta_1(season) + \beta_2(daytype) + \beta_3(sample\ period) + \beta_4(access\ point) + \beta_5(tide) + \beta_6(wind) + \varepsilon \quad \text{Equation 5.3}$$

Where $\beta_{0\text{-}i}$ are the estimated explanatory variables and ε the error (McCullagh and Nelder 1995).

<u>*(ii) Fishing duration*</u>

The length of time an angler spends fishing, fishing duration (*Fd*), is a continuous dependent variable. After log transformation *Fd* was approximately normally distributed and was therefore modelled with a GLM with the identity-link function as described in Chapter 3. Explanatory variables likely to influence *Fd* include season, day type, access point, walking distance, wind, tidal phase and the time that the angler commenced fishing. Walking distance was calculated as the distance from any one of the seven main vehicular access points which an angler covered in order to reach his/her fishing location. Walking distance was determined in ArcGis and categorised into 500, 1000, 2000, 4000 and 8000m intervals. Starting time was taken as the nearest hour to which an angler arrived at his fishing location and commenced fishing activities. All other explanatory variables were as described above. The following GLM was applied to model the effects of the explanatory variables on *Fd*:

$$Fd = \beta_0 + \beta_1(season) + \beta_2(daytype) + \beta_3(accesspoint) + \beta_4(walkingdistance) + \beta_5(tide) + \beta_6(wind) + \beta_7(startingtime) + \varepsilon \quad \text{Equation 5.4}$$

Where $\beta_{0\text{-}i}$ are the estimated explanatory variables and ε the error (McCullagh and Nelder 1995).

(iii) Probability of capture

The probability of capture for total (Pc^T)(all fish including elasmobranches) and retained catch (Pc^R) (all fish not released) were modelled using a binomial distribution. Values of 0 and 1 were used to represent failure or success in capturing one or more fish by the time of interview respectively. Logistic regression was conducted for both Pc^T and Pc^R. The logit-link function is commonly used for the binomial distribution in the form:

$$f(p) = log(\frac{p}{1-p}) \qquad \text{Equation 5.5}$$

Where p is the underlying continuous probability of the binary dependent variable, ranging from 0 to 1 (McCullagh and Nelder 1995).

With the exception of habitat type and effort, all explanatory variables influencing Pc were similar to those influencing An and Fd and are described above. Habitat was divided into two categories, rock or sand, and was recorded during the interview process. Effort was recorded as the angling time up until the interview and was included as an offset in the model (Maunder and Punt 2004). The influence of the explanatory variables on Pc^T was modelled with a GLM of the form:

$$\begin{aligned} log(p(Pc)/(1-p(Pc))) = \beta_0 + \beta_1(season) + \beta_2(sampleperiod) + \beta_3(accesspo\,int) \\ + \beta_4(habitat) + \beta_5(walkingdis\,tan\,ce) + \beta_6(tide) + \beta_7(effort) + \beta_8(wind) + \varepsilon \end{aligned} \qquad \text{Equation 5.6}$$

Where $\beta_{0\text{-}i}$ are the estimated explanatory variables and ε the error (McCullagh and Nelder 1995).

Additional explanatory variables influencing Pc^R include whether or not the angler caught an edible fish and whether it was above the MLS. These variables were included as continuous variables based on the proportions of the total catch comprised of edible and legally sized fish. To model the effect of explanatory variables on Pc^R a GLM of the following form was applied:

$$\begin{aligned} log(p(Pc)/(1-Pc))) = \beta_0 + \beta_1(season) + \beta_2(sampleperiod) + \beta_3(accesspo\,int) \\ + \beta_4(habitat) + \beta_5(walkingdis\,tan\,ce) + \beta_6(tide) + \beta_7(effort) + \beta_8(wind) \\ + \beta_9(edible) + \beta_{10}(legal) + \varepsilon \end{aligned} \qquad \text{Equation 5.7}$$

Where $\beta_{0\text{-}i}$ are the estimated explanatory variables and ε the error (McCullagh and Nelder 1995).

Where required post-hoc testing was conducted using the Bonferroni Adjustment as outlined in earlier chapters.

(iv) Positive CPUE

The positive CPUE for total and retained catch is a continuous dependent variable, which after log transformation approximates the normal distribution. The identity-link function is commonly used for the normal distribution (McCullagh and Nelder 1995) as described above. Explanatory variables likely to influence positive CPUE were the same as those described above for the probability of capture.

GLMs of the following forms were applied to model the effect of explanatory variables on $CPUE_{Pos}{}^{T}$ and $CPUE_{Pos}{}^{R}$:

$$CPUE_{Pos}{}^{T} = \beta_0 + \beta_1(season) + \beta_2(sampleperiod) + \beta_3(accesspoint) + \beta_4(habitat) + \beta_5(walkingdistance) + \beta_6(tide) + \beta_7(wind) + \varepsilon \qquad \text{Equation 5.8}$$

and

$$CPUE_{Pos}{}^{R} = \beta_0 + \beta_1(season) + \beta_2(sampleperiod) + \beta_3(accesspoint) + \beta_4(habitat) + \beta_5(walkingdistance) + \beta_6(tide) + \beta_7(wind) + \beta_8(edible) + \beta_9(legal) + \varepsilon \qquad \text{Equation 5.9}$$

Where $\beta_{0\text{-}i}$ are the estimated explanatory variables and ε the error (McCullagh and Nelder 1995).

(v) Estimation of annual catch and effort

The survey design was stratified in order to take the spatial and temporal variability into account. Spatial and temporal factors which were indentified as significant in the GLM analyses were used to stratify the calculation of annual Angler number (*An*), mean fishing duration (*Fd*) and adjusted CPUE in order to calculate the estimated total and retained catch for the recreational shore fishery.

Angler number

The estimated number of anglers utilising each access point per season was estimated using the following equation:

$$\overline{An_{ijk}} = \left[\frac{\sum^{n} An_{ijk}^{wd}}{n_{ijk}^{wd}} \times wd_k\right] + \left[\frac{\sum^{n} An_{ijk}^{nw}}{n_{ijk}^{nw}} \times nw_k\right] \times 2.48 \qquad \text{Equation 5.10}$$

Where $\overline{An_{ijk}}$ is the estimated number of anglers during period *i* in season *k* at access point *j*, An_{ijk}^{wd} and An_{ijk}^{nw} are the number of anglers encountered during work and non-work days respectively, wd_k and nw_k are the average number of work and non-work days in season *k*, n_{ijk}^{wd} and n_{ijk}^{nw} are the number of survey days conducted during period *i* in season *k* at access point *j*, and 2.48 is the estimated turnover rate of anglers during the course of the day (Brouwer *et al.* 1997; Brouwer 1997).

Fishing effort

Fishing effort was calculated as the product of the estimated number of anglers $\overline{An_{ij}}$ and mean fishing duration (*Fd*) per stratum as per the equation:

$$Effort_{ij} = \overline{An_{ij}} \times Fd_{ij} \qquad \text{Equation 5.11}$$

Where $Effort_{ij}$ is the estimated fishing effort in angler.hours^{-1} during period i at access point j, $\overline{An_{ij}}$ is the estimated number of anglers during period *i* at access point *j*, and Fd_{ij} is the mean fishing duration of anglers during period *i* at access point *j*.

Total annual fishing effort per access point, $\overline{Effort_j}$, was estimated as the sum of the seasonal effort at access point *j*, and annual effort for the survey area was estimated as the sum of all effort at each access point.

Adjusted CPUE was calculated for each access point as described above and used in the calculation of total and retained annual catch per access point as follows:

$$Catch_j = \overline{CPUE_j} \times \overline{Effort_j} \qquad \text{Equation 5.12}$$

Where $Catch_j$ is the number of fish caught annually at access point *j*, $\overline{CPUE_j}$ is either the $CPUE^T$ or $CPUE^R$ at access point *j*, and $\overline{Effort_j}$ is the estimated annual effort in angler-hours^{-1} at access point *j*.

Annual total and retained catch for the survey area was estimated as the sum of total and retained catch at each access point.

Non-parametric bootstrapping procedures (Efron 1981) were used to estimate 95% confidence intervals around the mean values for *An*, *Fd*, *Pc*, *CPUE* and catch by randomly selecting values with replacement from the existing datasets 1 000 times and recalculating fishing effort, CPUE and total catch for each randomly generated dataset. Confidence intervals were then calculated using the percentile method (Efron 1981).

(vi) Estimation of annual shore fishing effort in Algoa Bay

Angler counts from aerial surveys were used to determine the spatial distribution and number of anglers fishing within Algoa Bay. The ratio of angler number for the whole study area (Cape Recife to Bushmans River Mouth) and the detailed roving creel survey area was used to upscale the fishing effort and catch to determine annual estimates for Algoa Bay.

(b) Access point surveys

(i) Launching effort

Fishing effort in the recreational skiboat fishery was assessed through the number of vessels launching from each launch site determined through effort counts, and the number of crew per vessel and fishing duration as determined from catch interviews. The influence of different explanatory variables on these parameters was investigated using GLMs.

Explanatory variables potentially influencing vessel number (*Vn*) were launch site, year, month, day type and wind speed. The launch site was the location at which effort counts were undertaken, being either PEDSAC, Boknes or Kenton. All other explanatory variables were as described above under the analysis of roving creel data. The number of vessels launching per day is based on count data and therefore approximates the Poisson distribution for which the log-link function is commonly used as described above. To model the effects of factors on Vn the following GLM was applied:

$$log(Vn) = \beta_0 + \beta_1(launchsite) + \beta_2(year) + \beta_3(month) + \beta_4(daytype) + \beta_5(wind) + \varepsilon \qquad \text{Equation 5.13}$$

Where β_{0-i} are the estimated explanatory variables and ε the error.

(ii) Crew size

Crew size is also based on count data and was modelled in a similar fashion to launching effort above (Equation 5.13) with the exclusion of wind as an explanatory variable.

(iii) Fishing duration

Fishing duration (*Fd*) is a continuous variable and was normally distributed and was therefore modelled using a GLM with the identity-link function using the following equation:

$$Fd = \beta_0 + \beta_1(launchsite) + \beta_2(year) + \beta_3(month) + \beta_4(daytype) + \beta_5(wind) + \varepsilon \qquad \text{Equation 5.14}$$

Where $\beta_{0\text{-}i}$ are the estimated explanatory variables and ε the error.

(iv) CPUE

CPUE for each vessel was calculated as:

$$CPUE = \frac{n}{Cr \times Fd} \qquad \text{Equation 5.15}$$

Where CPUE is the number of fish.angler-hour^{-1}, n is the number of fish retained, Cr is the number of crew and Fd is the fishing duration from time of departure to time of return to the launch size. As CPUE was log-normally distributed the mean was calculated as the mean of the log-transformed values using the following equation:

$$\overline{CPUE} = exp(log\,CPUE + \frac{\sigma^2}{2}) \qquad \text{Equation 5.16}$$

Where $\overline{CPUE}$ is the mean catch rate in fish.angler-hour^{-1}, $logCPUE$ are the log-transformed CPUE values for each vessel, and σ^2 is the variance of $logCPUE$.

The influence of explanatory variables on logCPUE was investigated using a GLM with the identity-link function using the following equation:

$$CPUE = \beta_0 + \beta_1(launchsite) + \beta_2(year) + \beta_3(month) + \beta_4(wind) + \varepsilon \qquad \text{Equation 5.17}$$

Where $\beta_{0\text{-}i}$ are the estimated explanatory variables and ε the error.

(v) Estimation of annual catch and effort

Annual effort in boat-days for each access point was calculated as the sum of the monthly work and non-work day effort calculated using the following equation:

$$Em = \left[\frac{\sum Vn^{wd}}{n^{wd}} \times d^{wd}\right] + \left[\frac{\sum Vn^{nw}}{n^{nw}} \times d^{nw}\right] \qquad \text{Equation 5.18}$$

Where Em is the monthly effort in boat-days, Vn is the number of vessels launching on work (*wd*) and non-work (*nw*) days, n is the number of days on which effort counts were conducted during work (*wd*) and non-work (*nw*) days respectively, and d is the number of work (*wd*) and non-work (*nw*) days in the month.

Annual effort in angler hours per launch site was calculated using the following equation:

$$Ey_i = bd_i x\overline{CS_i} x\overline{Fd_i} \qquad \text{Equation 5.19}$$

Where Ey_i is the annual effort in angler hours at launch site i, bd_i is the estimated annual effort in boat-days, $\overline{CS_i}$ is the mean crew size, and $\overline{Fd_i}$ is the mean fishing duration at launch site i.

Total catch (number of fish) per launch site was estimated as the product of estimated annual effort and average CPUE using the following equation:

$$C_i = Ey_i x\overline{CPUE_i} \qquad \text{Equation 5.20}$$

Where C_i is the estimated total annual catch at launch site i, Ey_i is the estimated annual effort in angler-hours, and $\overline{CPUE_i}$ is the mean catch rate at launch site i.

Confidence intervals for CPUE, effort, crew size and fishing duration were estimated using non-parametric bootstrap procedures (Ehler 1981) by randomly resampling the datasets with replacement 1 000 times and recalculating the values. The percentile method (Efron 1981) was then used to estimate 95% confidence intervals around the mean.

5.2.4 Spatial indices of recreational activities

(a) Index of relative recreational importance (IRRI)

An index of relative recreational importance (IRRI) was developed in order to present the combined shore and skiboat recreational fishing effort spatially. The coastline between Cape Recife and Bushmans River Mouth was divided into 2km segments. The mean number of shore anglers per coastal segment was determined using aerial survey count data and relative shore fishing effort per segment was determined as the mean percentage of anglers per 2km segment using the following equation:

$$Seg_i = \frac{\overline{An_i}}{An_{total}} x100 \qquad \text{Equation 5.21}$$

Where Seg_i is the relative percentage of shore fishing effort in coastal segment i, An_i is the mean number of anglers determined from aerial survey data, An_{total} is the mean total number of shore anglers in the Algoa Bay study area from aerial survey data.

Offshore recreational fishing grounds in Algoa Bay were identified through access point interviews with skiboat anglers and were defined spatially using ArcMap 9.2. The relative importance of each fishing ground per access point was determined based on the number of fishing trips to each fishing ground relative to the total number of fishing grounds visited per access point using the following equation:

$$FG_{ij} = \frac{Ntrip_{ij}}{Ntotal_j} x100 \qquad \text{Equation 5.22}$$

Where FG_{ij} is the relative effort at fishing ground i originating from launch site j, $Ntrip_{ij}$ is the number of fishing trips to fishing ground i from launch site j, and $Ntotal_j$ is the total number of fishing trips to all fishing grounds originating from access point j.

The number of boat days fished per fishing ground was estimated as the relative effort at each fishing ground (%) multiplied by the estimated total effort per access point (Section 5.2.3b) using the following equation:

$$BD_{ij} = FG_{ij} x bd_j \qquad \text{Equation 5.23}$$

Where BD_{ij} is the estimated number of boat-days at fishing ground i originating from launch site j, FG_{ij} is the relative effort at fishing ground i originating from launch site j, and bd_j is the estimated annual effort in boat-days at launch site j.

The total effort in boat-days per fishing ground was determined as the sum of effort from each of the three launch sites and effort values were standardised to a percentage of the total estimated annual effort in the Algoa Bay recreational skiboat fishery.

$$Re lFG_i = \frac{\sum BD_i}{BD_t} x100 \qquad \text{Equation 5.24}$$

Where $RelFGi$ is the relative importance of fishing ground i, DB_i is the sum of the estimated number of boat-days at fishing ground i from all access points, BD_t is the estimated total recreational skiboat effort in Algoa Bay.

The spatial indices of shore and skiboat recreational fishing effort were integrated to form the IRRI. Each index was intersected with a 1km^2 grid of the study area and the relative importance of each grid cell was determined for the shore and skiboat sectors. The index of relative recreational importance was calculated as the sum of the relative shore and skiboat effort per km^2 using the equation below and was displayed spatially in ArcMap 9.2.

$$IRRI = (S_{Peffort}) + (B_{Peffort}) \qquad \text{Equation 5.25}$$

Where $IRRI$ is the index of relative recreational importance per km^2, $S_{Peffort}$ is the relative percentage of shore fishing effort per km^2, and $B_{Peffort}$ is the relative percentage of recreational skiboat effort per km^2.

(b) Economic index of relative recreational importance (Economic IRRI)

An economic IRRI was developed in order to take into account the differing economic values of the recreational sector in Algoa Bay and highlight areas which are potentially of greatest economic importance to the recreational fishery. The direct economic value of both shore and skiboat sectors was determined as the product of the estimated annual effort per sector and the average daily expenditure determined through roving creel and access point interviews respectively, and summed to determine the total economic value of the recreational sector in Algoa Bay as outlined in the following equation:

$$REv = \left(S_{effort} \times \overline{S_{expenditure}}\right) + \left(B_{effort} \times \overline{B_{expenditure}}\right) \qquad \text{Equation 5.26}$$

Where REv is the estimated annual economic value of the recreational sector in Algoa Bay, S_{effort} is the estimated annual shore effort in angler-days, $\overline{S_{expenditure}}$ is the mean daily expenditure of shore anglers, B_{effort} is the estimated annual recreational skiboat effort, and $\overline{B_{expenditure}}$ is the mean daily expenditure of recreational skiboat anglers in Algoa Bay. The relative proportion that each sector contributed to the total economic value was used to scale the relative levels of fishing effort per km^2. The economic IRRI was determined as the sum of the scaled shore and skiboat effort per km^2:

$$EIRRI = \left(S_{Peffort} \times \frac{SEv}{REv}\right) + \left(B_{Peffort} \times \frac{BEv}{REv}\right)$$

Where $EIRRI$ is the economic index of relative recreational importance per km^2, $S_{Peffort}$ is the proportion of shore effort per km^2, SEv is the estimated economic value of the recreational shore fishery in Algoa Bay, REv is the total estimated value of recreational fisheries in Algoa Bay, $B_{Peffort}$ is the proportion of skiboat effort per km^2, and BEv is the estimated economic value of the recreational skiboat sector in Algoa Bay.

5.3 Results

5.3.1 Recreational shore fishery

A total of 193 roving creel surveys were conducted within the survey area between 1 April 2006 and 31 March 2009, during which 1 916 people were encountered of which 790 (41.2%) and 70 (3.7%) were engaged in fishing and bait collection activities respectively (the remainder were engaged in non-consumptive recreational activities) (Table 5.2). A total of 557 fishery interviews were conducted representing 70.5% of the total number of anglers observed, with a total of 73 (9.2%) socio-economic fishery interviews conducted. With the exception of one subsistence angler[6] who produced a valid subsistence fishery permit, all anglers interviewed had recreational fishing permits or claimed to be recreational anglers.

(a) Angler density

Mean angler density per zone during roving creel surveys ranged from 0.037±0.08 anglers.km^{-1} in the Sundays Surf (SS) zone to 1.18±1.04 anglers.km^{-1} in the Hougham Park (HP) zone, while angler density determined by aerial surveys ranged from 0.01±0.02 anglers.km^{-1} to 1.85±2.91 anglers.km^{-1} at the SS and HP access points respectively (Table 5.1). Angler density was highest at easily accessible areas adjacent to urban residential areas. The overall mean density for the study area between Coega and the Boknes Estuary from roving creel and aerial surveys was 0.48 and 0.67 anglers.km^{-1} respectively. The mean angler density determined through aerial surveys for the whole Algoa Bay coastline from Cape Recife to the Bushmans River Mouth was 1.01 anglers.km^{-1}.

Table 5.1. Mean angler density per zone listed by access point (from west to east) with the coefficient of variation (CV).

Access point	Zone length (km)	Roving creel		Aerial	
		Density (anglers.km^{-1})	CV	Density (anglers.km^{-1})	CV
Hougham Park (HP)	6.565	1.18±1.04	88	1.85±2.91	158
Sundays River (SR)	7.728	0.41±0.51	124	0.81±0.82	101
Sundays Surf (SS)	17.216	0.03±0.08	258	0.01±0.02	282
Perdevlei (PV)	17.938	0.04±0.12	303	0.2±0.17	85
Perdekloof (PK)	12.732	0.12±0.22	181	0.24±0.21	89
Pump Station (PS)	6.39	0.24±0.46	191	0.31±0.48	154
Cannon Rocks (CR)	7.328	0.74±0.76	103	0.97±0.96	99
Boknes (Bo)	2.465	1.07±1.96	183	0.96±0.78	81

[6] A targeted household survey of subsistence anglers in the surrounding areas was conducted but results were not included in this thesis as effort was quantified through the shore fishery surveys.

Table 5.2. Survey effort and main recreational activities observed per survey zone.

Survey zone	Number of surveys	Number of people per activity group								Number of fishery interviews	Number of socio-economic interviews
		Total all activities	Fishing	Bait collection *	Walking	Walking with dogs	Swim / Play	Relax	Other		
Zone 1: Hougham Park (HP)	13	179	101	-	-	-	16	62	-	59	6
Zone 2: Sundays River to Perdevlei (SR - PV)	45	318	200	18	3	-	51	52	-	165	27
Zone 3: Perdekloof (PK)	45	178	101	22	13	2	34	10	8	68	10
Zone 4: Pump Station to Cannon Rocks (PS - CR)	45	197	112	9	35	2	21	27	-	83	13
Zone 5: Boknes to Cannon Rocks (Bo - CR)	45	1044	276	21	271	80	171	200	25	182	17
TOTAL	**193**	**1916**	**790**	**70**	**322**	**84**	**293**	**351**	**33**	**557**	**73**

* Many people were engaged in angling and bait collection simultaneously

(b) Factors influencing fishing effort and catch

(i) Angler number

Accessibility of the main access points along the coastline influenced the number of recreational shore anglers participating in the fishery significantly (Kruskal-Wallis ANOVA; df = 327, $p<0.001$). Greater numbers of anglers accessed the coast at easily (Median=2.0, 95% Confidence intervals (CI)=2.70-4.92) and moderately accessible (Median=2.5, CI:2.88-5.54) sites than at the difficult (Median=0, CI:0.743-2.680) and very difficult sites (Median=0, CI:0.54-1.32) (Figure 5.2a). Walking distance from vehicular access points affected angler number significantly (Kruskal-Wallis ANOVA; df=2 624, $p<0.001$) with the majority of anglers (63%) encountered within the first kilometer and angler number decreasing thereafter (Figure 5.2b).

Access point ($p<0.001$), season ($p<0.001$) and day type ($p=0.043$) were significant predictors of angler number (Table 5.3). The highest mean number of anglers were encountered at Hougham Park ($\bar{x}$=7.1, CI:4.6-11.1) followed by Cannon Rocks ($\bar{x}$=4.6, CI:3.4-6.1), Sundays River ($\bar{x}$=2.9, CI:2.0-4.1) and Boknes ($\bar{x}$=2.4, CI:1.6-3.6) (Figure 5.3a). The remaining four access points were utilised by fewer anglers and did not differ significantly from each other. Angler number was greatest over the summer months ($\bar{x}$=3.3, CI:2.6-4.4), declining over autumn ($\bar{x}$=2.2, CI:1.6-3.1) and spring ($\bar{x}$=1.7, CI:1.2-2.4) with the lowest numbers of anglers occurring during the winter months ($\bar{x}$=1.0, CI:0.6-1.6)(Figure 5.3b). The number of anglers fishing during non-work days ($\bar{x}$=2.2, CI:1.7-2.9) was significantly higher than during work days ($\bar{x}$=1.6, CI:1.2-2.1) (Figure 5.3c). Time of day, wind speed and tidal phase did not affect angler number significantly.

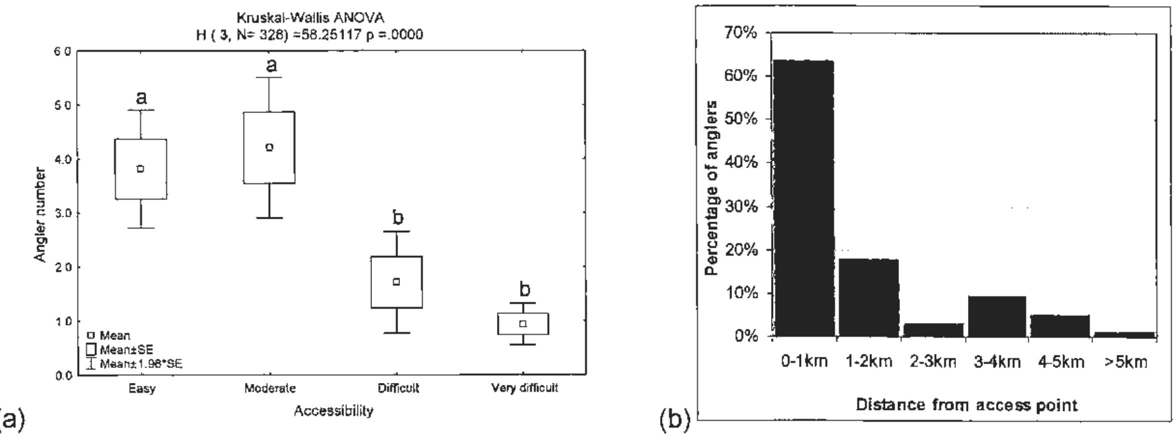


Figure 5.2. The effect of (a) accessibility to the coastline, and (b) walking distance on angler number. Letters above error bars denote significant differences determined by post hoc tests.

(ii) Fishing duration (Fd)

Access point ($p<0.001$), season ($p<0.001$) and walking distance ($p<0.001$) were significant predictors of *Fd* (Table 5.3). Fishing duration (in hours) was longest at Perdevlei ($\bar{x}$=11.6, CI:8.3-16.2), followed by Perdekloof ($\bar{x}$=8.7, CI:6.9-10.8) and Hougham Park ($\bar{x}$=8.7, CI:7.0-10.9). This was followed by the Pump Station ($\bar{x}$=8.7, CI:6.9-10.8), Sundays River ($\bar{x}$=5.9, CI:5.0-7.1) and Sundays Surf ($\bar{x}$=5.8, CI:4.3-7.8), while Cannon Rocks ($\bar{x}$=3.6, CI:3.1-4.2) and Boknes ($\bar{x}$=3.2, CI:2.6-3.9) had the shortest mean *Fd* (Figure 5.3d). Fishing duration in spring ($\bar{x}$=7.3, CI:6.4-8.2) was significantly higher than in winter ($\bar{x}$=5.6, CI:1.1-4.7), but not autumn ($\bar{x}$=5.5, CI:1.1-4.8) or summer ($\bar{x}$=6.7, CI:6.0-7.5) (Figure 5.3e). The mean *Fd* was significantly lower at 1km ($\bar{x}$=4.8, CI:4.2-5.6) than all other walking distances. However, *Fd* increased significantly with increasing walking distance from 1km onwards, with the longest *Fd* at 8km ($\bar{x}$=10.2, CI:7.6-13.8) (Figure 5.3f). Wind, tide, day type and starting time did not have a significant influence.

(iii) Probability of capture (Pc)

Access point ($p<0.001$), season ($p=0.008$), walking distance ($p<0.030$) and tidal phase ($p<0.047$) were significant predictors of Pc^T (Table 5.3). Pc^T was significantly lower at Hougham Park ($\bar{x}$=0.06, CI:0.02-0.16) than Perdekloof ($\bar{x}$=0.32, CI:0.15-0.55), Perdevlei ($\bar{x}$=0.64, CI:0.33-0.86), Sundays Surf ($\bar{x}$=0.49, CI:0.24-0.75) and Sundays River ($\bar{x}$=0.43, CI:0.27-0.61) (Figure 5.3g). Pc^T was higher in autumn ($\bar{x}$ =0.44, CI:0.30-0.60) and summer ($\bar{x}$=0.41, CI:0.30-0.55) than spring ($\bar{x}$=0.25, CI:0.16-0.37) and winter ($\bar{x}$=0.23, CI:0.12-0.40) (Figure 5.3h) Pc^T increased from distances of <1km ($\bar{x}$=0.32, CI:0.23-0.43) to 2-3km ($\bar{x}$=0.63, CI:0.27-0.88), but decreased after 3km with the lowest Pc^T at distances >4km ($\bar{x}$=0.08, CI:0.03-0.24) (Figure 5.3i). Pc^T was higher during neap tides ($\bar{x}$=0.38, CI:0.28-0.50) than spring tides ($\bar{x}$=0.28, CI:0.19-0.40). Sample period, habitat and wind speed were not important predictors.

Only the proportion of edible fish in the total catch was a significant predictor of the probability of capturing and retaining a fish, with probability increasing with the proportion of edible fish in the total catch (Table 5.3).

(iv) Positive CPUE

Access point ($p<0.001$) was the only explanatory variable which was an important predictor of positive $CPUE^T$ ($p<0.001$) (Table 5.3). $CPUE^T$ (in fish.angler-hour^{-1})was lowest at Hougham Park ($\bar{x}$=0.10, CI:0.06-0.18) and Perdekloof ($\bar{x}$=0.14, CI:0.10-0.22) and highest at Pump Station ($\bar{x}$=61, CI:0.41-0.91) and Sundays Surf ($\bar{x}$=0.63, CI:0.40-0.98) (Figure 5.3j).

$CPUE^R$ was influenced significantly by access point and the proportion of legally sized fish in the catch (Table 5.3). The CPUE of the retained catch component was greatest at Cannon Rocks ($\bar{x}$=0.44, CI:0.28-0.69) and Boknes ($\bar{x}$=0.44, CI:0.23-0.83), and lowest at Hougham Park ($\bar{x}$=0.15, CI:0.07-0.34) and Perdekloof ($\bar{x}$=0.07, CI:0.04-0.13) (Figure 5.3k). $CPUE^R$ of anglers who are compliant with fishery regulations is likely to be influenced by the size of fish landed as illegally sized fish are likely to be returned alive to avoid penalties. This was found to be the case as $CPUE^R$ increased with increasing proportion of legally sized fish in the catch.

Table 5.3. Main factors affecting effort and catch in the recreational shore fishery as determined using GLMs. Cells shaded in grey indicate factors which were excluded from the GLM analyses based on AIC criterion and those in black indicate factors which were irrelevant to the model. Cells highlighted in green and orange represent significant differences at p<0.05 and p<0.001 respectively.

Factor	Intercept d.f.=1	Access Point d.f.=7	Season d.f.=3	Day type d.f.=1	Period d.f.=2	Wind d.f.=1	Tide d.f.=1	Walking distance d.f.=4	Start hour d.f.=1	Habitat d.f.=2	Prop. MLS d.f.=1	Prop. Edible d.f.=1
Angler number (An)	p<0.001** WaldX^2=20.14	p<0.001** WaldX^2=68.93	p<0.001** WaldX^2=25.66	p=0.043* WaldX^2=4.10	p=0.156 ns WaldX^2=3.72	p=0.184 ns WaldX^2=1.72	N/A					
Fishing duration (*Fd*)	p<0.001** WaldX^2=426	p<0.001** WaldX^2=127.0 2	p=0.005* WaldX^2=12.96	p=0.050 ns WaldX2=3.83		N/A	N/A	p<0.001** WaldX^2=20.03	p=0.090 ns WaldX^2=2.87			
Probability of capture Total Catch (Pc^T)	p<0.001** WaldX^2=46.72	p<0.001** WaldX^2=30.58	p=0.008* WaldX^2=11.62		N/A	N/A	p<0.047* WaldX^2=3.95	p<0.030* WaldX^2=10.72		N/A		
Probability of capture Retained Catch (Pc^R)	p<0.001** WaldX^2=208	p=0.748 ns WaldX^2=4.27	p=0.95 ns WaldX^2=6.36		N/A	N/A	N/A	N/A		N/A	p=0.108 ns WaldX^2=3.58	p<0.001** WaldX^2=60.46
Positive CPUE Total Catch ($CPUE^T$)	p<0.001** WaldX^2=154.58	p<0.001** WaldX^2=66.12	p=0.17 ns WaldX^2=4.93		N/A	N/A	N/A	N/A		N/A		
Positive CPUE Retained Catch ($CPUE^R$)	p<0.001** WaldX^2=46.99	p<0.001** WaldX^2=30.63	p=0.316 ns WaldX^2=3.54		N/A	N/A	N/A	N/A		N/A	p=0.008* WaldX^2=6.91	N/A

N/A=not applicable, discarded after AIC analysis
ns=not significant
* p<0.05
** p<0.001

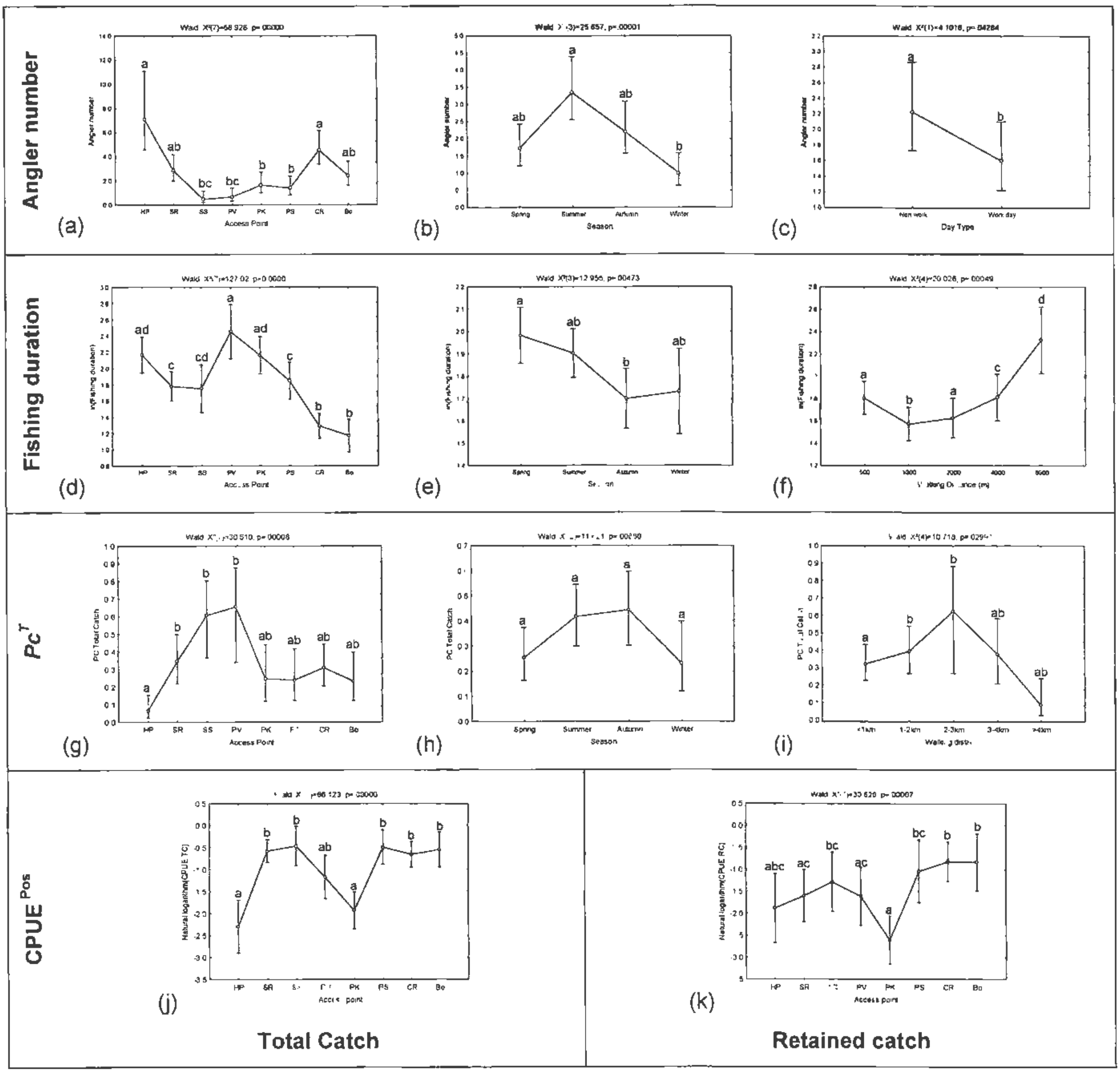


Figure 5.3. Predicted mean values of factors influencing effort and catch in the recreational shore fishery. (HP=Hougham Park; SR=Sundays River; SS=Sundays Surf; PV=Perdevlei; PK=Perdekloof; PS=Pumps Station; CR=Cannon Rocks; Bo=Boknes). Error bars denote 95% confidence intervals, letters above error bars denote significant differences determined by post hoc tests.

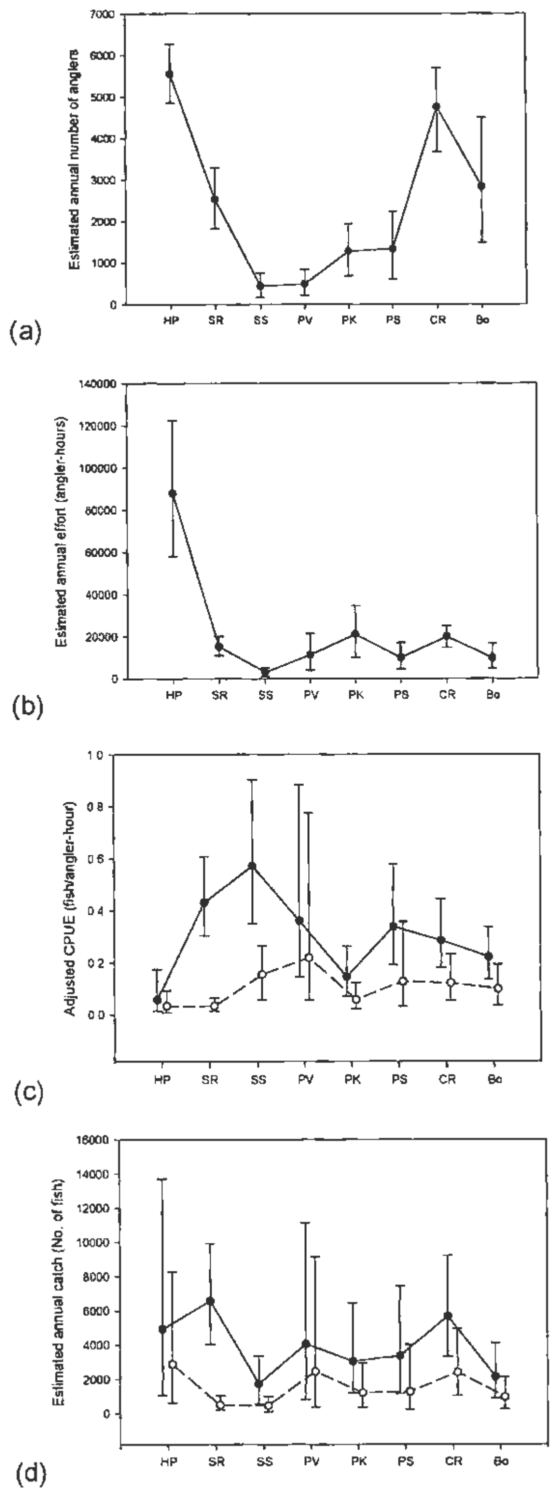


Figure 5.4. Estimated annual effort, adjusted CPUE and total (solid circles) and retained (open circles) catch per access point within the study area. Error bars denote 95% confidence intervals. HP=Hougham Park; SR=Sundays River; SS=Sundays River; PV=Perdevlei; PK=Perdekloof; PS=Pump Station; CR=Cannon Rocks; Bo=Boknes.

(c) Estimation of annual catch and effort

The total annual effort within the roving creel survey area (Coega to Boknes) was estimated at 19 198 angler-days or 178 035 angler-hours, with clear spatial patterns evident. The estimated number of angler-days was greatest in the Hougham Park area ($\bar{x}$=5 545, CI:4 664-6 244), followed by the Cannon Rocks ($\bar{x}$=4 755, CI:3 778-5 666), Boknes ($\bar{x}$=2 826, CI:1 418-4 582) and Sundays River areas ($\bar{x}$=2 529, CI:1 756-3 342) (Figure 5.4a). Angler numbers within the central regions of the survey area were far lower, ranging from 1 335 to 437 angler-days at the Pump Station (CI:602-2 258) and Sundays Surf (CI:161-774) areas respectively.

Due to a combination of angler number and fishing duration the spatial pattern of effort in angler-hours differed from that of angler number (Figure 5.4b). Effort in angler-hours was highest at Hougham Park ($\bar{x}$=87 886, CI:55 139-122 188) due to the high estimate of angler number and the long average fishing duration. Effort at the other access points ranged from 3 020 angler-hours at Sundays Surf (CI:1 112-5 152) to 21 045 angler-hours at Perdekloof (CI:10 374-35 415).

The adjusted CPUE ($PCxCPUE_{Pos}$) for $CPUE^T$ was highest at the Sundays Surf access point with a catch rate of 0.57 fish.angler-hour^{-1} (CI:0.33-0.90), followed by Sundays River ($\bar{x}$=0.43, CI:0.30-0.60) and Perdevlei ($\bar{x}$=0.36, CI:0.16-0.81), with lowest catch rates at Perdekloof ($\bar{x}$=0.15, CI:0.07-0.27) and Hougham Park ($\bar{x}$=0.06, CI:0.01-0.16) (Figure 5.4c). The overall mean $CPUE^T$ within the survey area was 0.30 fish.angler-hour^{-1}.

$CPUE^R$ was highest in the Perdevlei area, where 0.22 fish.angler-hour^{-1} were retained (Figure 5.4c). The lowest retained catch rate occurred at the Sundays River and Hougham Park access points where 0.03 fish.angler-hour^{-1} were retained. The mean $CPUE^R$ within the survey area was 0.11 fish.angler-hour^{-1}.

The estimated total and retained catches in the survey area were 31 475 and 12 102 fish per annum respectively. Estimates of total catch were greatest in the Sundays River ($\bar{x}$=6 566, CI:4 031-9 925), Cannon Rocks ($\bar{x}$=5 656, CI:3 327-9 199) and Hougham Park ($\bar{x}$=4 922, CI:1 046-13 712) areas (Figure 5.4d). However, estimates of the retained catch were highest in the Hougham Park ($\bar{x}$=2 884, CI:610-8 248), Perdevlei ($\bar{x}$=2 450, CI:379-9 139) and Cannon Rocks ($\bar{x}$=2 391, CI:1 029-4 939) access point areas (Figure 5.4d). Due to the variability in estimates of CPUE and effort there was high variability around the mean estimates of total and retained catch.

Results from aerial surveys confirmed the concentration of anglers around easily accessible areas (Figure 5.5) with a high proportion of anglers distributed along the Port Elizabeth beach front between Cape Recife and the HP access point, and few anglers occurring along the Sundays Surf areas. Total effort in Algoa Bay (Cape Recife to Bushmans River Mouth) was estimated at 54 483 angler-days.

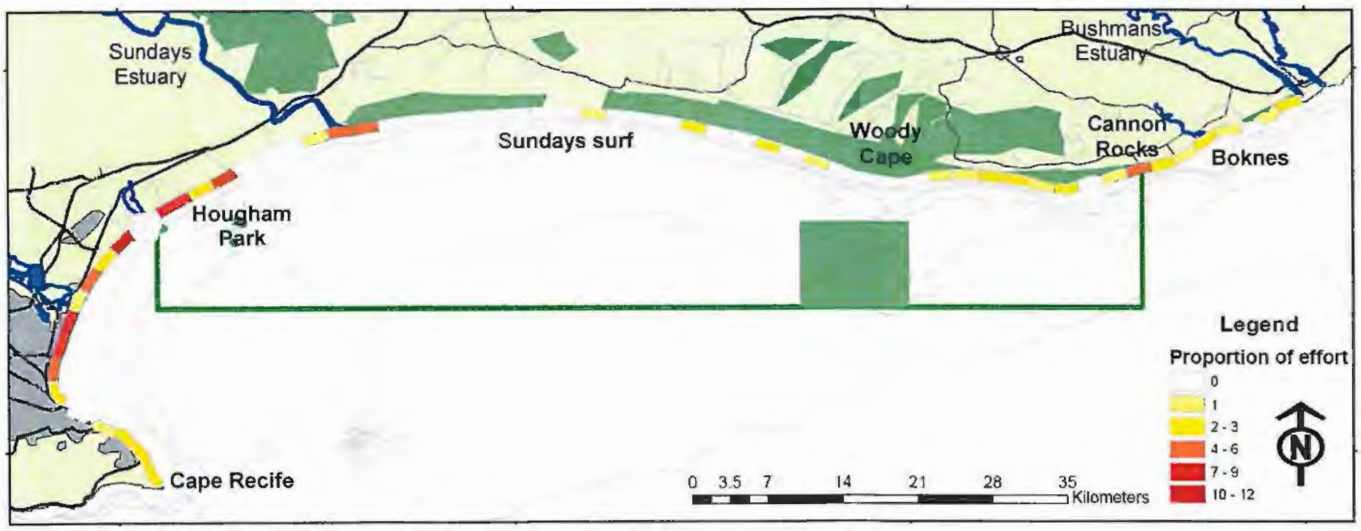


Figure 5.5. Spatial distribution of anglers across Algoa Bay area as determined by aerial surveys. Effort expressed as the proportion of mean number of anglers per 2km coastal stretch to the mean number counted between Cape Recife and Bushmans River Mouth. Green shaded areas represent the existing AENP areas, and the green line indicates the proposed AENP MPA footprint.

(d) Catch composition

A total of 606 fish representing 21 families and 34 species were landed by the recreational shore fishery (Table 5.4). Overall 22 teleost species from 13 families and 12 chondrichthyan species from 8 families were landed. Numerically five species accounted for 72% of the total catch. The lesser guitarfish (*Rhinobatos annulatus*) dominated the catch, constituting 36% by number. No other chondrichthyan species contributed greater than 5% to the overall catch by number. White seacatfish (*Galeichthys feliceps*) was the second most prominent species in the catches, accounting for 13%. The important and heavily targeted linefishery species, white steenbras (*Lithognathus lithognathus*), blacktail (*Diplodus sargus capensis*), dusky kob (*Argyrosomus japonicus*) and elf (*Pomatomus saltatrix*) accounted for 8%, 7%, 7% and 4% of the total catch by number respectively.

Composition of the catch varied spatially. The lesser guitarfish dominated catches at Sundays River (55%), Perdekloof (35%) and Perdevlei (30%), and with the exception of Boknes were landed at all other access points (Figure 5.6a). White steenbras dominated the catch at Sundays Surf, constituting 49%, while contributing 17 and 12% to the total catch at Perdevlei and Boknes respectively. Dusky kob were landed at all access points except Boknes, and comprised 13% of the catch composition at Hougham Park, Sundays River and Cannon Rocks. Blacktail dominated the individual species catches at Boknes (23%), Pump Station (17%) and Cannon Rocks (17%) and constituted 11 and 15% of the total catch at Perdevlei and Perdekloof respectively. White seacatfish comprised between 15 and 23% of the total catch at Perdevlei, Sundays River, Hougham Park and Perdekloof. There were no significant trends in seasonal catch composition.

Five species, including blacktail (20%), dusky kob (18%), white seacatfish (13%), elf (12%) and white steenbras (9%), comprised 72% of the retained catch by number with clear spatial trends apparent. Dusky kob comprised a large portion of the retained catch at Hougham Park (56%), Sundays River (36%), Perdevlei (23%) and Cannon Rocks (19%) (Figure 5.6b). White steenbras constituted 70% and 27% of the retained catch by number at Sundays Surf and Perdevlei respectively, while blacktail comprised between 20 and 36% of the catch number at Sundays Surf, Perdevlei, Perdekloof, Pump Station, Cannon Rocks and Boknes.

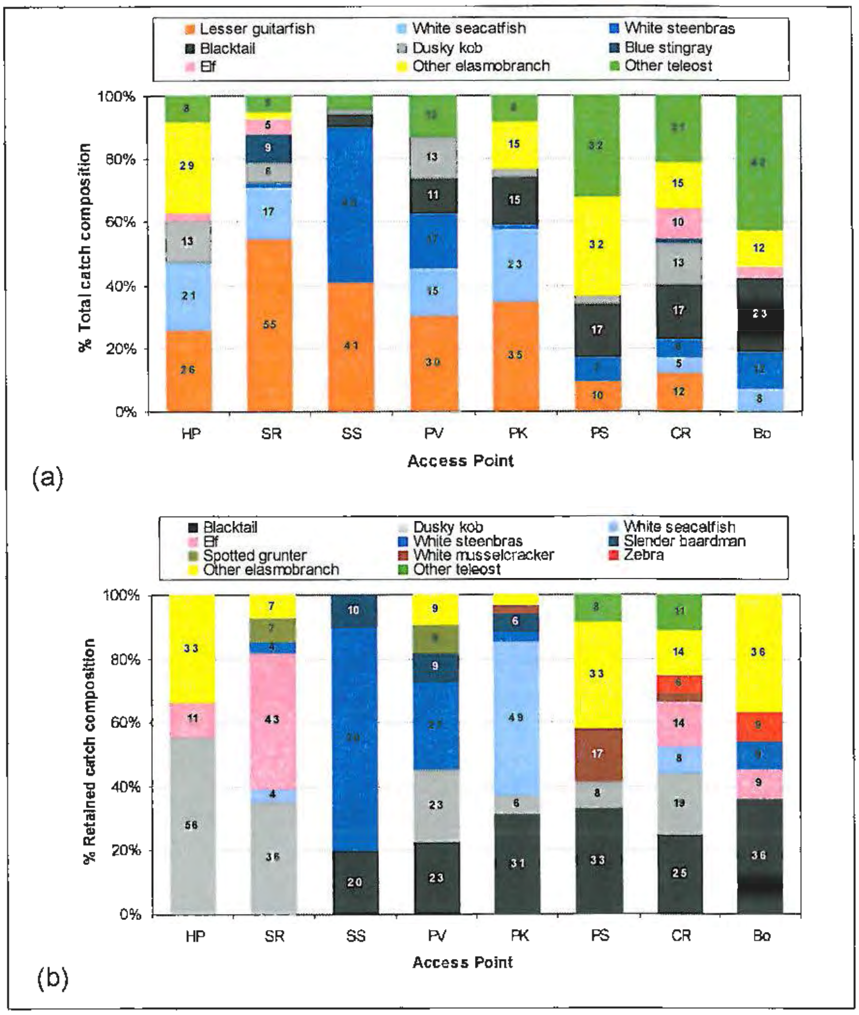


Figure 5.6. Relative contribution of the main species to the (a) total and (b) retained catch at each access point.

Table 5.4. Catch composition of the recreational shore fishery with each species expressed as percentage of total catch as well as the estimated number and weight retained annually. Species contributing greater than 5% and 10% to the total catch highlighted in green and orange respectively.

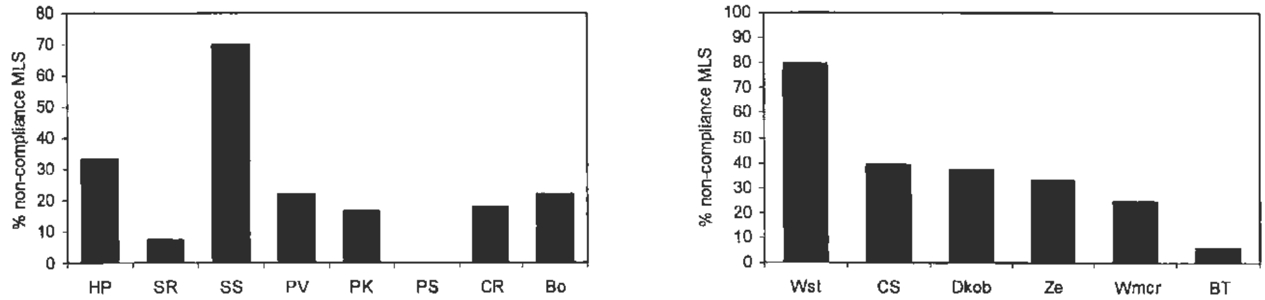
Class	Family	Scientific name	Common name	% of Total Catch	Est. number retained (95% CI)	Est. retained weight (kg)
CHONDRICHTHYES	Carcharhinidae	*Carcharhinus brachyurus*	Bronze whaler	0.5	0	
	Triakidae	*Mustelus mustelus*	Smooth-hound	0.2	0	
		Triakis megalopterus	Spotted gullyshark	0.8	0	
	Dasyatidae	*Dasyatis chrysonota*	Blue stingray	4.0	77 (48-138)	-
	Gymnuridae	*Gymnura natalensis*	Diamond ray	0.2	0	
	Myliobatidae	*Pteromylaeus bovinus*	Bull ray	1.0	0	
	Odontaspididae	*Carcharias Taurus*	Spotted ragged-tooth	1.2	0	
	Rhinobatidae	*Rhinobatos annulatus*	Lesser guitarfish	36.5	230 (145-413)	-
	Scyliorhinidae	*Haploblepharus fuscus*	Brown shyshark	2.3	0	
		Poroderma africanum	Pyjama catshark	2.5	0	
		Poroderma pantherinum	Leopard catshark	0.3	0	
	Squalidae	*Squalus megalops*	Bluntnose spiny dogfish	0.2	77 (48-138)	-
OSTEICHTHYES	Ariidae	*Galeichthys feliceps*	White seacatfish	13.0	1 609 (1 016-2 889)	1 060
	Carangidae	*Lichia amia*	Leervis	0.2	0	
		Trachinotus africanus	Southern pompano	0.2	0	
	Clinidae		Clinidae sp.	0.2	0	
	Haemulidae	*Pomadasys commersonii*	Spotted grunter	1.8	306 (194-550)	474
	Mugilidae		Mugilidae sp.	1.0	383 (242-688)	-
	Plotosidae	*Plotosus nkunga*	Eel-catfish	0.3	77 (48-138)	65
	Pomatomidae	*Pomatomus saltatrix*	Elf	3.6	1 455 (919-2 614)	1 450
	Sciaenidae	*Argyrosomus japonicus*	Dusky kob	6.9	2221 (1 403-3 990)	7 349
		Umbrina sp.	Baardman	0.8	383 (242-688)	1 032
	Serranidae	*Epinephelus marginatus*	Yellowbelly rockcod	0.3	0	
	Soleidae	*Dagetichthys marginatus*	White margined sole	0.3	153 (97-275)	-
	Sparidae	*Cymatoceps nasutus*	Black musselcracker	0.2	0	
		Diplodus cervinus hottentotus	Zebra	1.5	230 (145-413)	244
		Diplodus sargus capensis	Blacktail	7.4	2526 (1 596-4 540)	1 858
		Lithognathus lithognathus	White steenbras	7.9	1 149 (726-2 064)	1 883
		Pachymetopon grande	Bronze bream	0.2	77 (48-138)	92
		Rhabdosargus globiceps	Cape stumpnose	1.7	383 (242-688)	124
		Sarpa salpa	Strepie	1.2	460 (290-826)	118
		Sparodon durbanensis	White musselcracker	1.0	306 (194-550)	1 791
	Tetraodontidae	*Amblyrhynchotes honckenii*	Evileye puffer	0.7	0	
	Triglidae	*Chelidonichthys*	Gurnard	0.2	0	
TOTAL				100	24 206	17 541

Note: - indicates where it was not possible to estimate weights

(e) Bag sizes and compliance

Only 25.4% of anglers succeeded in catching a fish, while only 17.8% of anglers captured an edible fish and 11.1% of anglers retained at least one fish. No anglers retained more than the daily total bag limit of 10 fish per person and species bag limits were only exceeded on four occasions, once for kob and white steenbras, and on two occasions for elf.

Non-compliance with MLS, however, was noted for more than 33% of the retained fish, for which species size limits are available. Highest levels of non-compliance with the MLS occurred at Sundays Surf where 70% of the retained catch was under the MLS, while at Hougham Park 33% were below the MLS and 22% at both Boknes and Perdevlei (Figure 5.7a). Non-compliance with MLS was greatest for white steenbras, with 80% of the retained fish being below the MLS, followed by Cape stumpnose, dusky kob and zebra, where 40%, 38% and 33% of the retained fish were below the MLS respectively (Figure 5.7b).



(a) (b)

Figure 5.7. Percentage non-compliance with minimum legal size limits (MLS) per (a) access point and (b) per species. (Wst=white steenbras; CS=Cape stumpnose; Dkob=dusky kob; Ze=zebra; Wmcr=white musselcracker; BT=blacktail.

(f) Bait utilisation

A total of 23 bait types were identified during the creel surveys representing several functional groups (Table 5.5). Sardine (*Sardinops sagax*) and chokka-squid (*Loligo sp.*) were the most common bait types utilised with 73.7% and 60.1% of anglers respectively having them in their possession. Both are common commercial baits and are readily available in retail outlets and all anglers interviewed had purchased these bait types prior to commencing fishing. The mean quantity of sardine per angler was 1.47±1.31 kg with an average price of ZAR15.37±11.24 per kilogram, while anglers averaged 0.64±0.48 kg of chokka-squid at a price of ZAR46.42±30.52 per kilogram. Sand prawn (*Callianassa kraussi*) (12.1%), red bait (*Pyura stolonifera*) (11.0%) and sand mussels (*Donax serra*) (9.1%) were utilised by anglers to a lesser degree and were bought infrequently with most anglers collecting themselves. Pink prawn (*Penaeidae sp.*) was only utilised by 6.2% of the anglers but was the most expensive bait at ZAR85.75±38.31 per kilogram. Baits less commonly utilised included mussels (*Perna/Mytilus sp.*), alikreukel (*Turbo sarmaticus*), saddleback (*Dinoplax gigas*), abalone/venus ear (*Haliotis sp.*), bloodworm (*Arenicola loveni*) and wonder worm (*Eunice aphroditois*) and were all collected by the anglers themselves within the area they were fishing.

Sardine and chokka-squid were utilised at all access points throughout the study area, together comprising between 32-87% of the bait species utilised at each access point (Figure 5.8). A large proportion of the anglers entering the fishery at Sundays Surf (41%) and Perdevlei (24%), and to a lesser degree at Perdekloof (7%) and Sundays River (6%), had collected sand mussel in the area where they were fishing. Sand prawn (collected outside of the study area) and Cape reef-worm (collected inside study area) both contributed 16% to the bait type utilised by anglers at the Perdevlei access point. These soft baits are preferred by certain species, including white steenbras, spotted grunter and baardman (*Umbrina sp.*) and are typically used to target them in these areas. Red bait was utilised by between 3 and 13% of the anglers at Perdevlei, Perdekloof, Pump Station, Cannon Rocks and Boknes, being a popular bait for targeting reef associated species. Mullet were collected and used as live baits by 3% and 7% of anglers at Hougham Park and Sundays River respectively, indicating the targeting of large piscivorous species in these areas.

Table 5.5. Bait utilisation in the recreational shore fishery (quantity and cost presented as mean ± standard deviation, units are numbers of individuals unless indicated otherwise).

Bait group	Scientific name	Common name	% of anglers utilising bait type	Quantity per angler	% of times purchased	Cost per unit
Polychaeta	*Arenicola loveni*	Bloodworm	0.8	6±3.61	0.0	-
	Gunnarea capensis	Cape reef-worm	3.5	19.75±17.64	7.7	R4
	Eunice aphroditois	Wonder worm	0.5	3.00	0.0	-
Crustacea	*Panulirus homarus*	Crayfish	0.3	-	100.0	-
	Scylla serrata	Mud crab	0.3	1.00	0.0	-
	Upogebia africana	Mud prawn	3.2	31.02±29.1	41.7	ZAR0.86±0.80
	Penaeidae sp.	Pink prawn	6.2	0.35±0.13 kg	100.0	ZAR85.75±38.31/kg
	Callianassa kraussi	Sand prawn	12.1	44.2±40.81	9.3	0.66±0.29
Bivalvia	*Perna / Mytilus sp.*	Mussel	0.3	-	0.0	-
	Donax serra	Sand mussel	9.1	21.5±17.17	2.9	-
Cephalopoda	*Loligo sp.*	Chokka-squid	60.1	0.64±0.48 kg	98.6	ZAR46.42±30.52/kg
	Sepia vermiculata	Cuttlefish	0.5	0.75±0.35	0.0	-
	Octopus vulgaris	Octopus	5.1	0.83±0.29	5.3	-
Gastropoda	*Haliotis sp.*	Abalone	0.3	-	0	-
	Turbo sarmaticus	Alikreukel	0.8	3.5±3.91	0.0	-
Polyplacophora	*Dinoplax gigas*	Saddleback	4.8	8.17±6.71	0.0	-
Teleostei	*Rhabdosargus holubi*	Cape stumpnose	1.1	0.67±0.47	0.0	-
	Scomber japonicus	Mackerel	1.3	3.31±2.56	100.0	R5
	Mugilid species	Mugilidae sp.	4.0	11.07±11.84	0.0	-
	Sardinops sagax	Sardine	73.7	1.47±1.31 kg	100.0	ZAR15.37±11.24/kg
	Sarpa salpa	Strepie	2.4	1.5±0.71	11.1	-
Tunicata	*Pyura stolonifera*	Red bait	11.0	4.13±3.27	7.5	-
Artificial	Artificial lures	Artificial lures	1.6	-	100.0	-

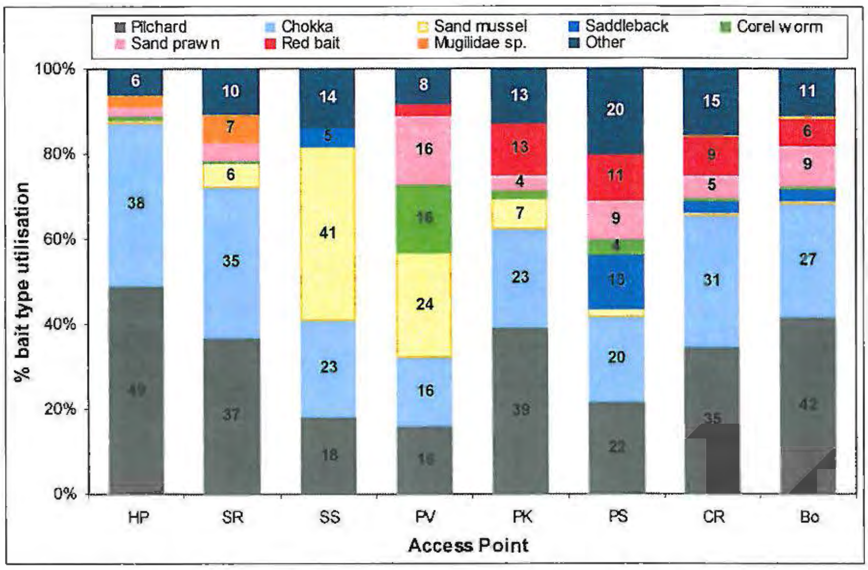


Figure 5.8. Spatial utilisation of bait species by recreational shore anglers.

(g) Economics

Recreational shore anglers fished for 47±66 (n=77; $\bar{x}$±sd) days per year and spent ZAR189±190[7] per angling day for bait (ZAR32±57), tackle (ZAR51±105), travel expenses (ZAR75±108), food (ZAR43±64) and ghillies[8] (ZAR4±25). Furthermore anglers indicated that they were willing to pay ZAR247±202 per day's angling trip and had spent on average ZAR282±483 within the month prior to the interview on fishing tackle. The average value of anglers' equipment which they had present with them at the fishing location was ZAR2 593±2 026. The annual direct economic value of the recreational shore fishery was estimated at ZAR10.3 million.

5.3.2 Skiboat fishery

A total of 3 040 launch records spanning the period from June 2006 to May 2009 were obtained from PEDSAC with the logbook data accounting for 52% of all days over this period. A total of 171 and 163 effort counts were conducted between May 2007 and April 2008 at Kenton and Boknes launch sites, representing 47% and 45% of possible launch days respectively. A total of 181 interviews were conducted between the three launch sites.

(a) Factors influencing recreational skiboat effort and catch

(i) Launching effort

The number of recreational vessels launching per day was influenced significantly by the launch site ($p<0.001$), month ($p<0.001$), day type ($p<0.001$) and wind speed ($p<0.001$) (Table 5.6). Launching effort was significantly different at each site with greatest effort at PEDSAC (4.7$\bar{x}$=, CI:4.3-5.2), followed by Kenton ($\bar{x}$=1.1, CI:0.9-1.5) and Boknes ($\bar{x}$=0.4, CI:0.3-0.6) (Figure 5.9a). Seasonally, launching effort was greatest during the summer months, declining through autumn and winter to lowest effort at the onset of spring. Highest launching effort was recorded in December ($\bar{x}$=2.6, CI:2.1-3.2) and lowest in August ($\bar{x}$=0.7, CI:0.4-1.0) (Figure 5.9b). Launching effort was over three times higher during non-work ($\bar{x}$=2.4, CI:2.0-2.9) than work days ($\bar{x}$=0.7, CI:0.57-0.86) (Figure 5.9c). Launching effort decreased with increasing wind speed.

(ii) Crew size

Crew size (number of anglers per vessel) was not influenced significantly by any factors (Table 5.6).

(iii) Fishing duration

The fishing duration of recreational skiboat anglers was influenced significantly by launch site ($p<0.001$) and wind speed ($p=0.028$) (Table 5.6). Mean fishing duration was significantly longer at PEDSAC ($\bar{x}$=7.6, CI:7.2-8.1) than both Boknes ($\bar{x}$=4.3, CI:3.6-5.0) and Kenton ($\bar{x}$=5.1, CI:4.7-5.4) (Figure 5.9d) and decreased with increasing wind speed.

[7] Mean daily expenditure does not equal the sum of all components as it was estimated independently by the respondent
[8] People paid to carrying equipment or assist with bait collection

(iv) CPUE

Launch site was the only factor which influenced CPUE significantly ($p<0.001$) (Table 5.6). Mean CPUE (fish.angler-hour^{-1}) at both Boknes ($\bar{x}=0.57$, CI:0.36-0.90) and Kenton ($\bar{x}=0.47$, CI:0.33-0.67) was significantly greater than at PEDSAC ($\bar{x}=0.21$, CI:0.16-0.30) (Figure 5.9e).

Table 5.6. Main effects influencing the number of vessels launching per day, mean fishing duration and CPUE of the recreational skiboat fishery in Algoa Bay. Cells shaded in grey indicate factors which were excluded from the GLM analyses based on AIC criterion and those in black indicate factors which were irrelevant to the model. Cells highlighted in green and orange represent significant differences at p<0.05 and p<0.001 respectively.

Factor	Intercept	Launch site	Year	Month/Season	Day type	Wind
Vessel number	p<0.001** WaldX^2=85.61	p<0.001** WaldX^2=155.38	p=0.056 ns WaldX^2=5.76	p<0.001** WaldX^2=120.60	p<0.001** WaldX^2=303.99	p<0.001** WaldX^2=95.46
Crew size	p<0.001** WaldX^2=812.64	p=0.215 ns WaldX^2=3.08	N/A	N/A	N/A	
Fishing duration	p<0.001** WaldX^2=399.71	p<0.001** WaldX^2=99.07	N/A	N/A	N/A	p=0.028* WaldX^2=4.84
CPUE	p<0.001** WaldX^2=174.16	p<0.001** WaldX^2=88.01	N/A	N/A		N/A

N/A=not applicable, discarded after AIC analysis
ns=not significant
* p<0.05
** p<0.001

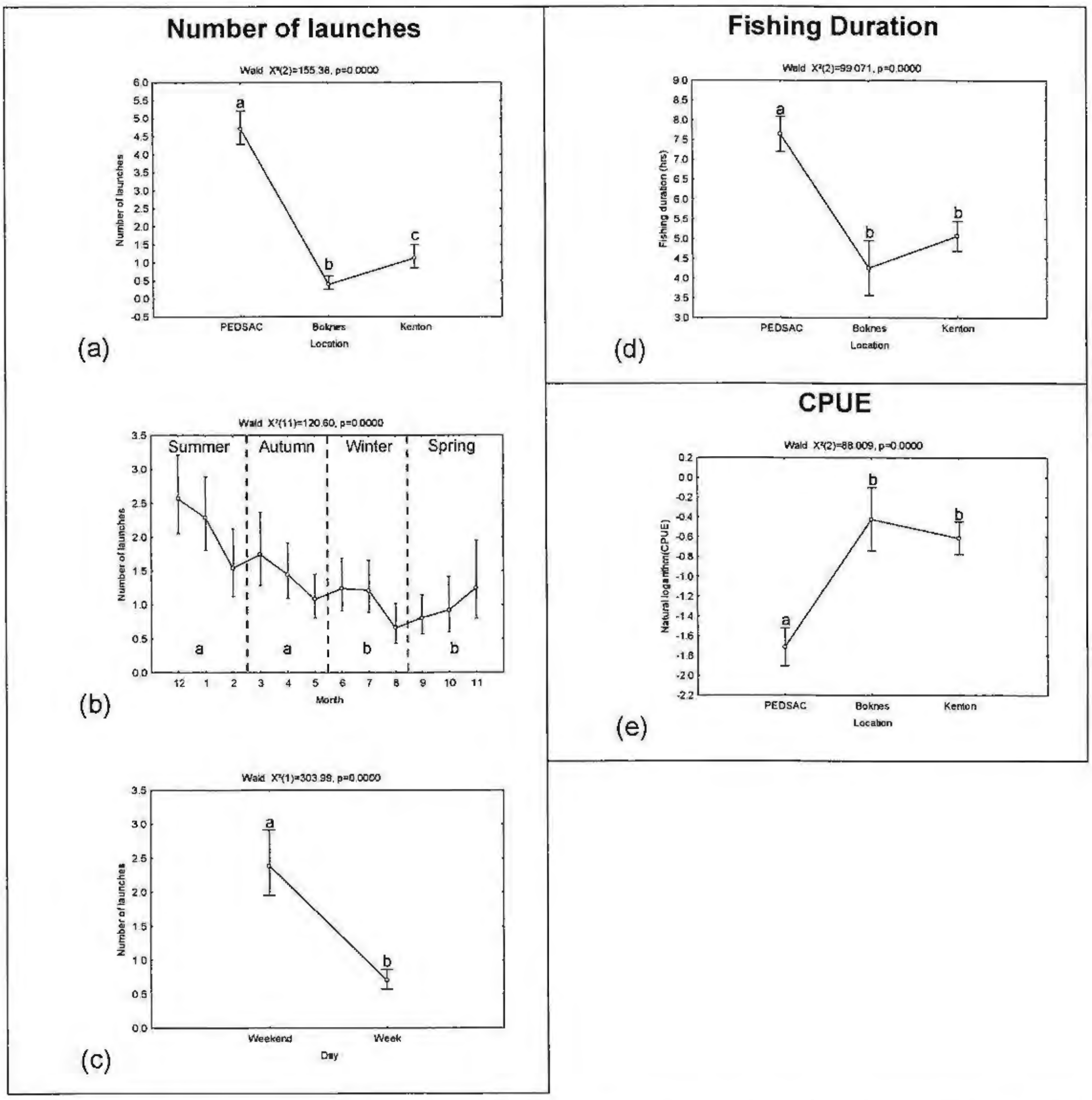


Figure 5.9. Main factors influencing the number of vessels launching (left), fishing duration (top right) and CPUE (middle right) of the recreational skiboat fishery in Algoa Bay. Letters above error bars denote significant differences determined through post hoc tests.

(b) Estimation of annual effort and catch

Annual recreational skiboat fishing effort within Algoa Bay was estimated at 2 118 (CI: 1 925-2 333) boat-days with the PEDSAC launch site accounting for approximately 75% of the annual effort (Table 5.7). The mean fishing duration was lowest at Boknes (4.3hrs) and highest at PEDSAC (7.4hrs) and crew size ranged from 3.5 to 4.3 at Boknes and PEDSAC respectively, but did not differ significantly. This resulted in PEDSAC accounting for over 80% of the effort in angler-hours, with Boknes only accounting for 4%. CPUE, however, was highest at Boknes (0.83 fish.angler-hour^{-1}) followed by Kenton (0.70 fish.angler-hour^{-1}) and PEDSAC having the lowest catch rate (0.27 fish.angler-hour^{-1}). The estimated number of retained fish was highest at the PEDSAC launch site (13 869, CI:9 966-18 659) followed by Kenton (5 392, CI:3 539-7 671) and Boknes (1 792, CI:965-2 861) with an overall estimate of 20 873 (CI:16 438-26 429) fish harvested from Algoa Bay by the recreational skiboat fishery annually. Estimated annual harvest ranged from 3 530 to 38 806kg at the Boknes and PEDSAC launch sites respectively totalling 52 113kg for Algoa Bay. The contribution of each species to annual harvest is presented in Table 5.8.

Fishing effort was concentrated in the western sector of Algoa Bay, with greatest levels of estimated annual effort at Cape Recife point (561 boat-days; 27%) and at the Goodsheds (467 boat-days; 22%) (Figure 5.10). An estimated 212 (10%) boat-days of fishing effort occurred in the St Croix area between Coega Harbour and Sundays River mouth, with lower levels of effort occurring offshore at the Riy Banks (153 boat-days; 7%) and the South-west Grounds (75 boat-days; 4%). Only low levels of fishing effort occurred in the nearshore areas between Sundays River mouth and Cannon Rocks (<4% per area). Fishing effort was lower on the eastern side of the Bay with highest effort around Kenton (174; 8%) with the adjacent areas experiencing less than 75 boat-days per year of fishing effort.

Table 5.7. Annual estimates of catch and effort for the recreational skiboat fishery in Algoa Bay. (Mean with 95% confidence limits below).

Estimate	PEDSAC	Boknes	Kenton	Algoa Bay
Effort (boat-days)	1 573 (1 435-1 719)	147 (90-211)	397 (274-543)	2 118 (1 925-2 333)
Mean fishing duration (hrs)	7.6 (7.2-8.1)	4.3 (3.7-4.9)	5.1 (4.8-5.4)	-
Mean crew size (n)	4.3 (3.9-4.8)	3.5 (3.1-3.7)	3.8 (3.6-4.1)	-
Effort (angler-hours)	51 149 (43 564-59 673)	2 171 (1 256-3 239)	7 754 (5 190-10 703)	61 074 (53 309-69 717)
Mean CPUE (fish.angler-hour^{-1})	0.27 (0.20-0.35)	0.83 (0.61-1.09)	0.70 (0.59-0.82)	-
Retained catch (no. of fish)	13 689 (9 966-18 659)	1 792 (965-2 961)	5 392 (3 539-7 671)	20 873 (16 438-26 429)
Estimated weight (kg)	38 806	3 530	9 632	51 968

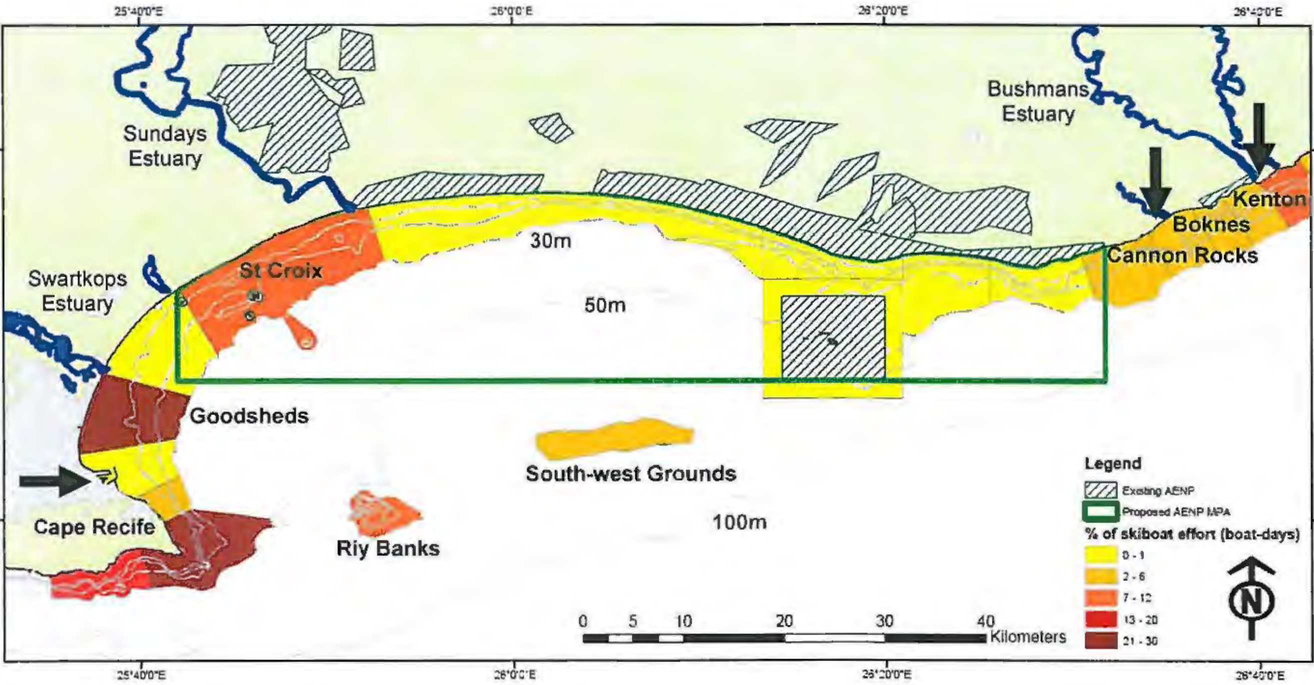


Figure 5.10. Spatial distribution of recreational skiboat fishing effort in Algoa Bay. Arrows indicate location of launch sites assessed in this study.

(c) Catch composition

Overall 26 species were landed and retained by the recreational skiboat fishery in Algoa Bay (Table 5.8) with catch composition differing spatially. Santer (*Cheimerius nufar*) accounted for the greatest proportion of the overall standardised catch for Algoa Bay (all three sites) (37%), and dominated catches at the Boknes (51%) and Kenton (36%) launch sites (Figure 5.11). However, santer only contributed 24% to the landed catch at PEDSAC, with geelbek (*Atractoscion aequidens*) dominating the catches (41%) at this launch site, while comprising small proportions of the catch at Boknes (10%) and Kenton (17%). Silver kob (*Argyrosomus inodorus*) also contributed substantially to the catch, ranging from 8% at Boknes to 25% of the catch by number at Kenton. Roman (*Chrysoblephus laticeps*) was landed at all launch sites, accounting for between 6 and 12% of the catch. Dageraad (*Chrysoblephus cristiceps*) and black musselcracker (*Cymatoceps nasutus*) were landed predominantly in the eastern region of Algoa Bay, contributing 8 and 4%, and 4 and 2% at the Boknes and Kenton launch sites respectively. Elf contributed 8% to the catch at PEDSAC.

(d) Bag sizes and compliance

Compliance with the overall daily bag size of 10 $fish.person.day^{-1}$ was high with only one vessel inspection revealing catches above the maximum daily catch limit per person. Compliance with species bag limits was lower, being exceeded on eight occasions (5%). Similarly 5% of the fish measured were below MLS. Due to low sample sizes for certain species the proportion of undersized fish landed for these species was high (Table 5.8).

Table 5.8. Species composition expressed as percentage of total retained catch by number, mean length per species (±standard deviation), percentage of each species below the minimum legal size (MLS) and estimated annual number caught (95% confidence intervals) per species within Algoa Bay by the recreational skiboat fishery. Species contributing greater than 5% and 10% to the retained catch highlighted in green and orange respectively.

Family	Scientific name	Common name	Proportion of retained catch (%)	Mean length (mm FL)	Proportion below MLS (%)	Est. number retained annually	Est. annual weight (kg)
Ariidae	*Galeichthys feliceps*	White seacatfish	0.1	-	N/A	46	16
Carangidae	*Seriola lalandi*	Giant yellowtail	0.1	605±35	N/A	22	80
Gadidae	*Merluccius capensis*	Shallow-water hake	1.1	539±77	N/A	109	136
Pomatomidae	*Pomatomus saltatrix*	Elf	1.8	366±61	0	560	719
Sciaenidae	*Atractoscion aequidens*	Geelbek	18.9	779±151	5	4 662	29 331
	Argyrosomus inodorus	Silver kob	20.9	626±116	6	3 368	8 450
Scombridae	*Sarda orientalis*	Striped bonito	0.3	576±48	N/A	96	460
Serranidae	*Epinephelus andersoni*	Catface rockcod	0.2	547±90	50	53	74
	Epinephelus marginatus	Yellowbelly rockcod	0.3	560±14	100	35	147
Sparidae	*Cymatoceps nasutus*	Black musselcracker	2.2	550±67	0	457	927
	Diplodus sargus capensis	Blacktail	0.1	322±2	0	46	39
	Pachymetopon aeneum	Blue hottentot	1.1	319±35	N/A	181	67
	Pachymetopon grande	Bronze bream	0.1	350	0	6	6
	Argyrozona argyrozona	Carpenter	2.7	378±89	24	428	425
	Chrysoblephus cristiceps	Dageraad	3.7	379±58	44	812	770
	Chrysoblephus anglicus	Englishman	0.1	390	0	6	8
	Boopsoidea inornata	Fransmadam	0.1	-	N/A	13	2
	Argyrops spinifer	King soldierbream	0.1	-	N/A	37	9
	Pterogymnus laniarius	Panga	2.0	339±44	N/A	223	146
	Petrus rupestris	Red steenbras	0.1	500	100	6	13
	Chrysoblephus gibbiceps	Red stumpnose	0.4	482±41	0	116	140
	Chrysoblephus laticeps	Roman	6.6	363±41	0	1 712	2 121
	Cheimerius nufar	Santer	36.3	380±59	0	7 721	7 601
	Polysteganus praeorbitalis	Scotsman	0.6	559±86	0	138	410
	Diplodus cervinus hottentotus	Zebra	0.1	-	N/A	6	4
Triglidae	*Chelidonichthys capensis*	Cape gurnard	0.1	505±148	N/A	13	10

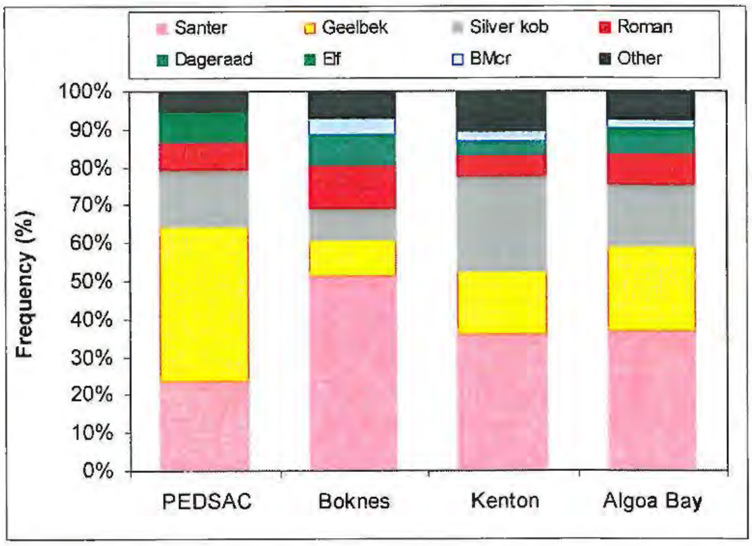


Figure 5.11. Contribution of the dominant species to the catch at each launch site and the standardised contribution to the Algoa Bay catch.

(e) Bait utilisation

The majority of recreational skiboat anglers purchased commercially available baits, with 87% of anglers utilising sardine, 51% chokka-squid and 15% artificial lures. A further 21% of respondents indicated that they utilised freshly captured fish as fillet or live bait, which included several small sparids and elf. Mackerel, octopus and pink prawn were used by few anglers.

(f) Economics

Recreational skiboat anglers fished for 19±17 days annually and their expenditure on the day of interview was ZAR769±606[9]. Boat fuel contributed the most to the overall cost (ZAR524±447), with bait (ZAR119±84), tackle (ZAR81±78) and food (ZAR89±71) accounting for a smaller portion. Skiboat anglers indicated that they were willing to spend ZAR1 512±2 309 per day on skiboat fishing activities, twice the value which they indicated they had spent on the day of interview. Investment into vessels and offshore fishing equipment was considerable at ZAR217 268±196 560. The direct economic value of the recreational skiboat fishery in Algoa Bay was estimated at approximately ZAR1.6 million per annum.

5.3.3 Index of relative recreational importance (IRRI)

The IRRI provides a useful tool to evaluate the spatial distribution of fishing effort and determine the social significance of certain areas to the recreational fishery. Recreational fishing effort was spatially heterogeneous across Algoa Bay with areas of greatest importance located along the sheltered western section of Algoa Bay in close proximity to the city of Port Elizabeth (Figure 5.12). The nearshore areas were of greater importance due to the higher levels of participation in the shore fishery and the overlap between the shore and skiboat sectors in close proximity to the coastline. The central shoreline and offshore region of Algoa Bay were of low importance to the recreational fishery due to the lower population densities in this area, fewer access points and greater travel distances, time and costs to access these regions of the study area. There were, however, some important areas along the remote section of coastline as a result of shore access points. The recreational importance of the shoreline increased towards the eastern sector of the study area due to the locality of more access points and launch sites and the small resident populations in the coastal towns of Cannon Rocks, Boknes and Bushmans River Mouth. The IRRI provides easy means to visualise the data and can be used to incorporate spatial recreational fisheries data in future spatial planning, and for designing monitoring and enforcement strategies to target high use areas.

The economic IRRI indicates a similar heterogeneous distribution of areas of economic importance with key areas situated on the western and eastern ends of the study area (Figure 5.13). The economic IRRI further highlights the economic importance of the coastal based fishing areas adjacent to urban and residential areas. This is due to the greater estimated economic value of the recreational shore fishery as a result of higher levels of participation.

[9] Mean daily expenditure does not equal the sum of all components as it was estimated independently by the respondent

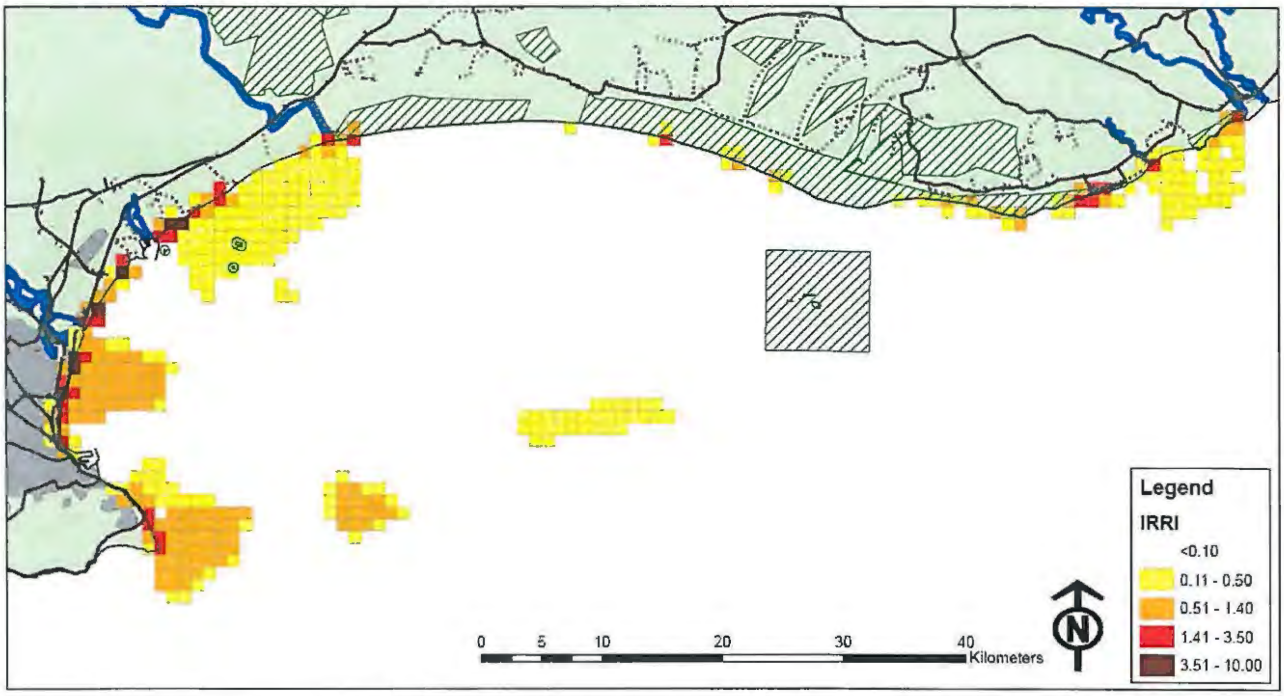


Figure 5.12. Spatial representation of cumulative recreational fishing effort in Algoa Bay based on an Index of Relative Recreational Importance (IRRI). Hashed green areas indicate existing AENP boundaries. Grey lines and dashed grey lines indicate main access roads and tracks respectively.

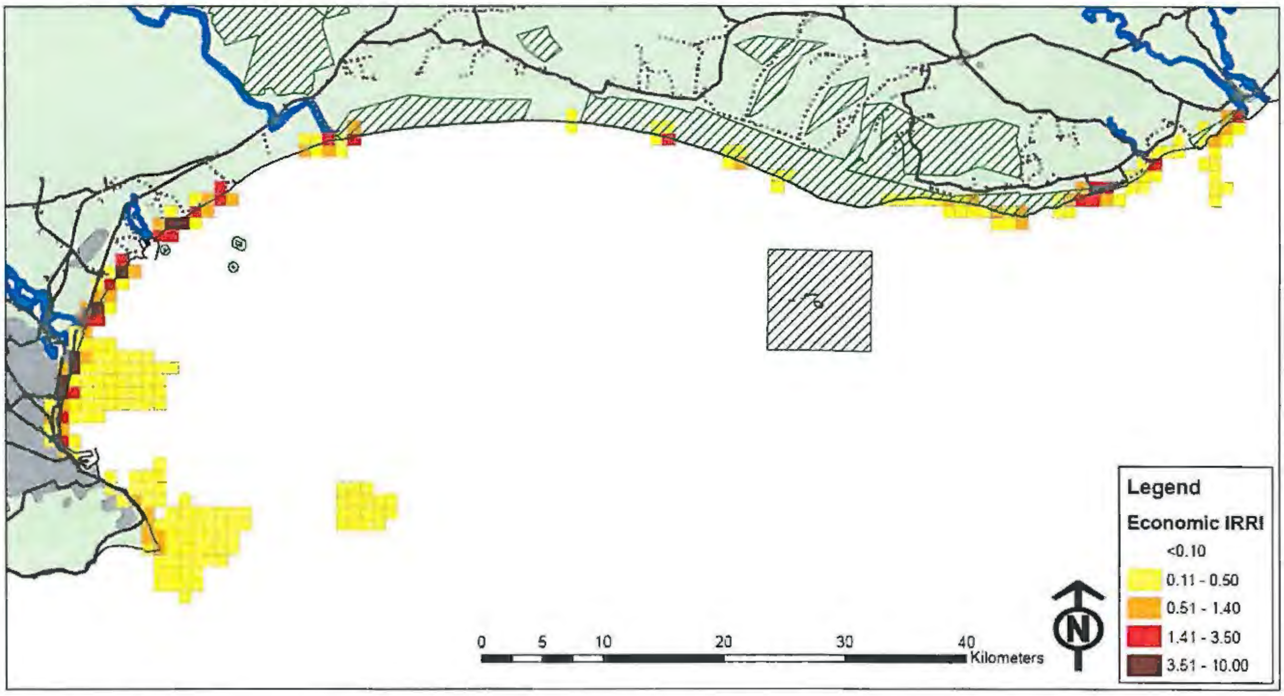


Figure 5.13. Spatial representation of cumulative recreational fishing effort in Algoa Bay based on an Economic Index of Relative Recreational Importance (Economic IRRI). Hashed green areas indicate existing AENP boundaries. Grey lines and dashed grey lines indicate main access roads and tracks respectively.

Summary of key findings

Recreational shore fishery

- Fishing effort was spatially heterogeneous and largely determined by the location of access points
- Effort was concentrated in areas in close proximity to vehicular access points
- Retained catch rates were low
- There was large spatial and temporal variability in catch rate and catch composition
- The direct economic value of the shore fishery was high

Recreational skiboat fishery

- The location of launch sites played a major role in determining the distribution of skiboat fishing effort and fishing trip duration
- Launching effort was strongly influenced by season and type of day (work/non-work)
- Catch rate and catch composition varied spatially
- The estimated annual harvest was significant (approximately 25% of commercial linefish landings)
- The direct economic value of the skiboat sector was lower than the shore fishery, but the investment into equipment was significantly greater
- The IRRI and economic IRRI are useful spatial indices for integrating recreational fisheries data into future spatial planning

5.4 Discussion

5.4.1 Recreational shore fishery

(a) Fishing effort

Spatial trends were apparent with a non-uniform distribution of anglers and angling effort within the study area, which can be ascribed to travel distances to and the accessibility of public access points. Higher angler numbers occurred adjacent to urban areas and at access points which provided easy vehicular access to fishing sites and opportunities for longer duration overnight trips. Easily accessible fishing sites were utilised by 3-4 times the number of anglers than those more difficult to access, and accounted for 80% of the overall fishing effort. Similar concentration of recreational shore anglers and angling effort around urban and peri-urban areas has been reported in previous studies in South Africa (Els and McLachlan 1990; Brouwer *et al.* 1997; Brouwer 1997; Mackenzie 2005; King 2005), Australia (Lynch 2006; Smallwood *et al.* 2006), Canada (Post *et al.* 2002) and Portugal (Rangel and Erzini 2007).

Angler density from roving creel surveys conducted between Coega and Boknes, a fairly remote and inaccessible stretch of coastline, was similar to previous estimates for the Eastern Cape in the mid 1990s (Brouwer *et al.* 1997; Brouwer 1997; Brouwer and Buxton 2002) (Table 5.9). However, angler density from aerial surveys along the whole Algoa Bay coastline (Cape Recife to Bushmans River Mouth), including urban, peri-urban and remote areas, was higher than the mean estimate for the

Eastern Cape in the mid 1990s (Brouwer *et al.* 1997) suggesting an increase in angler participation in the recreational shore fishery between the two study periods. Furthermore the study in the mid 1990s was conducted prior to the ban of ORVs in the coastal zone (DEAT 2001). This ban has restricted access to remote stretches of the coastline where public infrastructure is limited. The ban is likely to have further influenced the distribution of anglers during the current study due to the lengthy coastline with limited number of access points. The increase in angler participation along other regions of the South African coastline, which offers greater opportunities to access to the shoreline, is therefore likely to have been more pronounced than observed in this study. This was shown by Mackenzie (2005) who illustrated a shift in angler effort from remote areas to more easily accessible areas following the introduction of the ORV ban leading to increased effort around access points.

Previous studies estimated annual growth rates of 5.5% and 6.5% in the Eastern Cape (Clarke and Buxton 1989) and KwaZulu-Natal (van der Elst 1986) recreational shore linefisheries respectively and more recently McGrath *et al.* (1997) estimated a compound annual growth rate of 2% for the South African shore linefishery. It is somewhat surprising that angler density estimates in the current study were far lower than those reported by Mackenzie (2005). However, this study has shown that accessibility and the proximity of urban settlement areas play an important role in regulating the distribution of shore anglers, hence confirming the findings of previous authors (Clarke and Buxton 1989; Brouwer *et al.* 1997; Mann *et al.* 2003; Mackenzie 2005). The remoteness of the study location and poor accessibility and public infrastructure are therefore likely to have contributed to the lower density estimates obtained in the current study.

Angler density between Coega and Boknes in the current study was similar to that found by King (2005) in the Plettenberg Bay region of the Western Cape, and Mann *et al.* (2003) in the Transkei region of the Eastern Cape (Table 5.9). All study areas have similar site characteristics with small coastal settlements and large stretches of remote and inaccessible coastline, accounting for the lower angler densities than in the urban regions of Port Elizabeth (Clarke and Buxton 1989; Brouwer 1997) and urban and peri-urban areas of the Western Cape (Attwood and Farquhar 1999). The variability in site characteristics and angler densities between different areas of the coastline highlights the importance of and need for detailed baseline studies over spatial scales relative to management requirements, as well as for accurately assessing long-term changes in recreational fishing patterns.

Table 5.9. Angler density estimates from previous studies of recreational shore angling in South Africa.

Angler Density (anglers.km^{-2})	Author	Survey type	Study area
1.2	Clarke and Buxton 1989	Roving creel	Port Elizabeth, Eastern Cape
1.3	Brouwer 1997	Roving creel	Port Elizabeth, Eastern Cape
0.39	Brouwer *et al.* 1997	Roving creel	Eastern Cape where access was possible
0.36	Brouwer *et al.* 1997	Aerial	Eastern Cape
0.79	Mann *et al.* 2003	Aerial	Transkei
0.66	King 2005	Roving creel	Plettenberg Bay area, Western Cape
1.42	Attwood and Farquhar 1999	Roving creel	Cape Hangklip-Walker Bay, Western Cape
2.9	Mackenzie 2005	Roving Creel	Eastern Cape
0.48	**This study**	**Roving creel**	**Coega to Boknes, Eastern Cape**
0.67	**This study**	**Aerial survey**	**Coega to Boknes, Eastern Cape**
1.01	**This study**	**Aerial survey**	**Algoa Bay (Cape Recife to Bushmans River Mouth), Eastern Cape**

Although accessibility was shown to play an important role in determining angler distribution in the current study, the influence of substrate type may also have been an important factor, as fishing techniques differ between rock and surf (sandy beach) habitats. Rocky areas provide relatively stable habitats for resident reef associated species and shore anglers may therefore return to the same fishing sites to target these species. However, sandy beaches are highly dynamic with sandbank formations changing rapidly, influencing the distribution of the targeted species in these areas. Anglers cannot therefore return to the same known fishing locations along sandy beaches and are required to traverse large stretches of coastline in order to identify suitable fishing sites on each outing. In the past this was easily undertaken using ORVs, but the change in regulations has restricted anglers to public vehicular access points and to within walking distances from these sites, with only few anglers traversing long distances on foot. The majority (60%) of the study area in which roving creel surveys were conducted consisted of sandy beaches. Historically the Sundays Surf (Sundays River to Woody Cape) coastline was the most popular surf angling beach in the Eastern Cape (Els and McLachlan 1990) with 70% of club anglers fishing in this area. Furthermore 50% of anglers indicated that they preferred surf fishing over rock or offshore fishing (Els and McLachlan 1990). Prior to the ban of ORVs the mean density along the Sundays Surf was 0.24 vehicles.km^{-1} with a maximum density of 0.67 vehicles.km^{-1} (Els and McLachlan 1990) indicating the high level of utilisation along this stretch of coastline in comparison to the findings of the current study. The ban of ORVs has therefore led to a reduction in surf fishing effort and the concentration of anglers around vehicular access points. Despite the wide expanses of sandy beaches in the study area, the majority of anglers (63%) stayed within 1km of car parks in the current study providing evidence for this change in fishery characteristics. The predominance of sandy beaches and poor accessibility in the current study area is therefore a significant contributor to lower than anticipated angler density estimates, and the ban of ORVs has led to the concentration of fishing effort around vehicular access points which are typically situated along rocky stretches of coastlines and at estuary mouths.

Although angler density was lower in less accessible areas of the coastline, the duration of the fishing trip increased with distance from vehicular access points, and was longer at more remote locations, with the exception of Hougham Park as it was used as a camping location by several angling groups. The mean fishing trip duration in the current study was 7.6 hours compared to 5 hours previously reported for the Eastern Cape (Brouwer and Buxton 2002). This is due to the time and effort invested by anglers in order to access more remote sites on foot and can be attributed to the remoteness of large stretches of the coastline in the current study. Mackenzie (2005) reported a 16% reduction in fishing effort along the Algoa Bay coastline following the implementation of the ORV ban. In addition anglers were encountered in closer proximity to vehicular access points (mean distance prior to ban 7.2±11.6km; after ban 2.7±2.5 km) after the ORV ban was implemented (Mackenzie 2005).

Temporal trends were also apparent with higher angler numbers during non-work days and over the summer months as has been reported in several studies along the south coast (Brouwer *et al.* 1997; Hanekom *et al.* 1997; King 2005). Higher effort in summer and autumn coincides with the main Christmas and Easter holiday periods in South Africa as well as warmer weather conditions and longer day lengths providing better conditions for anglers wishing to enjoy the general fishing and outdoor experience. The peak in effort during summer also coincided with an increase in the proportion of non-local resident anglers, indicating the importance of recreational anglers from further afield during peak holiday periods. Similar trends in the recreational shore fishery in South Africa have been observed by Brouwer *et al.* (1997) in the Port Elizabeth area and King (2005) in the Plettenberg Bay fishery. Mann *et al.* (2003), however, found higher levels of effort in the former Transkei over the spring and winter months which he attributed to the strong seasonal influence of weather and swell conditions in the region which limited fishing during the summer months. In the current study weather conditions were shown not to be a significant factor influencing fishing effort.

(b) Catch rate

A Delta model was employed in the current study due to the zero-inflation typical of recreational fisheries catch rate data. This allowed the probability of capture and the positive catch rates to be modelled separately and factors influencing each to be identified and taken into consideration in the calculation of harvest for the fishery. Several factors influencing the probability of capture and positive CPUE were identified and contributed to the high variability in the adjusted catch rate for the recreational shore fishery. Geographical location was the most important aspect influencing catch rate, with significant variations in CPUE between the different access points. Furthermore walking distance from the main access points also influenced the CPUE significantly. Although this may suggest localised depletion of stocks around popular fishing sites which are more heavily fished, high variability in catch rate is common in many fisheries, although causes of variation are little understood (Hilborn 1985). In recreational fisheries the angler's success in catching a fish has been ascribed to the individual's skill and fishing experience as well as their local knowledge of the area (Brouwer and Buxton 2002). Furthermore the general aim (competitive vs. general enjoyment) varies considerably between users. As a result most of the recreational catch can be attributed to a few highly skilled and dedicated anglers, with many anglers landing no fish at all (Jones *et al.* 1995). These factors, which are difficult to quantify, may therefore have a more significant effect on the catch rate than spatial effects alone, complicating the interpretation of CPUE data in recreational fisheries.

The catch rate of competitive and experienced club anglers is two to six times higher than that of non-club anglers and they typically target and land larger fish than non-club anglers (Clarke and Buxton 1989; Brouwer 1997). In the current study the percentage of club anglers was highest at the Pump Station, Perdevlei and the Sundays River access points (Table 5.10), where catch rates were generally higher than at other sites, while lowest percentages of club anglers occurred at the Hougham Park, Boknes and Perdekloof access point, coinciding with the lowest catch rates. This suggests that club anglers travel to more remote locations and walk greater distances to access fishing sites where fishing quality is perceived to be better. It also suggests that angler experience may confound the influence of geographical location on estimation of catch rate. Angler experience and motivation is therefore likely to account for a considerable amount of the variability in the catch rate in recreational fisheries and the use of recreational catch rate data as an index of abundance to infer trends in resource status could therefore be misleading. However, due to the more consistent skill levels and similar competitive objectives among club anglers, competition and club records are likely to provide more accurate trends in CPUE. Such analyses in South Africa have confirmed a decrease in catch rate in the Border region of the Eastern Cape (Pradervand and Govender 2003; Pradervand *et al.* 2007) but no clear trend was apparent in KwaZulu-Natal or the Transkei (Pradervand 2004; Pradervand *et al.* 2007).

Table 5.10. Percentage of club anglers utilising each site.

Access Point	% club members
Hougham Park	17
Sundays River	54
Sundays Surf	33
Perdevlei	40
Perdekloof	13
Pump Station	56
Cannon Rocks	36
Boknes	17

The catch rate calculated using the Delta-X approach for the total and retained catch in the current study was 0.301 and 0.105 fish.angler-hour^{-1} respectively. This was slightly higher than CPUE estimates (0.281 and 0.082 fish.angler-hour^{-1} respectively) calculated using the mean of ratios method, which is often used in similar analyses. Nonetheless the estimates of retained catch rate (0.105 or 0.082 fish.angler-hour^{-1}) in this study are less than half those of previous studies conducted in the 1980s where catch rates (retained catch) of 0.288 and 0.218 fish.angler-hour^{-1} were reported in the Port Elizabeth (Clarke and Buxton 1989) and KwaZulu-Natal (Joubert 1981) recreational shore fisheries respectively. CPUE for the retained catch was also more than three times lower than estimated for the Plettenberg Bay shore linefishery (0.374 fish.angler-hour^{-1}) (King 2005).

Estimated retained catch rate per angler per day in the current study was 0.275 fish.angler^{-1}.day^{-1} approximately 8 times lower than that estimated for the Eastern Cape shore fishery in the mid 1990s (2.06 fish.angler^{-1}.day^{-1}) (Brouwer *et al.* 1997). However, the catch rate was similar to that reported for the shore fishery in Richards Bay (0.064 fish.angler^{-1}.hour^{-1}) (Beckley *et al.* 2008). These trends suggest both a significant temporal reduction in the catch rate of the recreational shore fishery in the Algoa Bay region, as well as spatial differences over large geographic areas along the South African coastline. Changes in fishery regulations, differences in the level of enforcement as well as improvement in fishing equipment influences spatial and temporal comparisons with earlier studies. Trends in catch composition and mean size may therefore be more effective for monitoring changes in the fishery.

(c) Harvest and catch composition

Although highly variable, the estimation of catch rate is important for the quantification of harvest in the recreational shore fishery. Harvest (retained catch) was estimated at 12 102 fish annually and varied spatially with estimates ranging from 264 to 2 884 fish per access point depending on the level of effort and catch rate at each site. This was far lower than the ±31 000 fish harvested annually in the Plettenberg Bay shore linefishery (King 2005) despite the greater effort in the current study (178 035 vs. 102 566 angler-hours).

Blacktail, dusky kob, elf, white seacatfish and white steenbras accounted for 74% of the catch retained. In some cases recreational harvest can exceed that of commercial sectors (Gartside *et al.* 1999) and recreational fisheries can therefore lead to overexploitation of stocks to the point of population collapse (McPhee *et al.* 2002; Schroeder and Love 2002; Cooke and Cowx 2004; Cooke and Cowx 2006; Aas and Schramm 2008; Arlinghaus and Cowx 2008). The decline of several linefish species in South Africa has been attributed directly to the recreational shore fishery (Griffiths and Lamberth 2002). As the recreational fishery is open access, effort regulation is difficult and the fishery is currently managed by output controls (bag and size limits). However, many of these restrictions have failed to limit recreational anglers' catches in the past (Attwood and Bennett 1995) and have therefore not contributed to a reduction in fishing mortality for many species. Review of the recreational regulations has indicated that for some species an 80% reduction in the bag limit is required to achieve a sustainable level of fishing mortality (Griffiths and Lamberth 2002). In order to facilitate the recovery of overexploited species and achieve targeted mortality rates, spatial or temporal closures may be the only effective measure to reduce the fishing effort in critical habitats (Attwood 2003). Long-term assessment of changes in the harvest and catch composition is therefore essential for the evaluation of management measures intended to reduce recreational catch.

Both the total and retained catch composition varied considerably between access points. Lesser guitar fish comprised large proportions of the total catch along the sandy beaches, with white steenbras and white seacatfish also abundant in these areas. Dusky kob was caught in most areas but was predominantly abundant in the surf zone along the sandy beaches and in close proximity to the estuary mouth. Blacktail was an important component of the retained catch, particularly in the eastern region of the study area, where inshore reef was more abundant.

The spatial variability in catch composition was largely due to differences in substrate type and species preferences for certain habitats. Knowledge of the spatial patterns in catch composition is therefore important for developing management recommendations in order to enhance the protection for specific species. Furthermore monitoring trends in catch composition is important for detecting declines in the relative proportion of species and may be more informative than monitoring catch rate for individual species due to the high variability in CPUE data. Changes in the catch composition have been noted in the South African shore fishery, particularly a decline in the proportion of reef associated species and an increase in the proportion of elasmobranchs (Brouwer *et al.* 1997; Brouwer and Buxton 2002). This is often due to the replacement of one targeted species or group by another in an overexploited fishery, resulting in serial overfishing and significant changes in the catch composition (Pauly *et al.* 1998). Even though overall harvest may remain constant, increasing angler effort and changes in the species composition of catches may mask declining catch rates. This should be re-assessed periodically by estimating the total catch of each species (Clarke and Buxton 1989) in order to evaluate the effectiveness of the current management measures in regulating fishing effort and therefore fishing mortality.

5.4.2 Recreational skiboat fishery

(a) Fishing effort

Offshore skiboat fishing is largely governed by three licensed vessel launching sites in Algoa Bay. These are PEDSAC, situated in the sheltered Port Elizabeth harbour in the western region, and two beach launch sites situated in the eastern region at the Boknes and Kenton coastal settlements. Two additional beach launch sites, Hobie and Kings beaches, are licensed in the Port Elizabeth metropolitan area; however, these are primarily utilised by smaller recreational sailing dinghies and jet skis and few fishing vessels launch from these sites. Smaller recreational fishing vessels (usually inflatables) also launch periodically through the Swartkops, Sundays and Kariega estuary mouths. The contribution to the overall fishing effort in Algoa Bay from these sites, however, is considered to be low and was not quantified in this study. A total of 375 recreational fishing vessels were identified between the three launch sites in Algoa Bay, which represents 34% of the 1 100 vessels recreational vessels estimated to participate in the Eastern Cape recreational fishery (Sauer *et al.* 1997; Brouwer and Buxton 2002; Donovan 2010). This represents a substantial proportion of the recreational vessels active within the Eastern Cape, but is not surprising as Port Elizabeth is the largest coastal city in the province and similar concentrations of recreational vessel fishing effort around urban centres has been reported along the Queensland (Webley *et al.* 2010) and west coasts of Australia (Sumner and Williamson 1999; Sumner *et al.* 2008).

Fishing effort was highest in the western sector of Algoa Bay, which is closest to the most densely populated area. The number of boat-days.year^{-1} originating from the PEDSAC launch site was approximately three times higher than the combined effort from the Boknes and Kenton launch sites in the eastern sector of Algoa Bay. This is due to the large resident population of skiboat anglers in Port Elizabeth and the presence of an active fishing club which holds regular competitions. Contrarily Kenton and Boknes have small resident populations and few recreational skiboat anglers who are active throughout the year. Furthermore PEDSAC accounted for approximately 85% of the annual fishing effort in angler-hours. This can be attributed to the longer fishing durations of vessels launching from PEDSAC and larger vessel sizes (8.3±1.9) than those from Boknes (5.1±1.2) and Kenton (5.6±1.5), hence having larger crew sizes (PEDSAC 4.3; Boknes 3.5; Kenton 3.8). These differences are driven by the local characteristics of each launch site.

The PEDSAC club has formal mooring facilities in the Port Elizabeth harbour accommodating larger vessels (>8m) which cannot be trailered easily, as well as having a protected slipway facilitating the launching and retrieval of larger vessels. However, neither the Boknes nor Kenton launch sites, although legally licensed, have a slipway and vessels are launched and retrieved from the beach using ORVs and modified breakneck trailers. Furthermore, vessels utilising these sites are required to launch through the surf zone and therefore need to be particularly manoeuvrable to navigate the waves safely. These factors account for the smaller size of vessels fishing in the eastern sector and hence contribute to the shorter fishing trip durations and lower overall effort due to smaller crew sizes.

Fishing effort was influenced by wind speed, with fewer vessels launching and shorter fishing trip durations occurring during windier conditions. In the Eastern Cape both swell height and direction have been shown to be correlated to wind speed (Donovan 2010) which therefore serves as a proxy for general weather and fishing conditions. Strong winds and rough sea conditions are common in the Eastern Cape limiting the number of seagoing days for small recreational craft (Smale and Buxton 1985; Donovan 2010). Weather conditions may have contributed to the spatial disparity in fishing effort as the western sector of Algoa Bay is more sheltered from the dominant westerly winds and swell therefore allowing greater opportunity for vessels to put to sea. The harbour breakwaters allows safe launching in most sea conditions while the beach launch sites at Boknes and Kenton are unprotected and exposed to large surf conditions following strong westerly winds therefore preventing vessels from launching under such conditions. Furthermore, larger vessels allow anglers to fish safely and more comfortably in rougher sea conditions thereby allowing anglers from PEDSAC to fish more regularly and for longer periods during windy conditions which typically arise and then persist from mid-morning in the Eastern Cape. Fishing trip duration of recreational vessels at Boknes (4.3hrs) and Kenton (5.1hrs) was similar to recent findings for Port Alfred (4.9 hrs) and Plettenberg Bay (4hrs) (Smith 2005b; Donovan 2010), while PEDSAC (7.4hrs) was considerably higher being similar to earlier estimates for the Eastern Cape (7.2hrs) in the mid 1990s (Brouwer and Buxton 2002).

Fishing trip duration in the Port Alfred recreational skiboat fishery declined from 1988 to 2007/8 and this has been attributed to three possible causes; (i) increasing costs leading to reduced travel distances, hence shorter times, (ii) improved technology (GPS and sonar) leading to shorter search times for fish and (iii) more stringent regulations leading to anglers attaining daily bag limits in a shorter period than previously, contributing to shorter fishing durations (Donovan 2010). These reasons are also likely to be the most pertinent for the lower fishing trip duration at Boknes and Kenton where the majority of fishing effort occurred within 5-10km of each access point. However, at PEDSAC capital investment into vessels and fishing equipment is far higher, suggesting that vessel owners may have greater financial resources and may therefore not be limited by the increasing running costs of vessels. Although most fishing occurs within a 30km radius of the Port Elizabeth harbour (Smale and Buxton 1985), many of the larger vessels registered at PEDSAC actively target pelagic species, particularly the yellow-fin tuna (*Thunnus albacares*). This species is more abundant in offshore waters in the Eastern Cape during the summer months (Smale and Buxton 1985) and recreational skiboat anglers may therefore travel longer distances (>50km) over these periods specifically to target this species, therefore contributing to longer fishing durations. Furthermore PEDSAC is an active competitive angling club and vessels remain at sea for as long as possible to maximise catches during club events.

Clear temporal trends in effort were also apparent with higher launching effort in December and January coinciding with the peak summer holiday period, as well as being higher over non-work days throughout the year. Smale and Buxton (1985) found greatest fishing effort in the western region of Algoa Bay from December to May, while Smith (2005b) reported peaks in January, December and April in the Plettenberg Bay fishery coinciding with the main Christmas and Easter holiday periods. Although long-term temporal trends indicate annual increases in recreational skiboat fishing effort

(Smale and Buxton 1985; Hecht and Tilney 1989; Sauer *et al.* 1997; Brouwer and Buxton 2002; Donovan 2010), the estimated effort at the PEDSAC launch site in boat-hours in the current study was similar to that reported in 1980 (Smale and Buxton 1985). The lack of a significant increase in the number of boat-hours between 1980 and the current study is somewhat surprising as the reduction in commercial linefish rights led to many vessels being eliminated from the commercial sector and the subsequent movement of these vessels into the recreational sector. A substantial increase in recreational fishing effort has occurred in the Port Alfred skiboat fishery which was attributed to this reduction in commercial rights (Donovan 2010), yet this was not observed in the Algoa Bay recreational linefishery. During the current survey 273 individual vessels were identified from the launch registers from the PEDSAC club, which is considerably higher than the 150 boats that were registered (may not have been active) at the club in 1982 (Smale and Buxton 1985) suggesting that participation in the fishery has indeed increased substantially. Furthermore fishing effort in angler-hours was approximately 21% higher than earlier estimates due to larger vessels and crew sizes indicating that recreational skiboat fishing effort has indeed increased.

(b) Catch rate

The longer fishing trip duration at PEDSAC coincided with a lower catch rate than at Boknes and Kenton launch sites. This may be due to longer travel distances and hence reduced fishing time as suggested above, or alternatively the influence of the new bag limits restricting the retained catch. Anglers who have travelled longer distances and have attained their daily bag limit for certain species may continue to fish in the hope of catching other species until they reach their total daily bag limit. Alternatively they may release smaller fish in the hope of catching larger fish later in the day, or continue fishing throughout the day purely for recreational purposes thereby contributing to lower observed catch rates.

Numerous factors influence recreational CPUE, hence the data from this study are insufficient to suggest that spatial differences in catch rates are due to localised declines in fish abundance resulting from higher levels of fishing pressure in this area. However, results from the fisheries independent survey (Chapter 3) confirm lower catch rates at both the Bell Buoy and St Croix areas (situated in the western sector of Algoa Bay) in comparison to several other reef areas in Algoa Bay. Furthermore this study has shown high levels of recreational fishing effort in these areas suggesting that the lower relative abundance of fish may be due to higher levels of fishing pressure.

The CPUE at all launch sites in the current study was lower (2.0-3.5 fish.angler^{-1}.day^{-11}) than that reported for previous assessment in the Eastern Cape (5.3 fish.angler^{-1}.day^{-1})(Brouwer and Buxton 2002) and in the Plettenberg Bay skiboat linefishery (4.7 fish.angler^{-1}.day^{-1}) (Smith 2005b). As mentioned this may not necessarily be due to lower abundances of fish, but rather the impact of more stringent fisheries regulations limiting the number of fish retained by anglers per day. However, using commercial catch data, which is not influenced by changes in the recreational fishing regulations, Donovan (2010) showed declines in the catch rate for several targeted species in the Port Alfred linefishery, and several studies have reported similar long-term declines in the South African linefishery (Crawford and Crous 1982; Smale and Buxton 1985; Brouwer 1997; Griffiths 2000).

Changes in recreational fisheries regulations can influence catch rate estimates significantly and independent methods need to be employed in order to monitor changes in resource abundance. This highlights the importance and need for fishery independent surveys for long-term temporal comparisons. The research conducted in Chapter 3 forms a comprehensive baseline for monitoring future trends in the state of the reef linefish resources.

(c) Harvest and catch composition

The total harvest in Algoa Bay was estimated at approximately 21 000 fish per year or 52 tons and was dominated by few species. This estimate does not take into account fishing effort from vessels launching from estuary mouths and from the Hobie and Kings beach launch sites and is therefore conservative. However, effort from these sites is low (Beach Manager's Office *pers. comm.*) and contribution to the overall effort and catch in Algoa Bay is therefore minimal.

The estimated number of fish landed at PEDSAC (13 689) in the current study was considerably lower than that of the estimated harvest in 1980 (25 138) (Smale and Buxton 1985). Furthermore the entire estimated catch (number of fish) for Algoa Bay (all three sites 20 873) was less than the previous estimate for PEDSAC. This is despite a notable increase in fishing effort at PEDSAC from approximately 29 000-32 000 angler-hours^{-1} in 1979-1980 to 51 149 angler-hours^{-1} per annum in the current study. This indicates a considerable reduction in the number of fish retained between the two study periods despite a significant increase in the participation in the recreational skiboat fishery in Algoa Bay. The lower estimated annual harvest (number of fish) in the current study could either be due to the more restrictive recreational fishery regulations currently in place, with bag limits restricting the daily catch, or alternatively to lower abundances of the target species as a result of heavy exploitation over the past three decades. Based on the current study it is not possible to differentiate between the potential effects of these factors on the retained catch. However, based on commercial catch data, Donovan (2010) showed that relative abundance of several linefish species in the Port Alfred area has declined from 1985 to 2007. Furthermore several studies have indicated the declining catch rates of important linefishery species in South Africa (Griffiths 1997a; Griffiths 1997b; Griffiths 2000). This suggests that the reduced harvest in the current study is likely to be due to a combination of both lower abundances of targeted fish and the more restrictive fishery regulations now in place.

Although the number of fish retained in the current study was lower than in the past, the total weight was higher. An estimated 52 tons of fish was landed by the recreational skiboat sector in Algoa Bay (all three sites) with PEDSAC accounting for 39 tons, which was higher than the 32-33 tons estimated by Smale and Buxton (1985) for the PEDSAC recreational fishery in 1979-1980. The difference in harvested weight between the two studies can be attributed to a change in the fishery regulations as well as the species composition of the catch between the two periods. MLS for many targeted species has increased between the two study periods and has resulted in a greater mean weight in the current study (2.5kg) compared to that found by Smale and Buxton (1985) (1.4kg). Furthermore, geelbek was abundant during the study period (Donovan 2010) and dominated the catch at the PEDSAC launch site, having a mean weight of 4.5kg. This was contrary to the findings of Smale and Buxton (1985) who reported the dominance of silver kob of a smaller average size (1.2kg). The change in dominant species from silver kob to larger sized geelbek in combination with increased MLS limits both contributed to the greater harvest (weight) estimated in the current study. Estimated annual harvest in the Port Alfred recreational fishery over the same time period was approximately 22 tons (Donovan 2010), while Smith (2005b) estimated the recreational harvest in the Plettenberg Bay skiboat fishery at 13 tons. This highlights the contribution of the skiboat fisheries to the total harvest of linefish along the east coast and the contribution of the recreational sector to the declining stock of many species. Historically recreational skiboat angling is estimated to have contributed 20-80% to the catches of at least nine collapsed stocks in South Africa (Griffiths and Lamberth 2002) and stringent management of the sector is therefore needed to ensure ongoing ecological and economic viability.

Approximately 75% of the Algoa Bay catch by number was comprised of three species, namely, santer, silver kob and geelbek. Spatially, differences in catch composition were evident which may be due to differences in the local abundance of fish or differences in fish habitat (Webley *et al.* 2010). Furthermore species-specific targeting can influence the composition of the catch. Santer dominated recreational catches in the eastern region of Algoa Bay in agreement with findings from the Port Alfred recreational skiboat fishery (Donovan 2010), while geelbek dominated in the western region. Silver kob was landed at all launch sites with its contribution ranging from 8 to 25% of the total catch, contrary to the Port Alfred recreational fishery where it was the most important species (Donovan 2010). Notably, during the controlled angling survey conducted within Algoa Bay (Chapter 3) a large proportion of the silver kob were below the current MLS possibly accounting for the low proportions of silver kob retained by recreational skiboat anglers in the current study in comparison to the previous study conducted in Algoa Bay. Recreational catches of silver kob in the Port Alfred skiboat fishery were half that of the commercial sector (Donovan 2010) highlighting the contribution of the recreational sector to the total harvest of this species despite a revised daily bag limit of five fish per angler. This highlights the magnitude of the recreational harvest in Algoa Bay and the contribution to overall stock declines in the South African linefishery. Regulation of fishing effort is key to reducing the recreational harvest yet difficult due to the open access nature of the fishery. Spatial and temporal closures may therefore be the only viable option for reducing recreational skiboat effort and in combination with revisions of bag and size limits may reduce the harvest of the recreational sector. The baseline data outlined in this chapter is valuable for integrating recreational fisheries information into future spatial planning in Algoa Bay.

Temporal changes in the relative proportions of species in the total catch of the recreational skiboat fishery are apparent in the Eastern Cape (Smale and Buxton 1985; Hecht and Tilney 1989; Brouwer 1997; Donovan 2010). The contribution of geelbek and santer to the catches of vessels at the PEDSAC lunch site has increased from 3% and 17% to 41% and 24% respectively, while that of silver kob and dageraad have decreased from 35% and 12% to 15% and less than 1% respectively (Smale and Buxton 1985). Higher catches of geelbek in the Port Elizabeth and Port Alfred skiboat fisheries during 2007-2008 can be attributed to the shoaling and migratory nature of the species resulting in large spatial and temporal fluctuations in abundance and hence availability to the fishery. Higher catches in the recreational sector, however, occurred despite the recent implementation of a stringent two fish daily bag limit for geelbek. This suggests that they are heavily targeted when they are present within Algoa Bay, or alternatively compliance with the regulations is poor.

Donovan (2010) suggested that stricter bag limits and higher operating costs has led to fewer vessels fishing deeper offshore waters, leading to increased targeting of shallow inshore reef species. This shift in fishing effort could account for reduced catches of carpenter and higher proportion of santer in the recreational catches. Similarly the contribution of carpenter, which is more abundant in deeper offshore waters (Hecht and Tilney 1989; Brouwer and Buxton 2002), to the recreational skiboat catch in the current study (2% vs. 4-7%) was lower than that reported by Smale and Buxton (1985). However, a decline in dageraad catches, which is an inshore species, was also apparent and may be due to reduced abundance as a result of higher levels of exploitation (Griffiths 2000), or alternatively a shift in the distribution of fishing effort leading to changes in the catch composition, or a combination of both. Nonetheless it is probable that the decline in the proportion of historically heavily targeted species such as red steenbras, dageraad and silver kob can indeed be attributed to high levels of exploitation and that the increase in the proportion of santer and roman in recent catches indicates signs of serial overfishing.

5.4.3 Compliance

Compliance of recreational shore anglers with bag limits was generally good. This, however, was largely due to the low catch rates with few anglers attaining the daily bag limits. Although the daily bag limit for dusky kob and white steenbras is one fish per person per day, only 0.5% and 0.9% of anglers succeeded in capturing more than one of each species per day respectively. Retention of undersized fish, however, was higher with approximately 33% of retained fish below the MLS. This was particularly noticeable in the more remote areas or areas which fisheries inspectors did not frequent. Non-compliance was also high for species considered overexploited, with 80% and 38% of the retained white steenbras and dusky kob below the MLS. Poor compliance is common in recreational fisheries (Brouwer *et al.* 1997; Brouwer 1997; Griffiths and Lamberth 2002; King 2005; Rangel and Erzini 2007) and the level of compliance as well as the knowledge of the regulations is strongly correlated with the level of enforcement in the fishery (Brouwer *et al.* 1997; Griffiths and Lamberth 2002). Past studies in the Eastern Cape have indicated that only 0.75% of shore anglers were inspected over a 12-month period, which was largely due to a lack of enforcement capacity as the number of inspectors per kilometer of coastline was low (0.03 inspectors/km) (Griffiths and Lamberth 2002). Improved enforcement and monitoring of the shore fishery will not only improve compliance but will also contribute to increased angler awareness of the fishery regulations.

Compliance of the recreational skiboat fishery with the regulations was generally good, with only one vessel exceeding the angler daily bag limit during the study period. The majority of anglers (70%) indicated that they did not often attain their daily bag limits; however, species-specific regulations were exceeded on several occasions. Anglers often claimed to be unfamiliar with the new regulations despite most vessels having marked measuring stickers on their vessels. This suggests that anglers willingly contravene the regulations when fish are abundant hoping that they will not be inspected. Compliance with MLS regulations was generally good, with only 5% of fish below MLS. Skippers indicated that they were either never inspected or inspected less than once in 100 outings by a fisheries officer. Based on the average of 19 fishing days per vessel per year this suggests that vessels were inspected at most once every five years, contrary to the 2.7 inspections per boat per year reported for the Eastern Cape (Griffiths and Lamberth 2002). Similarly in the Plettenberg Bay recreational skiboat fishery knowledge of the regulations was poor and the majority of recreational skiboat anglers were inspected less than once in 50 outings (Smith 2005b).

5.4.4 Economics

The total economic impact of recreational angling in South Africa was estimated to be at least 80% higher than that of the commercial sector in 2007 (Liebold and van Zyl 2008). Within Algoa Bay recreational shore anglers fished for approximately 48 days per annum with a daily expenditure of ZAR189 indicating that shore anglers spent approximately ZAR9 000 per annum on recreational angling activities. The local economic value of the recreational shore fishery in Algoa Bay is therefore significant despite the poor accessibility and low catch rate along most regions of the coastline. The economic value of the South African recreational shore fishery is estimated at between ZAR1.6 and ZAR2.5 billion and contributes to the employment of approximately 100 000 people (Griffiths and Lamberth 2002; Liebold and van Zyl 2008). This highlights the direct and indirect economic contribution of the shore based recreational fishery to the local economy.

The daily expenditure of recreational skiboat anglers in Algoa Bay was high (mean ZAR769), with approximately 70% of the cost attributed to fuel, and investment into fishing equipment (boats and tackle) was considerable (mean ZAR217 268). Due to the recent increase in fuel costs it is likely that many recreational skiboat anglers are restricted to fishing in close proximity to launch sites (Donovan 2010). However, it is apparent that the more affluent anglers with larger high value vessels are still willing to travel great distances offshore to target pelagic species. Despite the commercial skiboat linefishery accounting for 79% of the catch, the recreational sector accounts for an estimated 81% of local employment opportunities generating approximately 82% of the revenue from the linefishery (Griffiths and Lamberth 2002). The recreational linefishery is therefore of considerable importance to the economy of many coastal regions. However, due to the open access nature of the recreational fishery, effort is continually increasing, and in combination with a reduction in commercial fishing, recreational skiboat anglers are likely to account for larger proportions of the total catch in the future. Nonetheless the direct economic impact of recreational skiboat angling in South Africa was estimated at ZAR5.33 billion with fuel purchases accounting for 24% of the variable costs (Liebold and van Zyl 2008) further indicating the magnitude of the sector. Similar findings have been reported in the Mediterranean where the Spanish recreational boat-based fishery resulted in a 20% higher economic impact than the Spanish Mediterranean commercial fishery (Gordoa *et al.* 2004).

The contribution of recreational fisheries to both regional and national economies (Cowx and Arlinghaus 2008) as well as to stock decline (Griffiths and Lamberth 2002; Cooke and Cowx 2004) is now widely recognised and future management of the sector must take cognisance of both the biological and socio-economic impacts to ensure ongoing sustainability. This further emphasises the importance of obtaining accurate spatial baseline information on recreational fisheries for use in spatial planning initiatives to ensure long-term social, economic and ecological sustainability in the sector.

5.4.5 Management considerations

Management of the South African linefishery is complicated due to the multi-user and multi-species nature of the fishery. Historically this has resulted in poor and ineffective management, which is evident through the decline of several species (Griffiths 2000) and the declaration of a state of emergency in the fishery (DEAT 2000). Although large scale reduction in effort in the commercial skiboat fishery has occurred through a reduction in the number of rights holders, effort in the recreational skiboat fishery has increased, due to both a movement of vessels eliminated from the commercial sector into the recreational sector, as well as a general increase in recreational participation in the fishery. Both the recreational shore and skiboat fisheries are open access fisheries and are only managed through output controls with no limitation on effort. Effective reduction in effort can therefore only be achieved through spatial or seasonal closures. Although the benefits of such management actions on a local scale are most apparent to resident reef associated species, nomadic and migratory species may also benefit from the restriction of fishing during specific times when they form dense aggregations or when they congregate in specific areas. Several targeted linefish species are susceptible to commercial and recreational fishing pressure due to their predictable spatial and temporal patterns (Griffiths 2000). These known patterns can be used to effectively reduce fishing pressure on selected species in need of additional protection through the closure of sensitive aggregation areas. The spatial indices of effort and economic importance developed in this study provide a valuable means to integrate recreational fisheries considerations into spatial planning to ensure that sufficient areas remain open and accessible to the public, yet sensitive areas in which vulnerable and depleted stocks are targeted are protected from excessive recreational pressure.

Internationally the lack of monitoring data for recreational fisheries in the past has contributed to the poor recognition of the magnitude (Cooke and Cowx 2004) and impacts of this sector on the biological resources (Post *et al.* 2002). Similarly, the collection of recreational fisheries data in South Africa has been neglected, with the national survey in the mid 1990s being the first and only comprehensive assessment of catch and effort. Furthermore current day spatial planning and management requires high resolution spatial data, which is not available from past studies, and may not be captured during national surveys due to the broad objectives of the survey design. Detailed localised studies are therefore required to provide baseline data for both planning and long-term evaluation. Changes in management, and in particularly marine spatial zoning and the development of MPAs, may lead to the redistribution/displacement of fishing effort leading to further concentration of fishing effort, possibly resulting in negative ecological and socio-economic impacts (Dinmore *et al.* 2003; Baelde 2005; Hilborn *et al.* 2006; McPhee *et al.* 2008). However, studies quantifying such impacts are yet to be completed (McPhee *et al.* 2008) and baseline information prior to the establishment of spatial restrictions and MPAs is therefore essential for the subsequent monitoring of ecological and socio-economic conditions.

It is evident from this and other studies that recreational fisheries exert considerable pressures on the resources and output controls are the main regulatory means with which the fishery is managed, with little control on the regulation of effort. Evaluating long-term changes in the recreational fishing effort, and potential spatial changes as a result of new regulations, is therefore critical to determining the success of management initiatives. This study has indicated the magnitude of recreational fisheries in Algoa Bay both in terms of its economic value and the pressures (harvest) it places on the local marine resources. Comparison with past studies indicates a considerable increase in effort and overall harvest despite the implementation of more stringent fishery regulations. This raises concerns over the future sustainability of the stocks of most targeted species. Furthermore technological advances contribute to the increasing efficiency with which anglers are able to target their prey, placing increasing pressure on the natural resources. Regulation of recreational effort is therefore becoming increasingly important and may be most effectively implemented through spatial and temporal closures. This study has provided detailed baseline data and led to the development of spatial indices of recreational importance which can be used to facilitate future spatial planning. Furthermore, this baseline data can serve as a benchmark against which future changes in fishery characteristics in Algoa Bay can be evaluated.

5.4.6 Conclusions

This study employed multiple survey techniques to obtain a detailed understanding of the recreational shore and skiboat fisheries in Algoa Bay. Although some aspects of each sector have been documented in previous studies, none have provided information of sufficient spatial resolution across Algoa Bay to aid spatial planning. In order for localised spatial management initiatives to be successful, high resolution spatial data of the recreational fishery activities is required. On-site creel surveys could not be conducted throughout Algoa Bay due to financial and logistical constraints, but aerial surveys were highly effective at quantifying the spatial distribution of shore fishing effort, which is critical information required to support future spatial planning. These surveys revealed a highly heterogeneous spatial distribution of shore fishing effort and led to the identification of key factors influencing the distribution of anglers. Skiboat effort was quantified using a combination of effect counts and angler interviews which provided the necessary spatial information required for planning. Offshore recreational skiboat fishing effort also showed a high degree of spatial variability across Algoa Bay, an important consideration for future spatial planning. Both the shore and skiboat recreational effort data were integrated into a spatial index to depict areas of greatest importance to the recreational sector. Survey data were used to estimate annual effort and harvest of the shore and skiboat sectors, and determine an economic value for each. This contributed to the development of a spatial index of economic importance for the recreational sector. These spatial indices provide valuable information for inclusion of recreational fisheries data into marine spatial planning in Algoa Bay.

On-site surveys provided detailed information on catch and effort which were shown to be highly variable within the study area and influenced by numerous factors. This contributed to understanding the spatial and temporal variability and dynamics in both shore and skiboat sectors, and provided important information for the development of enforcement and monitoring programmes to improve compliance and evaluate changes in effort and catch in the long-term. Furthermore, the data provides the basis against which future monitoring studies can be compared to quantify changes in catch and effort as a result of new management regulations.

This chapter has therefore provided the data outlined in Table 5.11 required for future spatial planning and monitoring in Algoa Bay.

Table 5.11. Contribution of chapter results to spatial planning and monitoring in Algoa Bay.

Chapter 7: Systematic conservation planning	Chapter 8: Monitoring and evaluation
1. Development of spatial indices of relative shore and skiboat fishing effort 2. Development of integrated indices of relative recreational importance and relative economic importance	1. Baseline fishing effort and catch composition data for future comparative assessments 2. Identification of key factors influencing spatial and temporal dynamics in the fisheries 3. Understanding spatial and temporal variability to aid the design of monitoring programmes

CHAPTER 6
SPATIO-TEMPORAL ASSESSMENT OF COMMERCIAL FISHERIES WITHIN ALGOA BAY

6.1 Introduction

Commercial fisheries are regarded as one of the main drivers of change in marine ecosystems (Nelson 2005) and in many areas where they are not effectively managed they threaten the future ability for the sustained provision of marine ecosystem services on which human populations are reliant. The global decline in fish stocks and alteration of marine ecosystems is heavily influenced by commercial fisheries exploitation (Hilborn *et al.* 2003; Pauly *et al.* 2003; Cooke and Cowx 2006) due to the targeting and subsequent serial depletion of fish stocks, affecting the trophic structure and energy flow through marine ecosystems (Pauly *et al.* 1998). The industrialisation of commercial fisheries was driven by rapid technological advances during the 1950s and 1960s leading to massive increases in fishing effort in many regions, and the expansion of commercial fisheries into regions and habitats which were previously less heavily targeted, or were previously unfished. Global harvests escalated while simultaneously the impact of fisheries activities on marine habitats intensified through increasing use of destructive gears and more powerful vessels capable of fishing further afield and at greater depths. Government subsidisation contributed to the overcapitalisation of commercial fishing fleets (Hilborn *et al.* 2003; Pauly *et al.* 2005b; Pauly 2008) and continues to contribute to the exploitation of already depleted stocks by supporting commercial fisheries which are biologically and economically unsustainable (Pauly *et al.* 2002; Pauly *et al.* 2005b).

Both target and non-target species are affected by the activities of commercial fisheries. Some forms of bottom fishing are amongst the most detrimental commercial activity as it modifies the substratum, homogenising habitats and reducing species diversity (Thrush and Dayton 2002). Bycatch and discarding in many commercial fisheries is a major problem (Zeller and Pauly 2005; Cooke and Cowx 2006; Walmsley *et al.* 2007a; Walmsley *et al.* 2007b) resulting from poor gear selectivity or illegal targeting of bycatch species to increase fishery yields above quota allocations. Global discards of undersized or non-target species by commercial fisheries is estimated at 8% of the catch approximating 7.3 million tons annually (Kelleher, 2005). Poor enforcement and management of many fisheries has also contributed to the decline in fish stocks through increased illegal, unreported and unregulated (IUU) fishery activities which are major contributors to the overexploitation of commercial stocks (le Gallic and Cox 2006).

Cumulatively these factors contribute to ecosystem degradation through the removal of biomass, and damage to supporting habitats by several fishery sectors targeting different species in the same areas. In many cases conflicts arise due to competing interests between fishery sectors. Spatial assessment of commercial fisheries and management of fisheries activities through marine spatial planning is central to adopting a precautionary approach to fisheries in light of uncertainties in stock status and where limited scientific information is available (Pikitch *et al.* 2004). The design and implementation of spatial plans on a localised level requires high resolution spatial data which have often not been available for commercial fisheries in the past. The implementation of observer programmes and the advent of vessel monitoring systems (VMS) overcome these past limitations and in combination with log-book data allow for marine spatial planning to be conducted on a localised level.

6.1.1 Commercial fisheries in Algoa Bay

Numerous commercial fisheries operate along the South African coastline. Preliminary investigation of commercial activities along the east coast identified five sectors which operate within the Algoa Bay study area. The nature and activities of these commercial fisheries within Algoa Bay are outlined briefly below.

(a) Commercial linefishery

The South African commercial linefishery is a multi-species fishery catching up to 250 species (DEAT 2005c) of which 20 are commercially important (Lamberth and Joubert 1999). The boat-based linefishery grew rapidly during the 20th century due to the development of coastal infrastructure for launching and harbouring vessels, and technological improvements to vessels and fishing gear (DEAT 2005c) which contributed to improved catching efficiency. Initially effort was unregulated and by the end of the 20th century the commercial linefishery fleet had grown to between 2 500 to 3 000 active vessels with 20 000 to 25 000 participants involved in the sector (Mann 2000; Griffiths 2000). Regulations were first introduced in the 1940s; however, it was only in 1985 that a detailed management framework was implemented which included revised minimum legal size limits, daily bag limits, closed seasons and a restricted species list (Penney *et al.* 1989). The new regulations also capped effort by restricting further entry into the fishery (Penney *et al.* 1989). Although detailed regulations were implemented, most were based on subjective opinion as there was little scientific information available for most linefish species on which to base decisions (Griffiths 2000). As a result many of the regulations failed to limit catch and were therefore unsuccessful in reducing fishing mortality of heavily targeted species (Attwood and Bennett 1995; Brouwer *et al.* 1997; Sauer *et al.* 1997; Brouwer 1997). Stock assessments were conducted for several important fishery species during the 1990s and raised concern as to the poor stock status of many species, including those previously considered 'resilient' to fishing pressure (Griffiths 1997a; Griffiths 1997b; Hutton *et al.* 2001). The linefishery was declared to be in a state of emergency in 2000 (DEAT 2000) and the Total Allowable Effort (TAE) was set at 450 vessels and 3 450 persons. In addition revised bag and size limits were implemented and long-term rights were allocated in 2005/2006. National annual harvest was in the region of 16 000 tons prior to the reduction in effort but has subsequently decreased to between 5-7 000 tons (DEAT 2005c). A history of the changes in management regulations in the sector is presented in Box 6.1.

Box 6.1. Management history of the South African commercial linefishery (Source: Donovan 2010).

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 subsidised their 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 A- and B-licensed commercial fishers. 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 daily to two black musselcracker, red steenbras and seventy four (up until 1998, where after a ban was introduced for the species).

Management period B: 1999 – 2001
In 1999 the A- and B- commercial licences were abolished in favour of the allocation of annual fishing rights. 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. These fishing restrictions, however, applied largely to the recreational fishers.

Management period D: 2006 -2008
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.

(b) Chokka-squid jig fishery

The chokka-squid (*Loligo reynaudi*) is an economically important species targeted by the commercial linefishery along the south-east coast of South Africa (Britz *et al.* 2001). It is a short-lived species resulting in a highly dynamic fishery due to large inter-annual variability in stock abundance (Augustyn and Roel 1998; Roel and Payne 1998). The chokka-squid stocks are distributed predominantly between Plettenberg Bay and Port Alfred (Augustyn 1990; Augustyn *et al.* 1992) with seasonal peaks in abundance on inshore nesting grounds coinciding with the peak spawning period in summer (Sauer *et al.* 1992). Spawning grounds and nesting sites are generally situated on soft substrata within coastal embayments and are utilised repeatedly in subsequent years (Sauer *et al.* 1992).

Chokka-squid was initially a bycatch species landed by the foreign and domestic trawl fleets (Augustyn *et al.* 1992; Sauer *et al.* 2003b). Exclusion of the foreign trawling and the establishment of an overseas market led to the establishment of a dedicated highly selective entrepreneurial chokka-squid jig fishery in the mid 1980s (Augustyn 1990; Sauer *et al.* 1992). The jig fishery accounts for 80-90% of the chokka-squid catch with the remainder landed as bycatch in the domestic demersal trawl sector (Augustyn *et al.* 1992; Roberts and Sauer 1994). A license management system was implemented in 1987 to curb the rapid growth in the sector (Augustyn and Roel 1998), and further reduced the number of vessels participating in the fishery by 43% (Augustyn *et al.* 1992; Glazer and Butterworth 2006). Market demands for improved higher quality product led to the transition from smaller skiboats (6-8m) to large deck boats (>20m) with onboard freezer facilities. This increased both quality of the product and harvest capacity as vessels could spend longer durations at sea (Dorfler 2006). Fishing effort was traditionally concentrated on inshore spawning grounds (<50m) (Sauer 1995). However, improvements in vessels, echo-sounding equipment and lighting (allowing fishing at night) have contributed to the expansion of effort to the offshore feeding grounds (Glazer and Butterworth 2006). In order to protect dense aggregations over spawning grounds in summer a closed season of variable length was introduced during the late 1980s (Augustyn and Roel 1998). This is currently enforced for approximately five weeks from late October to late November (DEAT 2007c). The fishery is currently managed on a TAE basis (Sauer 1995; Augustyn and Roel 1998) and following the long-term rights allocation process effort was restricted to 138 vessels (2006-2013) (DEAT 2007c). Of these 46 are based in St Francis Bay with the remainder based in Port Elizabeth (Hara 2009). The chokka-squid jig fishery is managed nationally with no limitations on vessel movement. The stocks are primarily distributed in the Eastern Cape resulting in the sector being of considerable socio-economic importance to the local provincial economy (Sauer *et al.* 2003a).

(c) Small pelagic purse seine fishery

The small pelagic purse seine fishery (SPPSF) is based on small short-lived species which display high levels of natural variability in abundance, with the sardine (pilchard) (*Sardinops sagax*) and the anchovy (*Engraulis encrasicolus*) being the main target species (Cochrane *et al.* 2004; DEAT 2005b; Fairweather *et al.* 2006b), Bycatch species include the red-eye round herring (*Etrumeus whiteheadi*) and the Cape horse mackerel (*Trachurus trachurus capensis*) (Cochrane *et al.* 2004). National sardine landings peaked at 410 000 tons in 1962; however, a subsequent collapse occurred with less than 100 000 tons landed in 1966 (Britz *et al.* 2001). The declining annual harvest between 1962 and 1966 led to the introduction of small mesh nets to allow targeting of anchovy, which accounted for 80% of the sector's total landings in 1987. The small pelagic stocks typically undergo large natural variability on different time scales and the fishery is managed taking this into account with the annual Total Allowable Catch (TAC) based on estimates from annual spawner biomass surveys. More recently these surveys have indicated declining trends in sardine biomass from 2002 leading to an annual reduction in TAC in order to account for this. Currently there is no effort regulation; however, due to concerns of under-reporting a TAE limitation may be considered in future management (DEAT 2005b). Approximately 100 vessels participate in the fishery and vessels move freely along the coastline following the distribution of the stocks. Effort is primarily focused along the Western Cape coast with only a few smaller bait fishery vessels being based locally and fishing the Eastern Cape waters. Along the Eastern Cape coastline sardine is the main target species with the distribution of stocks largely determined by environmental conditions resulting in large spatial and temporal variability in abundance. A pelagic MPA has recently been established around the St Croix Islands in order to protect the pilchard stocks adjacent to the penguin nesting sites.

(d) Inshore demersal trawl fishery

The demersal trawl fishery targets two main groups; the shallow-water hake (*Merluccius capensis*) and the east coast (or Agulhas) sole (*Austroglossus pectoralis*) (DEAT 2005a; DEAT 2006). Both hake species(*M.capensis* and *M.paradoxus*) are managed collectively with a TAC set annually, of which the inshore trawl fleet is allocated 6% (DEAT 2005a). The inshore demersal trawl fleet consists of 35 vessels which are subject to size, power and gear restrictions. They are permitted to trawl in depths less than 110m and within 20 nautical miles of the coastline from which the offshore trawl fleet is excluded (DEAT 2005a).

Although there is some overlap in the distribution of the two hake species, trawling effort east of 20°E on the Agulhas Bank primarily targets shallow-water hake (Wilkinson and Japp 2005). The east coast sole TAC is allocated in its entirety to the inshore demersal trawl fishery with 872 tons allocated per year since 1992 (DEAT 2007a). A large proportion of inshore trawling effort occurs along the 100m isobath from Port Elizabeth north eastwards along the shelf edge (Wilkinson and Japp 2005). Most inshore trawl vessels utilise ice with no onboard freezing capabilities and trips are therefore limited to a few days (Wilkinson and Japp 2005). Inshore trawling is subject to spatial restrictions through the existing MPAs network, and closed trawling areas within embayments along the south-east coast. Illegal targeting of bycatch is a major problem, with catches of kingklip and linefish species being of particular concern (DEAT 2005a). Over ZAR100 million has been invested in assets in this sector, which supports the employment of 1 100 people and an annual catch worth approximately ZAR60 million (DEAT 2005a).

(e) Demersal shark longline fishery

Directed longline permits for targeting sharks were first introduced in the early 1990s (Japp 1999) and were subsequently separated into demersal and pelagic sectors in 2005 with the allocation of long-term rights (Clarke and Smith 2007). The demersal shark longline sector is an inshore fishery which utilises bottom-set gear in waters generally shallower than 100m. The smooth-hound shark (*Mustelus mustelus*) and the soupfin shark (*Galeorhinus galeus*) are the two main target species, but several species of carcharhinids, sphyrnids and the batoids are also landed (Japp 1999; Da Silva and Bürgener 2007). Following the allocation of long-term rights in 2005/2006 the TAE for the demersal shark longline fishery was set at six permits (Da Silva and Bürgener 2007; DEAT 2007b).

6.1.2 Data sources

Permit conditions for all commercial rights holders require daily catch returns to be submitted reporting catch, effort and spatial information. For most commercial sectors in South Africa a long time series of data therefore exists, but the validity and accuracy of the data has often been questioned (Sauer *et al.* 1997; Attwood and Farquhar 1999; Griffiths 2000; Da Silva and Bürgener 2007). Studies have, however, indicated that the inaccuracies are due to non- or underreporting which leads to an underestimation of the total harvest, yet major trends in CPUE and catch composition, which are independent of reporting of total catch, are generally accurate (Penney *et al.* 1997; Attwood and Farquhar 1999; Griffiths 2000).

Observer programmes have also been implemented in order to monitor catches and validate catch return data submitted by permit holders. In fisheries with larger vessels which spend several days at sea, onboard observers monitor catches and record fishing locations, while the landings of fisheries utilising small vessels are monitored on return at access points. Independent observer reports provide a valuable means to validate catch return data. However, in some fisheries, no dedicated fisheries monitoring programme exists, precluding verification of the logbook catch return data.

The advent of GPS units has allowed for improved monitoring of the spatial distribution of fishing effort by onboard observers. However, in fisheries where access points are monitored spatial data are provided by vessel skippers leading to potential reporting bias as no independent means for validation was historically available. The development of satellite tracking has allowed for accurate spatial monitoring of fishing activity through Vessel Monitoring Systems (VMS). VMS have typically been used for enforcement and monitoring of fishing activity within and adjacent to MPAs and little use of the data have been made for research purposes in South Africa. There has been an increasing worldwide trend in the use of VMS data for research and monitoring (Deng *et al.* 2005; Hiddink *et al.* 2006; Mills *et al.* 2007; Campanis and Thompson 2007; Palmer and Wigley 2009; Lee *et al.* 2010; Bastardie *et al.* 2010; Gerritsen and Lordan 2011) as it provides highly accurate spatial information allowing for detailed tracking and mapping of vessel movement and inference of fishing activity based on vessel behaviour. No such studies have previously been conducted in South Africa; however, the South African National Biodiversity Initiative (SANBI) is currently analysing VMS data to aid in the design of offshore deepwater MPAs in South Africa's territorial waters.

Fisheries information available for commercial sectors in South Africa is restricted to assessments of single species, or national assessments of the different sectors, with few regional or local level assessments having been conducted. Planning and developing management strategies on a local level is therefore difficult due to the poor spatial resolution of data. The overall aim of this chapter was to assess all commercial fisheries sectors which target resources within Algoa Bay using existing data sources described above. This assessment is required for and will contribute to marine spatial planning in Algoa Bay (Chapter 7) and the development of monitoring protocols for long-term evaluation (Chapter 8). The main objectives of this chapter were:

1. to identify the commercial fishery sectors active within Algoa Bay;
2. to determine the local spatial and temporal trends in catch, effort and CPUE for each sector;
3. to assess long-term temporal changes in the commercial sectors active within Algoa Bay; and
4. to develop spatial indices for the commercial fisheries in order to inform future spatial planning and monitoring in Algoa Bay.

6.2 Materials and methods

6.2.1 Commercial linefishery

(a) National Marine Linefish System catch and effort data

The National Marine Linefish System (NMLS) was developed by DAFF (formerly MCM), the regulatory authority, in 1985 to recorded compulsory catch return data submitted by commercial rights holders. Data recorded in the system includes the vessel identification/rights holder, date, fishing duration, number of crew, catch weight per species and spatial information reported as reference to coastal location and estimated distance offshore. The spatial data for each catch entry have recently been converted to a 5' grid reference for the entire fishery (Wilke and Kerwath 2008). This data forms the basis for the evaluation of the commercial linefishery. The Algoa Bay study area was defined as 25°30'E to 26°40'E and all catch records for this region were obtained from DAFF from January 1985 to December 2008.

Effort within Algoa Bay was determined as the number of vessels reporting catches and the number of boat-days per month. Both are continuous response variables, were approximately normally distributed and were modelled using a GLM with the identify-link function as described in Chapter 3. To model effort the GLMs took the following form:

$$Vn = \beta_0 + \beta_1(year) + \beta_2(month) + \varepsilon \qquad \text{Equation 6.1}$$

$$Bd = \beta_0 + \beta_1(year) + \beta_2(month) + \varepsilon \qquad \text{Equation 6.2}$$

where *Vn* and *Bd* are the number of vessels and boat-days per month respectively, year and month are the respective temporal aspects of the model, β_{0-i} are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

CPUE has been widely used as an index of abundance for assessment of fish populations and is based on the assumption that catch rate is proportional to abundance (Harley et al. 2004). This is despite the fact that it has long been recognized that CPUE does not always accurately reflect trends in abundance of fish stocks (Beverton and Holt 1957). Hyperstability often occurs, meaning that the CPUE remains high while the actual abundance of the stocks declines, thereby giving the false impression on the state of the resources. This occurs due to the non-random spatial distribution of fishing effort and the predictable distribution of target stocks and is of particular concern for shoaling and aggregating species. Nonetheless CPUE is often the only data available and hence remains widely used in fishery assessments. In this chapter, however, CPUE is not used to quantitatively estimate stock abundance but rather to investigate temporal and spatial trends in each of the commercial sectors. Standardisation of the CPUE data is used to take into account other factors which may influence catch rate, and to identify which factors have important implications for management.

CPUE was calculated as landed weight in kilograms per vessel per day (kg.boat-day^{-1}) and was standardised with a GLM based on the gamma distribution and log-link function as outlined in Chapter 3. Year and month represent the respective temporal aspects of the model. Crew was the number of reported crew members on board the vessel each day and was used to account for vessel size. Area in the model refers to the six primary fishing locations identified within Algoa Bay through analysis of VMS data (see below). Reported catch and effort were assigned to each fishing area based on the spatial reference from the 5' grids system used in the NMLS. The GLM took the form:

$$Log(CPUE) = \beta_0 + \beta_1(year) + \beta_2(month) + \beta_3(crew) + \beta_4(area) + \varepsilon \qquad \text{Equation 6.3}$$

where β_{0-i} are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

Several changes in the commercial linefishery (Box 6.1) have occurred since the development of the first linefish management framework in 1985. During the early stages of the fishery when A (full-time) and B (part-time) licenses were allocated several vessels fished infrequently as the motivation for fishing differed based on the financial dependence of A and B rights holders on the fishery. In order to reduce the influence of less-active vessels on the estimation of CPUE for spatial and temporal comparisons, less active vessels were removed from the CPUE dataset (Punt *et al.* 2000). This pertained to vessels which had reported less than 24 catch returns per year and had not been active participants in the fishery for at least five years. CPUE was modelled on this reduced dataset as well as on the complete dataset for comparative purposes.

Due to the presence of several zero catch records in the dataset for species specific CPUE, a Delta-Gamma model was used as described in detail in Chapter 4. The Probability of Capture (*Pc*) was modelled on a Binomial distribution with the logit-link function and the GLM took the form:

$$log(p(PC)/(1-PC))) = \beta_0 + \beta_1(year) + \beta_2(month) + \beta_3(crew) + \beta_4(area) + \varepsilon \qquad \text{Equation 6.4}$$

The positive catch rate was modelled on a Gamma distribution with the log-link function and the GLM took the form:

$$Log(+veCPUE) = \beta_0 + \beta_1(year) + \beta_2(month) + \beta_3(crew) + \beta_4(area) + \varepsilon \qquad \text{Equation 6.5}$$

where β_{0-i} are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

Long-term temporal and spatial trends in the species composition were determined and presented graphically as the proportion of the catch weight of dominant taxa to the total annual catch. ANOSIM tests were conducted in Primer 6 on standardised species composition data following square-root transformation to test for differences in catch composition between management periods and spatially between fishing areas. The proportions of nine important linefishery species and all elasmobranchs combined was compared by management period, season and fishing area using a parametric one-way ANOVA or non-parametric Kruskal-Wallis ANOVA depending on the underlying distribution properties of each dataset.

(b) Vessel monitoring system data

New regulations implemented in April 2007 required all commercial linefish permit holders to have VMS units fitted to their vessels registered in the sector. These systems allow automated and continuous monitoring of fishing vessel positions and were implemented for compliance purposes to monitor activity within restricted fishing areas. The polling interval[10] for the commercial linefishery vessels in Algoa Bay was between 10 and 15 minutes per vessel providing high resolution data allowing for the accurate assessment of the spatial distribution of fishing effort.

VMS data were received for 12 vessels from April 2007 until January 2009 and was plotted spatially in ArcView 3.2. Data were screened for outliers and duplicate entries and all polling activity from harbours and areas known to be safe anchorage points where fishing is unlikely to have occurred were removed. Vessel travel speed was calculated as the straight line distance between successive VMS polling waypoints and a vessel speed rule was employed to differentiate between travel and fishing related activities. As most commercial linefishing effort occurs on anchor or at slow drift speeds all entries where travel speed was calculated to be above 4km.h^{-1} were regarded as non-fishing

[10] Time interval between successive transmissions of a vessel's positional data

activities and were not included in further spatial analyses. Fishing effort was determined as the number of boat-days per 1km^2 grid cells and presented spatially. Areas of high fishing intensity were identified and allowed for the demarcation of six main fishing areas used by the commercial fishing fleet. Catch and effort data reported by rights holders were assigned to one of the six fishing areas based on the spatial information provided in the NMLS.

Effort was determined as the number of boat-days.vessel^{-1} and was linked to records in the NMLS database using dates and vessel unique identifier codes. Monthly fishing effort per vessel was compared between the reported catch (log-book) and VMS data to determine the accuracy of the two datasets using a paired t-test or, alternatively Wilcoxon Matched Pairs Tests for normal and non-normal distributed data respectively.

(c) Observer programme data

Access point monitoring data for the Port Elizabeth and Boknes launch sites were obtained from DAFF for the 2008 year. Total weight of landings per vessel and landed weight for the dominant species per vessel recorded by the observers was compared to the respective catch weights reported by the permit holders on the same day and recorded in the NMLS using paired t-tests following tests for normality and homogeneity of variances.

6.2.2 Chokka-squid jig fishery

(a) Catch and effort data

Since the initiation of the chokka-squid jig fishery in the mid 1980s it was regarded as a sector of the commercial linefishery and catch returns were recorded in the NMLS. The NMLS was primarily designed to capture data for the linefish sector which is a multi-species fishery. Specific information on species targeting was not recorded, resulting in difficulties in differentiating between traditional linefish (baited hooks) and squid (jig) fishing effort. However, due to the high selectivity of the chokka-squid jig fishery it was assumed that vessels were targeting chokka-squid on days when squid were reported in the catch (J.P.Glazer *pers. comm.*). As a result all records where squid was reported in the NMLS for the Algoa Bay study area (25°30'E and 26°40'E) were extracted from the NMLS database for the period 1985 to 2005 for use in this analysis. Due to these limitations in the database effort may be underestimated as days on which vessels were targeting squid but none were caught are excluded, This inturn will overinflate the estimate of catch rate, however, the problem is unavoidable and likely to be negligible (R.W. Leslie *pers. comm.*).

A new logbook recording system and dedicated Squid Database were designed by DAFF during 2006 in order to overcome the limitations of the NMLS and improve the accuracy of future data. However, during 2006 there was overlap between the two reporting and database systems (J.P.Glazer *pers. comm.*) leading to inaccuracies in the estimation of annual catch and effort for this year. Due to these inaccuracies catch and effort for 2006 is not reported for the chokka-squid fishery. However, CPUE remains unaffected as it is calculated on a daily basis and was therefore calculated for 2006 using the data available. The new squid logbooks were fully implemented in 2007 and catch and effort data were obtained from this database for 2007 and 2008 and merged with the NMLS data (1985-2005) for further analysis. There remains some uncertainty as to the accuracy of the records which are currently in the process of being validated.

Effort in Algoa Bay was determined as the number of vessels reporting catches and the number of boat-days per month. Effort in both vessel number (Vn) and boat-days (Bd) were non-normally distributed being left skewed and they were therefore modelled on a Gamma distribution using a GLM with the log-link function, of the following forms:

$$log(Vn) = \beta_0 + \beta_1(year) + \beta_2(month) + \varepsilon \qquad \text{Equation 6.6}$$

$$log(Bd) = \beta_0 + \beta_1(year) + \beta_2(month) + \varepsilon \qquad \text{Equation 6.7}$$

where *Vn* is the number of vessels reporting squid catches per month, *Bd* is the number of squid directed fishing days per month and β_{0-i} are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

Chokka-squid are targeted on spawning aggregations and CPUE may therefore be hyperstable and mask a decline in stock abundance. However, no fisheries independent data are available for the chokka-squid fishery and logbook data remains the only available source of information for spatial and temporal assessment of trends. CPUE was determined per vessel as landed weight in kilograms.day^{-1} (kg.boat-day^{-1}) and was standardised with a GLM based on the Gamma distribution and log-link function. Year and month represent the respective temporal aspects of the model, with area being one of four broad fishing locations identified through the analysis of VMS data (see next section below). Vessel identification was included in the model to account for the TAE or crew complement of each vessel. The data were screened to remove vessels which fished infrequently during the earlier periods of the linefishery in order to reduce the influence of less active vessels on the estimation of catch rate (Punt *et al.* 2000). Vessels which reported less than 24 catch returns per year and which were not active within the fishery for at least five years were removed from the CPUE dataset. The model took the form:

$$log(CPUE) = \beta_0 + \beta_1(year) + \beta_2(month) + \beta_3(area) + \beta_4(vessel) + \varepsilon \qquad \text{Equation 6.8}$$

where β_{0-i} are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

Spatial comparisons of monthly effort, CPUE and annual landed catches were conducted between the four broad fishing areas using non-parametric Kruskal-Wallis tests. The analysis was limited to recent data subsequent to the allocation of medium term rights in 2002.

(b) Vessel monitoring system data

VMS data (described above) were requested for a one-year period for all vessels with active rights in the chokka-squid jig fishery. Data were received for 57 vessels for a 12-month period from 1 December 2006 to 30 November 2007. Data were plotted spatially in ArcView 3.2. Outliers and duplicate entries, as well as polling activity from harbours and areas known to be safe anchorage points where fishing is unlikely to have occurred were removed. Vessel travel speed was calculated as the straight line distance between successive VMS polling waypoints and a vessel speed rule was employed to differentiate between travel and fishing related activities. Chokka-squid vessels typically locate spawning aggregations in 30-40m water depth using sonar and deploy an anchor to maintain their position over these dense aggregations. A large proportion of fishing effort therefore occurs while stationary; however, 'parachute' fishing occurs in deeper waters outside of the spawning period and involves fishing at slow drift speeds in deeper water over the feeding grounds (Glazer and Butterworth 2006). In order to accommodate 'parachute' fishing an $8km.hr^{-1}$ vessel speed rule was employed to the VMS dataset. Effort was determined spatially per $1km^2$ grid cell and was used to identify the main fishing grounds in Algoa Bay. Unique vessel identification codes could not be obtained for the chokka-squid fishery VMS data and a cross validation with catch records could therefore not be undertaken.

6.2.3 Small pelagic purse seine fishery

(a) Catch and effort data

Historical catch return data for the SPPSF were received from DAFF for the period from 1990 to 2008. Effort in Algoa Bay was determined as the number of vessels and boat-days per month. Both effort as number of vessels reporting catches per month and effort in boat-days per month approximated the Gamma distribution. Effort was therefore modelled on the Gamma distribution with log-link function as described previously. GLMs took the following forms:

$$Log(Vn) = \beta_0 + \beta_1(year) + \beta_2(month) + \varepsilon \qquad \text{Equation 6.9}$$

$$Log(Bd) = \beta_0 + \beta_1(year) + \beta_2(month) + \varepsilon \qquad \text{Equation 6.10}$$

where *Vn* and *Bd* are the number of vessels and boat-days per month, year and month are the respective temporal aspects of the models, and β_{0-i} are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

CPUE in pelagic fisheries may be hyperstable due to the shoaling nature of the target species. Interpretation of the CPUE data for quantitative purposes must therefore take this into account. However, in this instance CPUE is used for comparative purposes to reflect spatial and temporal trends and is used in conjunction with indicators of effort. CPUE was calculated as landed weight per day and per haul. CPUE approximated the normal distribution and the identity-link function was used with the model taking the following form:

$$CPUE = \beta_0 + \beta_1(year) + \beta_2(month) + \beta_3(vessel) + \varepsilon \qquad \text{Equation 6.11}$$

where β_{0-i} are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

(b) Observer data

Observer data collected by onboard fisheries monitors were received for the period 2002 to 2008 and was used to assess the accuracy of catch returns submitted by vessel skippers. Observer and logbook data were linked by vessel registration and date to identify corresponding records. Where records corresponded a paired t-test was used to assess the accuracy of the reported weights and fishing locations.

(c) VMS data

Over 100 vessels participate in the SPPSF nationally, but eight vessels are known to operate locally within Algoa Bay on a regular basis and VMS data were requested and received for six of these vessels. Polling intervals ranged from 10 to 30 minutes and data were received for a two-year period (2007-2008). VMS data were plotted spatially in ArcView 3.2 and duplicate and erroneous data were removed. Distance between polling waypoints and polling interval was used to calculate vessel travel speed. Heading (direction) of the travel path between success polling locations was determined and change in direction was calculated between successive travel paths. As the target species are pelagic, fishing grounds are not as predictable as in other fisheries with distribution of fish strongly influenced by local environmental conditions with water temperature being a strong determinant. Fishing therefore typically involves long periods of searching for and locating shoals of sardine and relatively short period of gear deployment and retrieval. It is therefore difficult to identify catch locations from VMS data as they are highly dispersed. However, it is possible to identify areas in which vessels operate regularly suggesting important fishing areas. Speed and directional rules was used to differentiate between steaming and possible fishing activities in order to reduce the importance of areas through which vessels pass regularly, such as around the harbour entrances where fishing is unlikely to occur. Furthermore on locating a target shoal, vessels reduce speed and alter course to deploy nets. VMS data were used to identify areas in which vessels travelled at speeds less than $6km.hr^{-1}$ and altered their course by 45^{o}[11] relative to the previous travel path. Although it is recognised that not all these locations may be catch locations it is used to infer areas of vessel activity from the VMS data for the SPPSF in Algoa Bay.

6.2.4 Inshore demersal trawl fishery

(a) Catch and effort data

The demersal trawl sector utilises a larger 20' grid system for reporting and recording catch and effort data. As a result the study area for this sector was larger than that for other commercial sectors extending from 25°20'E to 26°40'E. Catch and effort data for all vessels reporting from the Algoa Bay region between 2000 and 2008 were obtained from DAFF. Data included a unique vessel identification number, trawl date, trawl duration, spatial reference (20' grid reference) and landed weight per species for each trawl conducted.

Effort in Algoa Bay was determined as the number of vessels reporting and the reported effort in trawl-hours per month. The data approximated the gamma distribution and was therefore modelled using a gamma GLM with log-link function which took the form:

$$log(Vn) = \beta_0 + \beta_1(year) + \beta_2(month) + \varepsilon \qquad \text{Equation 6.12}$$

where year and month are the respective temporal components of the model in which fishing took place, $log(Vn)$ is the log-transformed vessel number, and β_{0-i} are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

[11] Searching takes place at speeds greater than 6km/hr and travel paths are usually in straight lines

Effort in trawl-hours (TrHr) was log-normally distributed and was modelled on the normal distribution with identity-link function following a natural logarithmic transformation of the data (logTrHr). The model took the form:

$$log\,TrHr = \beta_0 + \beta_1(year) + \beta_2(month) + \varepsilon \qquad \text{Equation 6.13}$$

where year and month are the respective temporal components of the model in which fishing took place, *logTrHr* is the log-transformed data, and β_{0-i} are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

Model estimates were back transformed taking into account the variance in the log-normal distribution using the following equation (Singh *et al.* 1997; Daug *et al.* 2002; Dick 2004):

$$TrHr = exp(log\,TrHr + \frac{\sigma^2}{2}) \qquad \text{Equation 6.14}$$

where *TrHr* is the back transformed value, *logTrHr* is the log-transformed model estimate and σ is the variance estimated by the model.

CPUE was calculated as landed weight per trawl-hour (kg.trawl-hour^{-1}) which was standardised using a gamma GLM with log-link function. In order to reduce variability in the estimates of CPUE a subset of the data were selected based on vessels which were indicative of the local fishery using specific criteria (Punt *et al.* 2000). The criteria used for selection included vessels which were active in the Algoa Bay region for at least four out of the nine years, and vessels which had submitted data for at least 12 fishing days per year within Algoa Bay. The model took the form:

$$Log(CPUE) = \beta_0 + \beta_1(year) + \beta_2(month) + \beta_3(vessel) + \beta_4(grid) + \varepsilon \qquad \text{Equation 6.15}$$

where year and month are the temporal aspects of the model, vessel is the unique identifier code for each vessel included to take into account the fishing power of each vessel, grid is the 20' grid in which catch was reported, and β_{0-i} are the coefficient estimates and ε the error (McCullagh and Nelder 1995).

Differences in monthly effort, and annual landings between grid cells were tested with non-parametric Kruskal-Wallis ANOVAs, using the Bonferroni Adjustment to identify where significant differences occurred. Spatial differences in catch compositions were tested using an ANOSIM test with results plotted in a nMDS ordination.

(b) Observer data

Observer data recorded by an onboard fisheries monitor were received for a subsample of trawls conducted each year. This dataset included the start and end points of each trawl, the depth at the start of the trawl and the species composition of the landed catch. Trawl start and end points from the observer data were joined in ArcView to plot the distribution of trawl paths spatially and trawl distances were calculated as the straight line distance between the two points. Data which were obviously erroneous were removed from the dataset. Trawl paths were intersected with a 1km^2 grid in ArcView and the number of trawls.km^{-2} and the trawl distance.km^{-2} was determined and presented spatially as the percentage of Algoa Bay trawl effort per km^2. Similarly trawl distance per 20' grid cell was determined from the observer dataset for comparison with the reported fisheries dependent data.

Fishery independent catch composition from the observer dataset was determined as the percentage composition of dominant taxa per year which was compared to the reported annual catch composition using Wilcoxon Matched Pairs tests.

6.2.5 Demersal shark longline fishery

(a) Catch and effort data

Catch data were received from DAFF for the years 2006 and 2007 and included number of hooks set, duration of soak time, location (GPS coordinates) and landed weight per species. Data were not normally distributed and non-parametric Kruskal-Wallis ANOVAs or Mann-Whitney U tests were used to compare effort, landed weight and catch rate temporally and spatially. CPUE was calculated for each line set as the weight per hook-hour (kg.hook-$hour^{-1}$) using the following formula:

$$CPUE = \frac{TC}{St \times H} \qquad \text{Equation 6.16}$$

where *TC* is the total catch weight, *St* is the soak time calculated as the difference in time from deployment to retrieval of each line, and *H* is the number of hooks per line.

Waypoints of longline deployment sites were plotted spatially in ArcView 3.2 and the proportion of fishing effort per 1km^2-grid was determined.

6.2.6 Spatial indices of commercial fishery activities

(a) Index of relative commercial importance (IRCI)

To determine the cumulative importance of commercial fisheries in Algoa Bay a spatial index of relative commercial importance (IRCI) was developed. This was undertaken by adding the proportional levels of effort per km^2 across the five commercial sectors active within Algoa Bay as determined from VMS or observer data. A scaling parameter was introduced to take into account the relative importance of Algoa Bay to the national distribution of fishing effort for each sector. The scaling parameter was determined based on the proportion of the fishery sector effort occurring within Algoa Bay using:

- VMS data for the chokka-squid and SPPSF sectors;
- Catch return positional information for the demersal shark longline fishery;
- The proportion of Algoa Bay landings to the national landed catch for the inshore demersal trawl fishery;
- No scaling parameter was used for the commercial linefishery as vessels are small and based locally within Port Elizabeth with little to no movement to other areas of the coastline.

The scaling parameter for the SPPSF was designed to take into account the importance of Algoa Bay to smaller vessels based locally which fish predominantly in the surrounding waters. The index was calculated for each km^2 as follows:

$$IRCI = (CLx1) + (CSxP_{CS}) + (SPPSFxP_{SPPSF}) + (DTxP_{DT}) + (SLLxP_{SLL}) \qquad \text{Equation 6.17}$$

where CL is the percentage of Algoa Bay commercial linefishing effort per km^2, CS is the proportion of Algoa Bay chokka-squid fishing effort per km^2, P_{CS} is the proportion of national chokka-squid effort occurring within Algoa Bay, DT is the percentage of Algoa Bay demersal trawl effort per km^2, P_{DT} is the proportion of national inshore demersal trawl catch landed within Algoa Bay, SLL is the proportion of Algoa Bay shark longline fishing effort per km^2, and P_{SLL} is the proportion of shark longline fishing effort occurring within Algoa Bay. The results were scaled so that grid cells had a maximum possible value of 10 and were plotted to display the commercial fishery importance spatially.

(b) Economic index of relative commercial importance

The economic importance of Algoa Bay to each commercial sector varies based on the distribution of the target stock, the size (number of rights) and magnitude of investment per sector, and the economic value of the target species landed within the region. The overall commercial fishery importance of Algoa Bay is therefore dependent on the combined effect of the extent of fishing effort and the economic importance. In order to take economic importance into consideration, the IRCI was weighted based on the relative economic importance of each commercial sector to the overall commercial economic value of fisheries in Algoa Bay. The national economic value of the landings for each sector was obtained from published sources (Anon 2004). In order to obtain an estimate of the economic importance of the fisheries in Algoa Bay, the national economic value per sector was scaled by the ratio of the number of vessels reporting catches within Algoa Bay to the number of vessels in the national fleet. The relative contribution of each sector to the economic value of commercial fisheries in Algoa Bay was determined and used to scale the proportion of that sector's effort per km^2 in the IRCI. This ensured that fisheries which contributed a significant proportion of the effort, but were of lower local economic importance, were scaled to reflect their lower value.

$$EIRCI = (RE_{CL}x\frac{VAB_{CL}}{VT_{CL}}) + (RE_{CS}x\frac{VAB_{CS}}{VT_{CS}}) + (RE_{SPPSF}x\frac{VAB_{SPPSF}}{VT_{SPPSF}}) + (RE_{DT}x\frac{VAB_{DT}}{VT_{DT}}) + (RE_{SLL}x\frac{VAB_{SLL}}{VT_{SLL}})$$

Where $EIRCI$ is the commercial economic importance per km^2, RE_{CL} is the relative effort of the commercial linefishery per km^2, RE_{CS} is the relative effort of the chokka-squid fishery per km^2, RE_{SPPSF} is the relative effort of the small pelagic purse seine fishery per km^2, RE_{DT} is the relative effort of the demersal trawl fishery per km^2, RE_{SLL} is the relative effort of the demersal shark longline fishery per km^2, VAB is the mean number of vessels reporting catches in Algoa Bay per annum for each sector respectively, and VT is the total number of vessels nationally in each sector.

6.2.7 General

In all GLM analyses AIC was used to select the optimal combination of factors prior to running each model and diagnostic plots were used to assess the appropriateness of model fit. Where data met the assumptions of normality (prior to or after transformation) the Tukey's HSD test was used to identify differences between groups, while non-normal data were tested with a Kruskal-Wallis ANOVA and pairwise comparisons were conducted using the Mann-Whitney U test with a Bonferroni adjusted level of significance as described in earlier chapters.

6.3 Results

6.3.1 Commercial linefishery

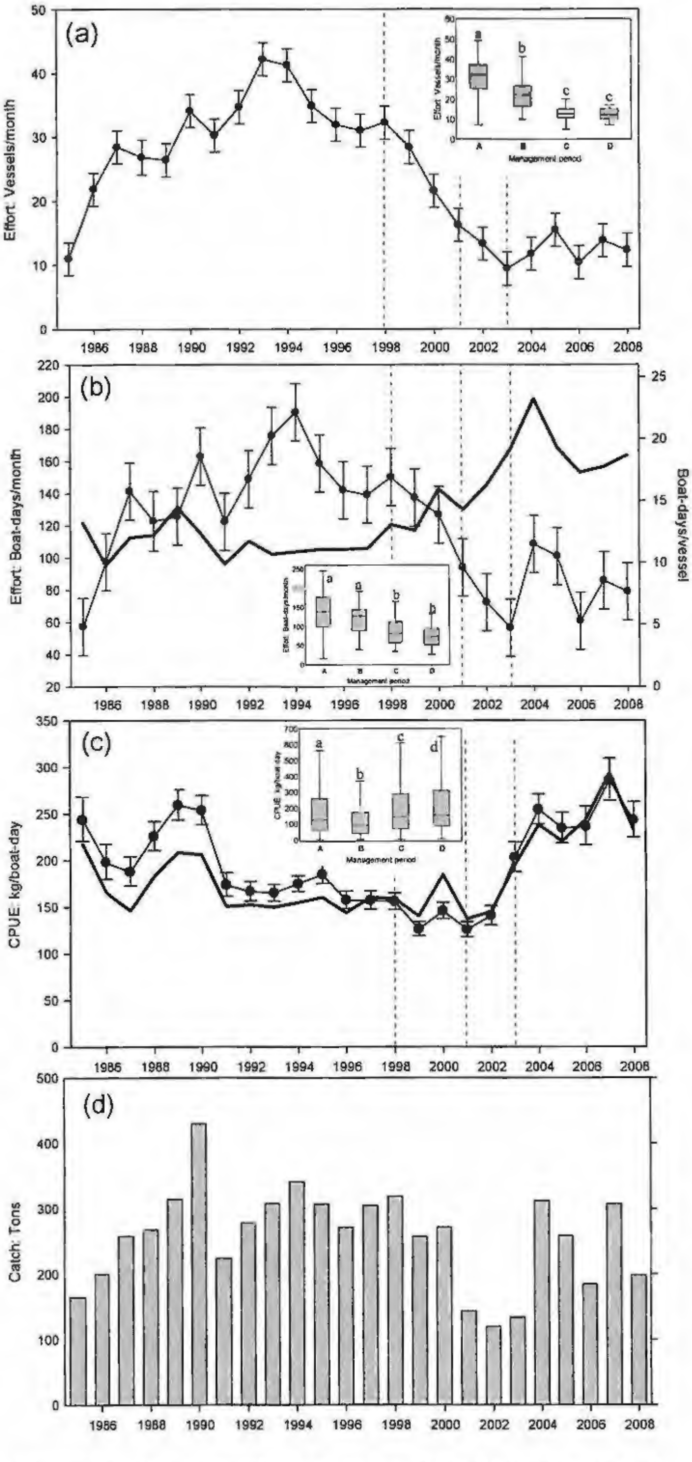


Figure 6.1. Annual trends in mean monthly effort (a) number of vessels, (b) number of boat-days, and average boat-days per vessel (thick solid line) (c) standardised CPUE using the reduced dataset and complete dataset (thick solid line) and (d) total landed catch. Dashed vertical lines denote separation between management periods. Differing letters above error bars denote significant differences.

(a) Temporal trends

Participation (vessel number) in the commercial linefishery in Algoa Bay was influenced significantly by year (Wald $X^2(23)=1412.3$, $p<0.001$) and month (Wald $X^2(11)=41.4$, $p<0.001$). The number of active vessels increased from 11.0 (95% CI: 8.4-13.6) to 42.2 (CI: 39.6-44.7) vessels.month^{-1} from 1985 to 1993 (Figure 6.1a), but subsequently declined to 9.4 (CI: 6-8-12.0) vessels.month^{-1} in 2003. Vessel number has remained stable from 2002 onwards ranging from 10.4 (CI: 7.8-13.0) to 15.5 (CI: 12.9-18.1) active vessels.month^{-1} (Figure 6.1a). Vessel number differed significantly between management periods (Kruskal-Wallis ANOVA H(3, 287)=149.71, $p<0.001$) with greater number of vessels active in period A than B, which were both significantly higher than periods C and D (Figure 6.1a insert). Seasonally vessel number was highest from January to August and lower from September to December.

Year (Wald $X^2(23)=403.8$, $p<0.001$) and month (Wald $X^2(11)=78.5$, $p<0.001$) were both significant predictors of fishing effort (boat-days). Annually effort increased from 57.3 (CI: 39.6-75.1) to a maximum of 190.3 (CI: 172.6-208.1) boat-days.month^{-1} from 1985 to 1994 respectively (Figure 6.1b). This was followed by a decrease in effort to 56.3 (CI: 38.6-74.1) boat-days.month^{-1} in 2003 but subsequently increased to 108.5 (CI: 09.7-126.3) boat-days.month^{-1} in 2004. Fishing effort was significantly higher in management periods A and B than periods C and D (Kruskal-Wallis ANOVA H(3, 287)=77.30, $p<0.001$) (Figure 6.1b insert). An increase in the mean boat-days.vessel.$^{-1}$month^{-1} occurred concomitantly with the decline in monthly effort in vessel number and boat-days (Figure 6.1a, 6.1b). Monthly effort decreased from July onwards being lowest from September to December.

Year (Wald $X^2(23)=1139.4$, $p<0.001$), month (Wald $X^2(11)=130.9$, $p<0.001$), area (Wald $X^2(5)=1964.7$, $p<0.001$) and crew size (Wald $X^2(1)=533.0$, $p<0.001$) were all significant predictors of CPUE (reduced dataset) (kg.boat-day^{-1}). CPUE decreased from 278 (CI: 264-291) kg.boat-day^{-1} in 1990 to 118 (CI: 105-133) in 2001 (Figure 6.1c). This was followed by an increase to 293 (CI: 276-311) kg.boat-day^{-1} in 2007. There were seasonal trends in CPUE with peaks from June to August and November to February. CPUE modelled using all data showed a similar trend; however, estimates were lower than the reduced dataset for the first management period, becoming very similar after the initial reduction in number of rights holders (Figure 6.1c). CPUE differed significantly between management periods (Kruskal-Wallis ANOVA $H(3, 17267)=415.87$, $p<0.001$) with all periods being significantly different from each other with highest CPUE in Period D, followed by C and A (Figure 6.1c insert). Annual harvest peaked at 430 tons in 1990 but declined to 120 tons in 2002. Harvest has subsequently increased to over 200 tons per annum from 2004 onwards (Figure 6.1d).

(b) Spatial tends

Analysis of VMS data identified six main fishing grounds in Algoa Bay. Each fishing ground was delimited based in the five-minute grid system in which commercial log book data are recorded by rights holders and reported to DAFF (Figure 6.2). Further spatial analysis was based on these fishing grounds with catch and effort data from the NMLS allocated to one of these fishing grounds using the 5' grid reference. VMS data indicted significantly higher fishing effort on the Cape Recife (CaRe) and Riy Banks (RB) fishing grounds than the Bird Island (BI) grounds, but no differences between other areas (Figure 6.2 insert).

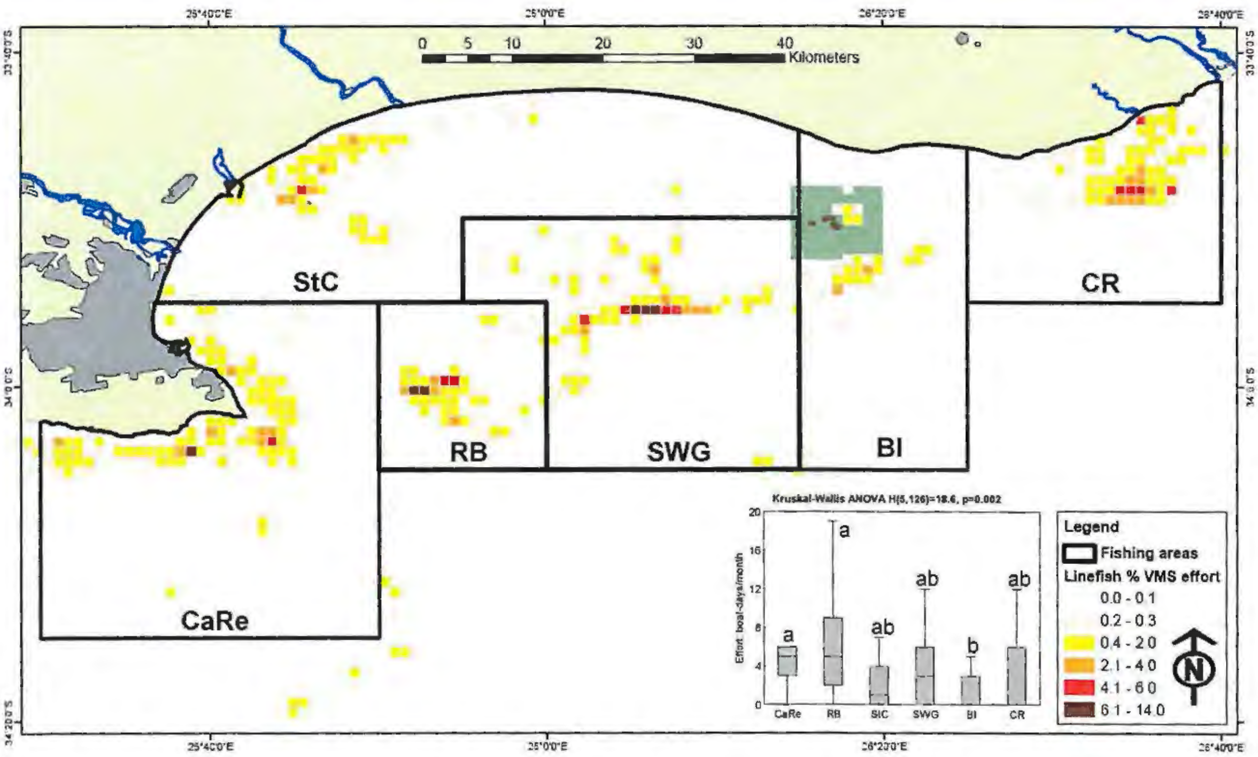


Figure 6.2. Spatial distribution of fishing effort based on analysis of VMS data and the identification of six main fishing areas (CaRe=Cape Recife; RB=Riy Banks; StC=St Croix; SWG=South West Grounds; BI=Bird Island; CR=Cannon Rocks). Differing letters above error bars denote significant differences.

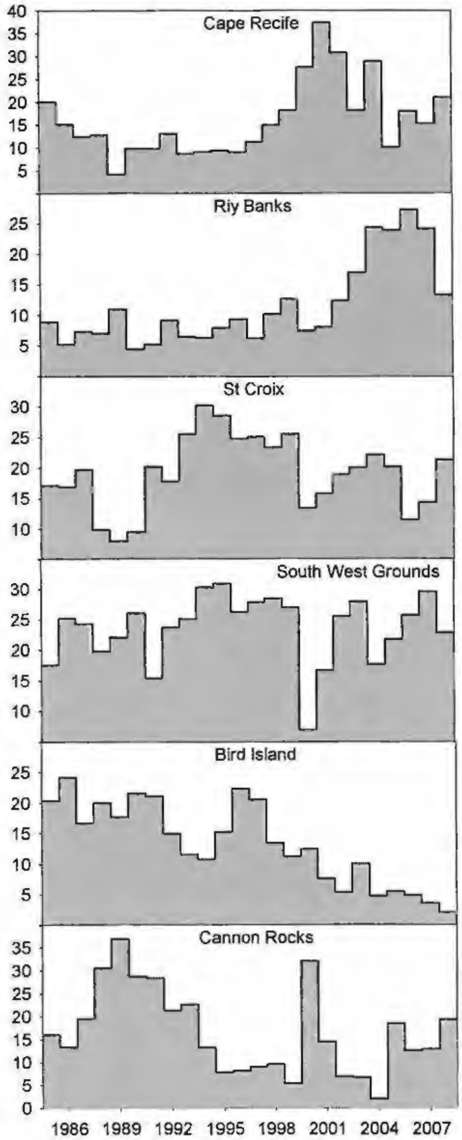


Figure 6.3. Temporal trends in the spatial distribution of commercial linefishing effort in Algoa Bay.

NMLS data indicates long-term changes in the spatial distribution of fishing effort between the six identified fishing grounds (Figure 6.3). The proportion of effort in CaRe was low during the early and mid 1990s but increased considerably from 9% in 1996 to 37% in 2001. The proportion of effort has subsequently declined to 21% in 2008. The proportion of effort in RB followed a similar trend increasing from 12% to 27% between 2001 and 2006 respectively. On the St Croix grounds (StC) the proportion of effort has fluctuated considerably during the history of the fishery, ranging from 8% in 1989 to 30% in 1994. Despite a significant decrease in 2000 the proportion of fishing effort on the South West Grounds (SWG) has been most consistent ranging from 15 to 31%. Effort in BI has declined progressively from 22% in 1996 to 2% in 2008. Similarly effort in Cannon Rocks (CR) decreased from 37% in 1989 to 5% in 1999, but has increased to between 12 and 19% from 2005 and 2008.

Seasonal trends were apparent in the spatial distribution of effort. Monthly effort (boat-days, data 2002-2008) peaked in winter in CaRe and SWG with highest proportional effort in May (28%) and June (35%) respectively (Figure 6.4). Contrarily the proportion of fishing effort in RB and StC was highest during summer, peaking in December (31%) and October/February (27%) respectively. No clear monthly trends in effort were apparent in BI while effort was higher during winter in CR, but lower than other fishing grounds.

Since the allocation of medium-term rights (Management Period C and D, Box 6.1) participation has differed spatially (Wald $X^2(5)=118.9$, $p<0.001$) with significantly fewer vessels fishing BI than all other grounds and fewer vessels in CR than CaRe, RB, StC and SWG (Figure 6.5a). Spatial differences in fishing effort (boat-days.month^{-1}) were also significant (Wald $X^2(5)=147.2$, $p<0.001$) with lowest effort in BI, and effort in CR being significantly lower than in CaRe and SWG areas (Figure 6.5b). Standardised CPUE for the total catch (all species) differed significantly between fishing grounds (Wald $X^2(5)=456.1$, $p<0.001$). CPUE was highest in BI, followed by RB, SWG, StC and CR with CaRe having the lowest CPUE (Figure 6.5c). Total annual catch differed significantly between fishing grounds (Kruskal-Wallis ANOVA $H(5, 42)=21.3$, $p<0.001$). Greatest annual catches were landed from SWG, CaRe, RB and StC. Annual landings in CR were significantly lower than the SWG, and BI landings were significantly lower than RB and SWG (Figure 6.5d).

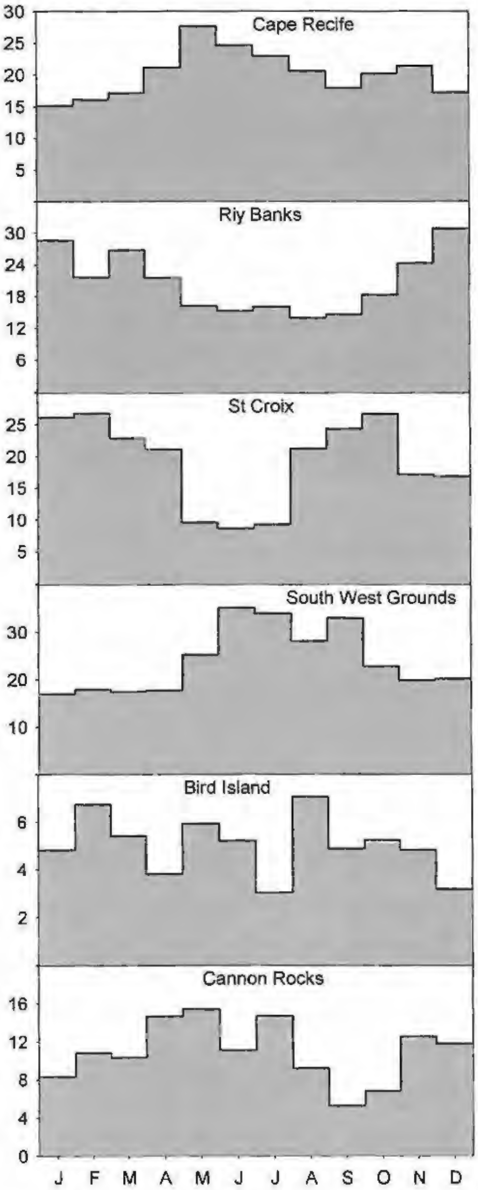


Figure 6.4. Monthly trends in the spatial distribution of commercial linefishing effort in Algoa Bay (data 2002-2008).

(c) Catch composition

The contribution of the major taxa to the annual landings changed temporally (Figure 6.6a). The proportion of sparids increased, initially peaking at 67% of the annual landed weight in 1990 and 1991, but subsequently declined progressively to 31% in 2000. Following a slight peak the lowest contribution to the landed catch occurred in 2004 where sparids only accounted for 23% of the landings. Sciaenids accounted in part for replacing the declining sparid landings with the proportion of landed catch weight increasing from 25% to 66% from 1989 to 2006, but subsequently declined to 42% in 2008. The contribution of scombrids and carangids to the annual landed catch also increased slowly from 1988 accounting for 9% and 10% of the catch weight in 2001 and 2003 respectively. The proportion of elasmobranches in the annual landings increased from 1% in 2000 to 8% in 2008.

The species composition differed significantly between management periods (ANOSIM Global R 0.753, p=0.01) with Period A (1985-1998) differing significantly from all other periods, while no differences existed between periods B, C and D (Figure 6.6b). Geelbek (*Atractoscion aequidens*) contributed most to the differences between management periods, with sharks and some of the sparid species (panga (*Pterogymnus laniarius*), dageraad (*Chrysoblephus cristiceps*), carpenter (*Argyrozona argyrozona*)) also contributing to the observed differences.

Species composition (data from management periods C and D 2002-2008) differed spatially between fishing grounds (ANOSIM Global R 0.401; p<0.01) (Figure 6.6c and 6.6d). Community structure in CaRe and RB, and CaRe and SWG was similar, but community structure in other fishing grounds differed significantly from each other. Sciaenids dominated the landings in CaRe (56%), RB (73%) and CR (76%), while sparids dominated in BI (55%). Sciaenids (46%) and sparids (47%) contributed almost equally to the landed catch in the SWG. The contribution of different taxa to the total landed catch was most evenly distributed in StC where sparids contributed 32%, sciaenids 28%, and elasmobranches 20% to the landed weight. This more even distribution contributed to the StC landings being most different to other areas. Carangidae accounted for 15% of the landed weight in BI but less than 5% in all other fishing grounds. The SIMPER routine indicated that carpenter, geelbek and silver kob (*Argyrosomus inodorus*) were the species which contributed most significantly to the differences between fishing grounds.

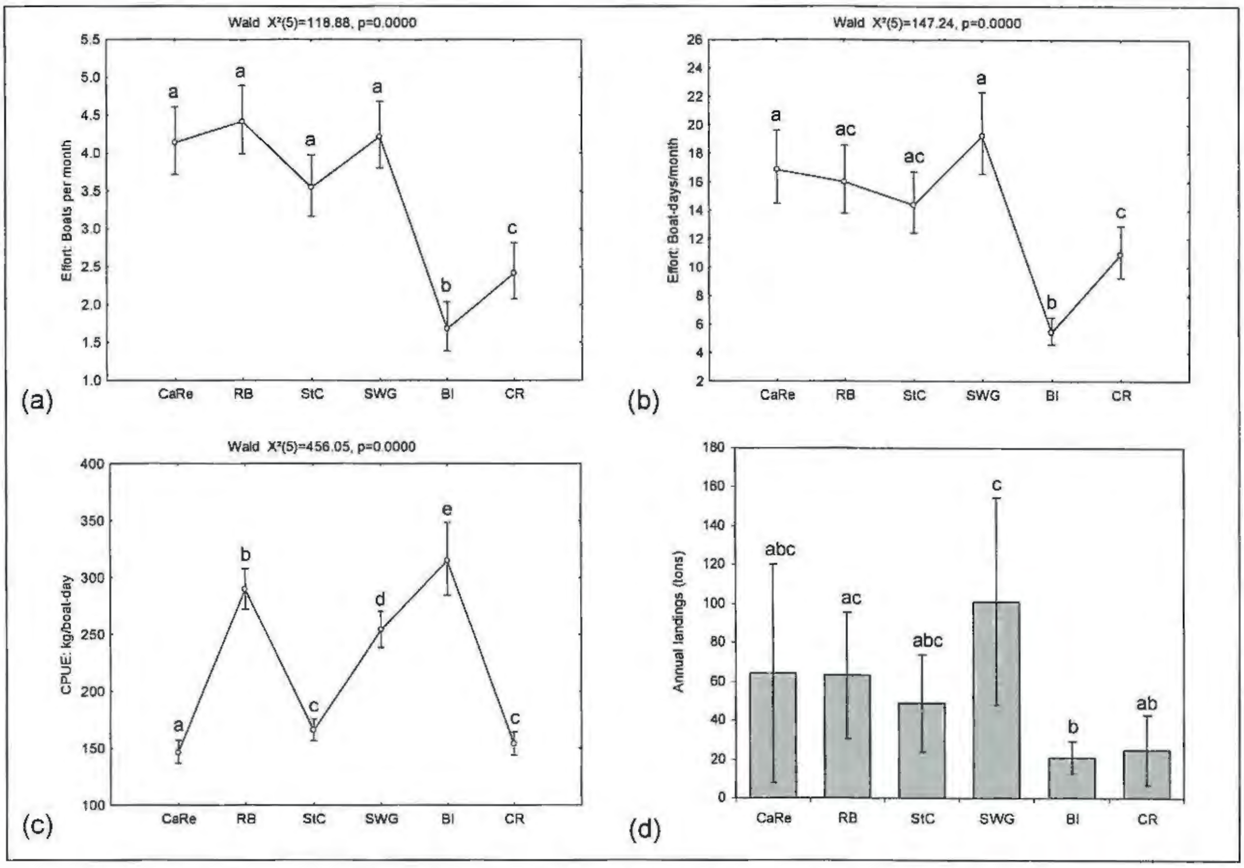


Figure 6.5. Spatial comparison of (a) the number of vessels reporting catches per month, (b) the number of boat-days per month, (c) CPUE and (d) annual landings. Data 2002-2008. Letters above error bars denote significant differences.

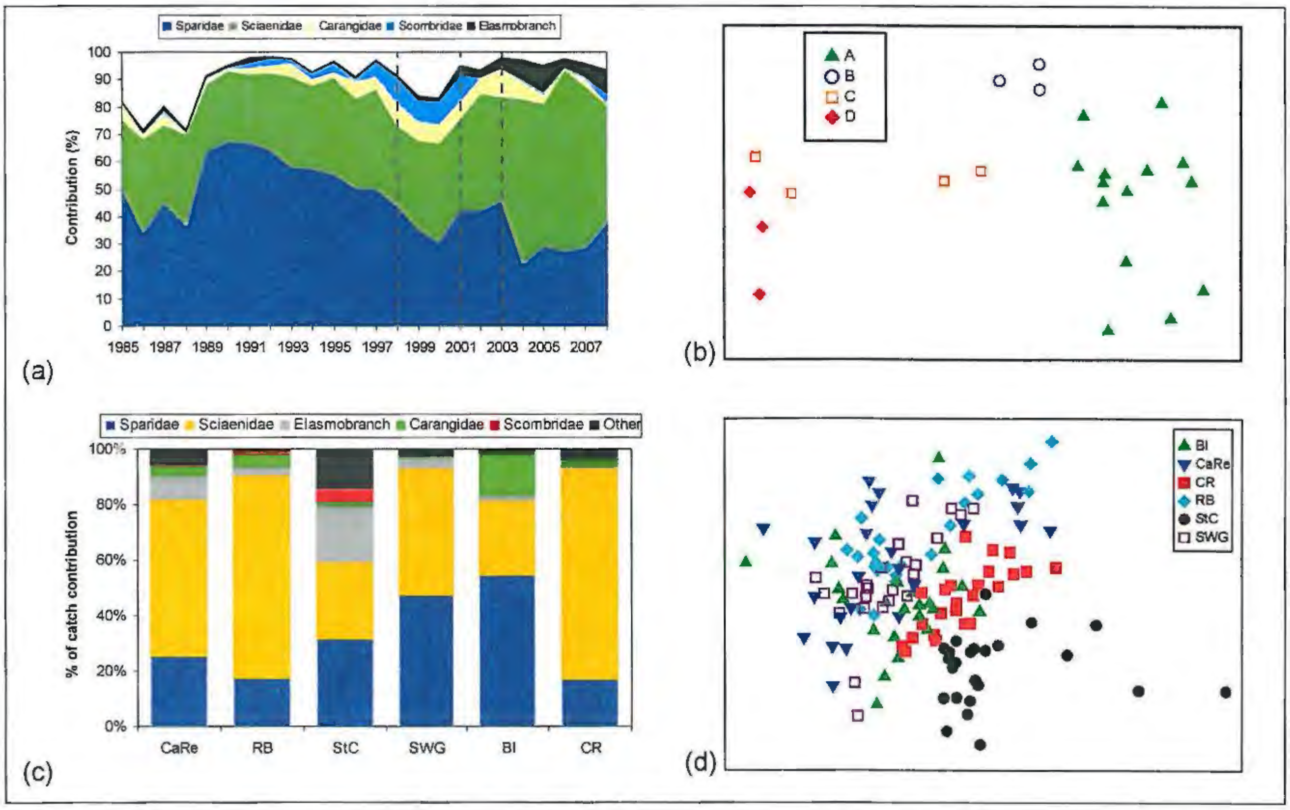


Figure 6.6. (a) temporal trends in the contribution of the dominant taxa to the annual landed catch (dashed lines demarcate management periods), (b) nMDS ordination of the changes in landed catch composition by management period (stress 0.09), (c) composition of the dominant taxa within each fishing area, and (d) nMDS ordination of the differences in species composition between areas (stress 0.21).

Spatial and temporal trends in the contribution of the main target species of the commercial linefishery to the landed catch and standardised CPUE are illustrated in Figure 6.7. Long-term declining trends in the contribution of carpenter, panga and dageraad to the annual landings were evident from the early 1990s to 2008 (Figure 6.7). The proportion of carpenter decreased from 54% in 1990 to 16% in 2004, panga from 12% in 1991 to <1% in 2006, and dageraad from 6% in 1985 to <1% in 2007 and 2008. Similarly carpenter CPUE decreased from 94.6kg.boat-day^{-1} in 1990 to 29.7kg.boat-day^{-1} in 2001, but had risen again to 50.9 kg.boat-day^{-1} in 2008. Panga CPUE decreased from 15.7 kg.boat-day^{-1} in 1991 to 1.5 kg.boat-day^{-1} in 2006, while dageraad CPUE decreased from 3.7 kg.boat-day^{-1} in 1993 to 0 in 2008.

Contrarily the CPUE and proportion of geelbek, santer (*Cheimerius nufar*), elf (*Pomatomus saltatrix*) and elasmobranches increased temporally (Figure 6.7). The proportion of geelbek in the landed catch increased from 3% in 1991 to a maximum of 59% in 2006, santer increased from <1% in 1985 to a maximum of 7% in 2008, elf increased from <2% during the early 1990s to a maximum of 6% in 2008 and elasmobranches increased from <2% prior to 2000 to a maximum of 9% in 2005. Geelbek CPUE increased from 2.3kg.boat-day^{-1} in 1986 to a maximum of 127.8kg.boat-day^{-1} in 2004, santer CPUE increased from 1.1kg.boat-day^{-1} in 1985 to 11.5kg.boat-day^{-1} in 2003, elf CPUE increased from zero catches prior to 1990 to 5.4kg.boat-day^{-1} in 2007 and 2008, while elasmobranch CPUE increased from zero catches prior to 1995 to a maximum of 15.1kg.boat-day^{-1} in 2005.

The proportion of giant yellowtail (*Seriola lalandi*) in the landed catch increased from <1% in 1988 to 10% in 2003 but has subsequently declined to <1% in 2008. Similarly CPUE initially increased from the early 1990s but declined from a maximum of 9.6kg.boat-day^{-1} in 2000 to 0.7kg.boat-day^{-1} in 2008 (Figure 6.7). The proportion of silver kob in the total landed catch declined from 24% in 1996 to 7% in 2004, however, CPUE fluctuated inter-annually and no clear trend was discernable. The proportion of catch and CPUE of roman (*Chrysoblephus laticeps*) initially increased from the early 1990s to 1998 and subsequently declined from 3% and 7.4 kg.boat-day^{-1} in 1998 to <1% and 2 kg.boat-day^{-1} in 2000 and has subsequently varied inter-annually.

Seasonal trends in species landings (data 2002-2008) were evident for carpenter (ANOVA $F(3,80)=6.6$, $p<0.001$), silver kob (ANOVA $F(3, 80)=8.1$, $p<0.001$), geelbek (ANOVA $F(3,80)=3.3$, $p=0.03$) and elf (Kruskal-Wallis ANOVA $H(3,84)=22.9$, $p<0.001$) (Figure 6.7; Table 6.1). The contribution of carpenter and silver kob in the monthly landings peaked in spring, while the proportion of geelbek peaked in autumn, and elf accounted for the lowest proportion of the landings in spring. Carpenter CPUE peaked at 61.9kg.boat-day^{-1} in August and silver kob CPUE peaked at 36.7kb kg.boat-day^{-1} in October. Geelbek CPUE was bimodal peaking at 126.6 kg.boat-day^{-1} in May and 117.2 kg.boat-day^{-1} in December. Elf CPUE peaked in April (3.1 kg.boat-day^{-1}), May (3.4 kg.boat-day^{-1}) and June (3.0 kg.boat-day^{-1}).

There were no significant seasonal trends in the proportion of panga (ANOVA $F(3,80)=0.34$, $p=0.79$), giant yellowtail (Kruskal-Wallis ANOVA $H(3, 84)=6.1$, $p=0.109$), dageraad (Kruskal-Wallis ANOVA $H(3, 84)=0.8$, $p=0.855$, santer (ANOVA $F(3,80)=1.6$, $p=0.208$), roman (ANOVA $F(3,80)=0.9$, $p=0.460$), and elasmobranches (ANOVA $F(3,80)=1.5$, $p=0.230$) in the landings. However, standardised CPUE for panga peaked in February at 6.6 kg.boat-day^{-1} but was relatively constant for the remainder of the year. Similarly dageraad CPUE was higher during summer and autumn, decreasing during winter and spring. The CPUE of elasmobranches was higher during the summer months dropping during winter. No clear seasonal trends in CPUE were apparent for giant yellowtail, santer or roman.

Spatially the proportion of carpenter (ANOVA $F(5,36)=14.5$, $p<0.001$), silver kob (ANOVA $F(5,36)=14.5$, $p=0.01$), geelbek (ANOVA $F(5,36)=37.4$, $p<0.001$), giant yellowtail (ANOVA $F(5,36)=8.2$, $p<0.001$), santer (ANOVA $F(5,36)=13.1$, $p<0.001$), elf (ANOVA $H(5,42)=23.4$, $p<0.001$) and elasmobranches (ANOVA $F(5,36)=16.5$, $p<0.001$) in the annual landed catches differed significantly by area (Figure 6.7; Table 6.1). Carpenter contribution to annual landings was highest in SWG (42%) followed BI (20%) and RB (16%), with these fishing grounds cumulatively accounting for 78% of annual carpenter landings. Carpenter CPUE was highest at SWG (103.7kg.boat-day^{-1}) followed by BI (69.7kg.boat-day^{-1}) and RB (39.5kg.boat-day^{-1}). Silver kob landings were higher in StC (35%) than RB (5%) but did not differ statistically from other fishing grounds. Silver kob CPUE was highest in CR (35.7kg.boat-day^{-1}) followed by BI (33.5kg.boat-day^{-1}). The greatest contribution of geelbek to the landed catch occurred in RB (46%) followed by SWG (20%) and CaRe (15%), while CPUE was highest at RB (176.0kg.boat-day^{-1}) followed by CR (87.1kg.boat-day^{-1}), SWG (68.0kg.boat-day^{-1}) and CaRe (67.9kg.boat-day^{-1}).

Highest proportion of giant yellowtail occurred in RB (42%) and BI (28%), with CPUE being greatest in BI (6.2kg.boat-day^{-1}) followed by RB (8.3kg.boat-day^{-1}), and CaRe (4.0kg.boat-day^{-1}). Santer landings were significantly higher in the StC (53%) than all other areas, while CPUE peaked in RB (11.5kg.boat-day^{-1}) followed by StC (8.0kg.boat-day^{-1}) and CR (7.5kg.boat-day^{-1}). Elf contribution to landed catch was highest in CaRe (28%) and StC (43%) and which had CPUE of 6.7 and 3.0kg.boat-day^{-1} respectively. The cumulative average annual contribution of elasmobranches differed significantly spatially with lower contributions in the BI (3%) and CR (2%) areas than the CaRe (21%), RB (18%), StC (37%) and SWG (19%). CPUE of elasmobranches was highest in the StC area (20.8kg.boat-day^{-1}) followed by CaRe (9.5kg.boat-day^{-1}) and SWG (9.4kg.boat-day^{-1}).

There was no significant spatial difference in the contribution of panga (ANOVA $F(5,36)=1.9$, p0.126), dageraad (ANOVA $H(5,42)=8.6$, $p=0.125$) and roman (ANOVA $F(5,36)=0.6$, $p=0.670$) to the annual landed catch. Panga and dageraad CPUE increased from west to east across Algoa Bay with highest catch rates in CR (6.8 and 0.1kg.boat-day^{-1} respectively). Roman CPUE was highest at BI (4.2kg.boat-day^{-1}).

(d) Data verification

Where NMLS and VMS data coincided fishing effort (boat-days.vessel^{-1}.month^{-1}) reported from the NMLS and recorded by VMS differed significantly (Wilcoxon Matched Pairs test $p<0.001$) between the two data sources. The median boat-days.vessel^{-1}.month^{-1} for the recorded VMS data was 0 (25:75 percentiles: 0-1.5, n=188) and for the reported NMLS data for the same vessels was 6.5 (25:75 percentiles: 4-9, n=188).

Although the overall effort between the two monitoring systems differed significantly the proportion of fishing effort (%boat-days.vessel.$^{-1}$month^{-1}.fishing ground^{-1}) per fishing ground did not differ significantly between the two data sources (Wilcoxon Matched Pairs Test p=0.669).

Comparison of landed catch weight between the NMLS and Observer Programme data for 2008 indicated that the reported and observed landings for the total catch did not differ significantly (Paired t-test p=0.064) with a mean weight of 205±174 and 260±245 reported from the NMLS and Observer databases respectively. Similarly paired t-tests confirmed no significant differences in reported and observed landings for the dominant species carpenter (NMLS 110±173; Observer 104±162; p=0.330), geelbek (NMLS 175±254; Observer 98±123; p=0.052) and santer (NMLS 18±26; Observer 25±146; p=0.719). However, there was a significant difference in the mean weight of silver kob reported and observed (NMLS 59±96; Observer 85±126; p=0.049).

Table 6.1. Statistics for seasonal and spatial comparisons of catch composition in the traditional linefishery. Cells highlighted in green and orange indicate significant differences at p<0.05 and p<0.001 respectively.

Species/Group	Season		Area	
	Test statistic (ANOVA / Kruskal-Wallace ANOVA)	Pairwise comparisons	Test statistic (ANOVA / Kruskal-Wallace ANOVA)	Pairwise comparisons
Carpenter *Argyrozona argyrozona*	F(3,80)=6.6 p<0.001 **	Winter, Spring>Summer, Autumn	F(5,36)=14.5 p<0.001 **	BI, RB, SWG>CR, StC CaRe>CR SWG>CaRe
Silver kob *Argyrosomus inodorus*	F(3,80)=8.1 p<0.001 **	Spring>Summer, Autumn Winter>Autumn	F(5,36)=3.6 p=0.01 *	StC>RB
Geelbek *Atractoscion aequidens*	F(3,80)=3.3 p=0.028 *	Autumn>Spring	F(5,36)=37.4 p<0.001 **	CaRe, RB, SWG>BI RB>CaRe RB, CaRe, SWG>StC RB>CR, SWG
Panga *Pterogymnus laniarius*	F(3,80)=0.3 p=0.800 ns	n/s	F(5,36)=1.9 p=0.126 ns	n/s
Giant yellowtail *Seriola lalandi*	H(3,84)=6.1 p=0.109 ns	n/s	F(5,36)=8.2 p<0.001 **	BI>CR, StC RB>CR, CaRe, StC, SWG
Dageraad *Chrysoblephus cristiceps*	H(3,84)=0.8 p=0.855 ns	n/s	H(5,42)=8.6 p=0.125 ns	n/s
Santer *Cheimerius nufar*	F(3,80)=1.6 p=0.208 ns	n/s	F(5,36)=13.1 p<0.001 **	StC, RB>BI StC>CaRe, CR, RB, SWG
Elf *Pomatomus saltatrix*	H(3,84)=22.9 p<0.001 **	Summer, Autumn, Winter > Spring	H(5,42)=23.4 p<0.001 **	CaRe, StC>CR
Roman *Chrysoblephus laticeps*	F(3,80)=0.9 p=0.460 ns	n/s	F(5,36)=0.6 p=0.670 ns	n/s
Elasmobranches	F(3,80)=1.5 p=0.230 ns	n/s	F(5,36)=16.5 p<0.001**	CaRe, RB, StC, SWG>BI, CR
Other	H(3,84)=14.1 p=0.003 *	Summer>Winter	F(5,36)=11.3 p<0.001 **	CaRe, RB, StC, SWG>BI CaRe, RB , StC, SWG>CR

n/s=not significant
p<0.05 *
p<0.001 **

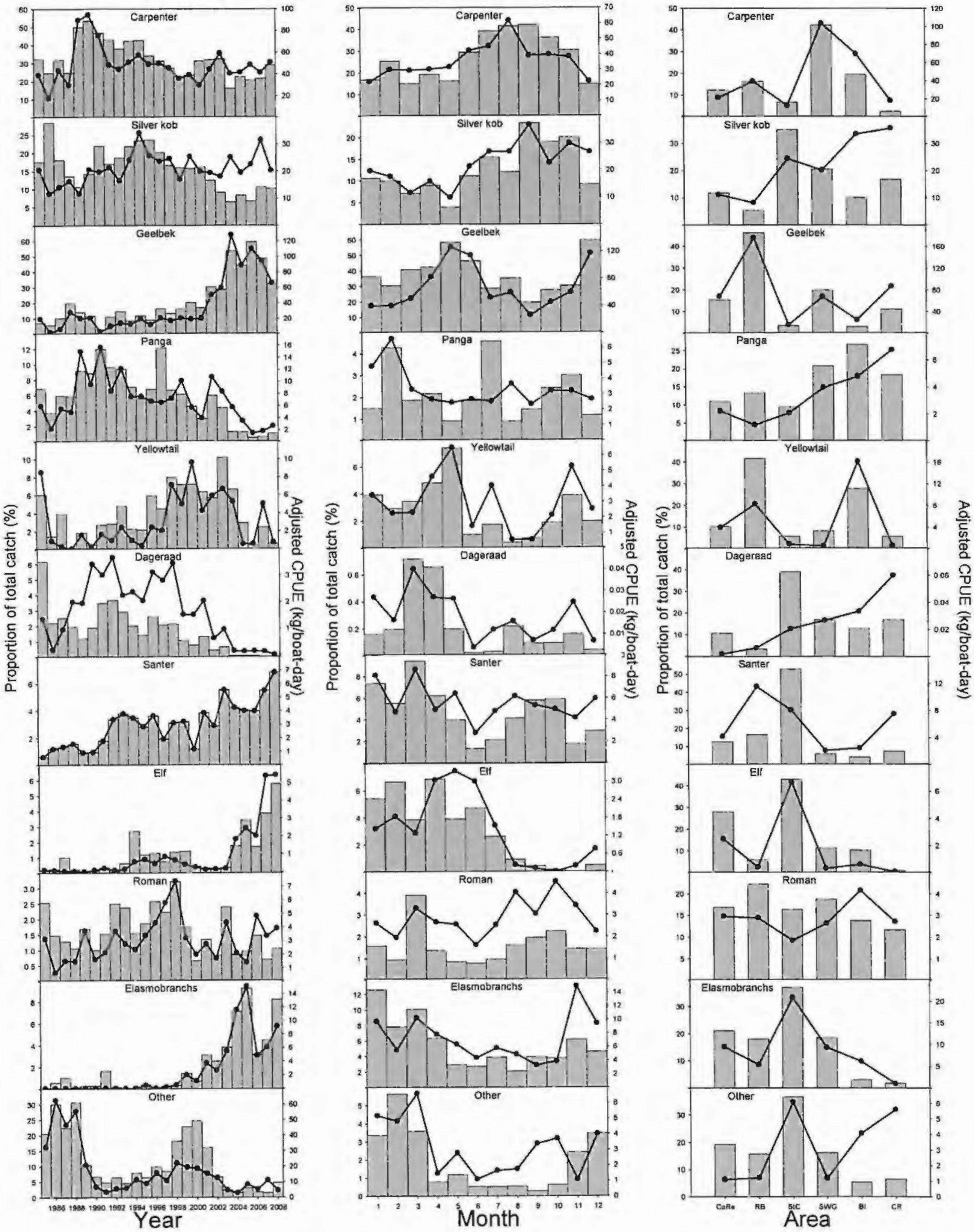


Figure 6.7. Left: Annual trends in the proportion of main species/groups to the total landings (bars) and standardised annual CPUE (line). Centre: Average monthly (2002-2008) proportion of main species/groups to the total landings (bars) and standardised monthly CPUE (line). Right: Average proportion of main species/groups per area to the total annual landings (2002-2008)(bars) and standardised CPUE per area (line).

6.3.2 Chokka-squid jig fishery

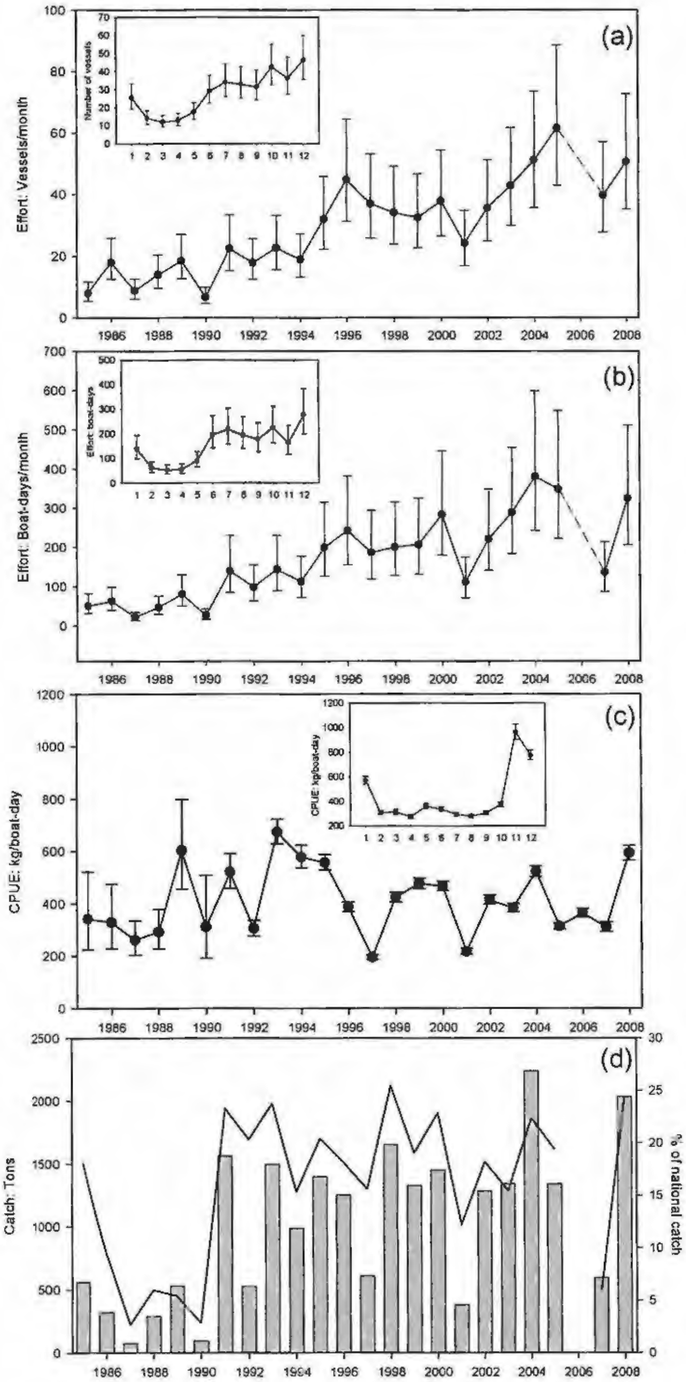


Figure 6.8. Annual and monthly (insert graphs) trends in mean monthly effort in the chokka-squid jig fishery in the Algoa Bay region (a) active vessels, (b) boat-days, (c) catch rate and (d) total landed catch (bars) and percentage of annual national catch (line).

(a) Temporal trends

The participation of chokka-squid vessels in Algoa Bay increased significantly by year (Wald $X^2(22)=233.2$, $p<0.001$) from approximately 8.7 (95% confidence intervals: 6.1-12.6) vessels.month^{-1} in 1985 to a maximum of 61.7 (42.9-88.6) vessels.month^{-1} in 2005 (Figure 6.8a). Similarly fishing effort increased significantly (Wald $X^2(22)=252.9$, $p<0.001$) from 27 (16.9-43.8) boat-days.month^{-1} in 1990 to a maximum of 381.0 boat-days.month^{-1} (242.3-599.0) in 2004 (Figure 6.8b), declining in 2007.

Standardised CPUE fluctuated significantly between years (Wald $X^2(23)=3084.7$, $p<0.001$) (Figure 6.8c). Greater variability in the annual CPUE estimates occurred between 1985 and 1990 due to the lower number of vessels that met the selection criteria for active vessels in the fishery which was used for the analysis of CPUE data resulting in fewer catch records per year during this period.

Annual landed catch in Algoa Bay varied considerably between years ranging from 73.9 tons in 1987 to 2 239.1 tons in 2004 (Figure 6.8d). The proportion of the national catch landed within Algoa Bay ranged from 12 to 25%.

Clear seasonal trends in effort were apparent with vessel number (Wald $X^2(22)=134.0$ $p<0.001$) (Figure 6.8a insert) and fishing effort (boat-days.month^{-1}) (Wald $X^2(22)=137.8$, $p<0.001$) (Figure 6.8b insert), both increasing from May onwards.

Vessel numbers peaked in December at 46.2 (CI: 35.6-60.0) being lowest in March (12.0 CI: 9.2-15.7) (Figure 6.8a insert). Fishing effort (boat-days.month^{-1}) peaks in December with 276.9 (CI: 199.7-383.9) days fished while lowest effort of 51.3 (CI: 36.7-71.7) boat-days occurred in March (Figure 6.8b insert). Monthly trends in catch rate were significant (Wald $X^2(11)=4843.5$, $p<0.001$) being low from February through to October, with peaks in November (962kg.boat-day^{-1}; CI: 901-1028), December (774kg.boat-day^{-1}; CI: 736-814) and January (570kg.boat-day^{-1}; CI: 539-603) (Figure 6.8c insert).

(b) Spatial trends

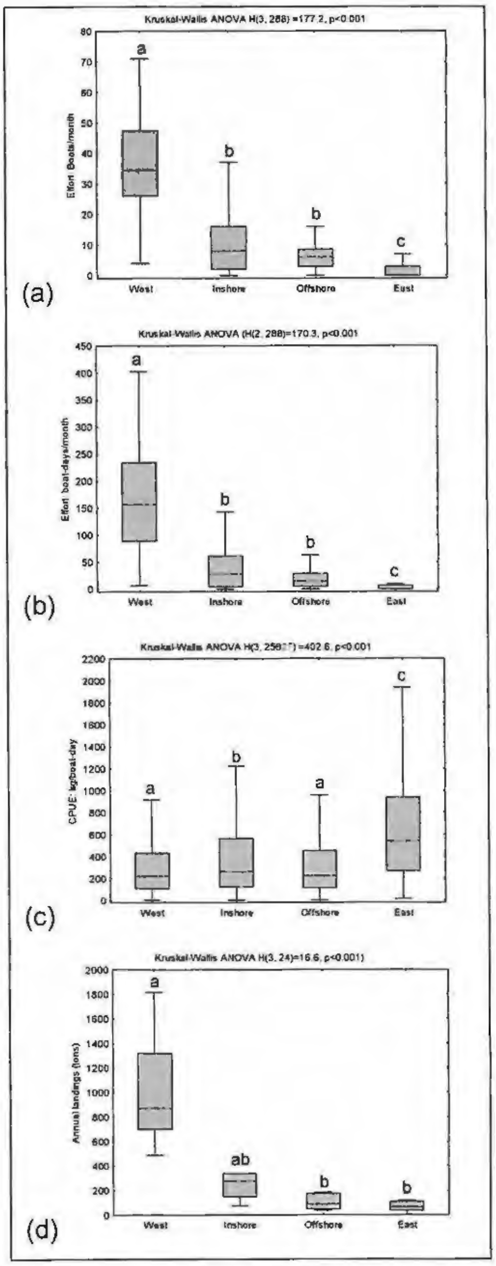


Figure 6.9. Spatial trends in effort and landed catch in Algoa Bay (2002-2008) (a) number of vessels reporting catches per month, (b) boat-days per month, (c) CPUE and (d) annual landings. Differing letters above error bars denote significant differences.

VMS data indicated effort was concentrated in the western region of Algoa Bay and four fishing areas were defined for the fishery (Figure 6.11). Long-term temporal trends were apparent in the spatial distribution of fishing effort within Algoa Bay. The proportion of effort in western region decreased from 96% in 1989 to 38% in 1999 (Figure 6.10). During this period effort shifted predominantly to the inshore region where effort increased from 1 to 39%. This trend reversed again with effort within the western region increasing from 1999 onwards, with a concomitant decrease in effort in the inshore area. The proportion of effort in the offshore and eastern regions was lower ranging from 2-21% and 1-11% respectively. Monthly vessel number (data 2002-2008) differed significantly between areas (Kruskal-Wallis ANOVA $H(3, 288)=177.2$, $p<0.001$) with more vessels fishing the western sector (Median 35.5; upper-lower quartiles: 26.0-47.5) than the inshore (8.0; 2.0-16.0) and offshore (6.0; 3.0-8.5) sectors, while vessel number was lowest in the eastern (0; 0-3.0) sector (Figure 6.9a). Fishing effort (boat-days.month^{-1}) indicated similar spatial trends with significantly more days fished (Kruskal-Wallis ANOVA $H(3, 288)$ 170.3, $p<0.001$) in the western sector (Median 156.5; upper-lower quartiles 88.5-234.0) than the inshore (28.0; 4.0-61.0) and offshore (14.0; 4.5-28.5) sectors, with lowest fishing effort in the eastern sector (0; 0-6.0) (Figure 6.9b).

CPUE differed significantly between areas (Kruskal-Wallis ANOVA $H(3, 25\,267)=402.6$, $p<0.01$) with higher catch rates in the eastern sector (Median 540; upper-lower quartiles 267-940kg.boat-day^{-1}) than the inshore sector (263; 123-564kg.boat-day^{-1}). CPUE was lowest in the offshore (224; 113-453kg.boat-day^{-1}) and western (221; 107-432kg.boat-day^{-1}) sectors (Figure 6.9c).

Annual landings differed significantly between sectors (2002-2008) (Kruskal-Wallis ANOVA $H(3,24)=16.6$, $p<0.001$) with highest landings in the western sector (median 870; upper-lower quartiles 699-1 317tons.year^{-1}) and lower annual landings in the inshore (275; 147-339tons.year^{-1}), offshore (90; 47-175tons.year^{-1}) and eastern (63; 36-109tons.year^{-1}) (Figure 6.9d) sectors.

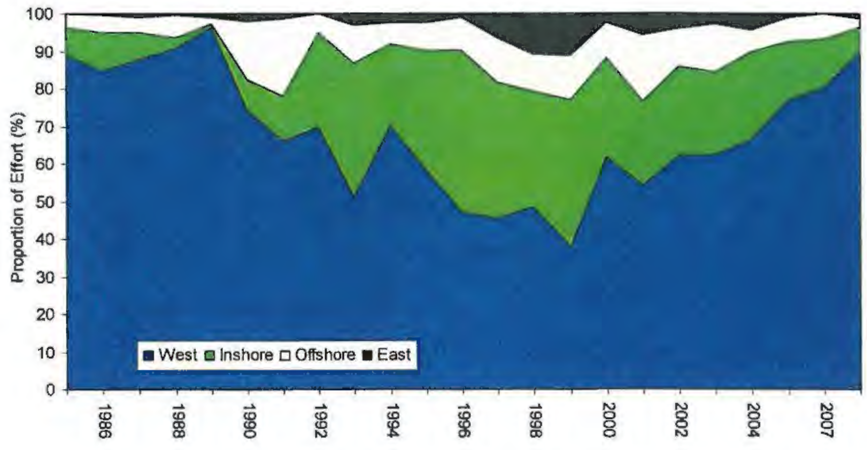


Figure 6.10. Temporal changes in the spatial distribution of fishing effort.

(c) Spatial and temporal trends from VMS data

VMS data were received for 57 vessels for the period from 1 December 2006 to 30 November 2007. Annual effort from these vessels was estimated at 6 495 boat-days along the south and south-east coast of South Africa. Forty-seven (84%) of these vessels were detected within Algoa Bay and the annual fishing effort within Algoa Bay was estimated at 1 533 boat-days, representing 24% of the national effort. Spatially fishing effort (boat-days.month^{-1}) differed significantly within Algoa Bay (Kruskal-Wallis ANOVA $H(3, 48)=29.0$, $p<0.001$) with highest fishing effort occurring in the western sector (Figure 6.11 insert).

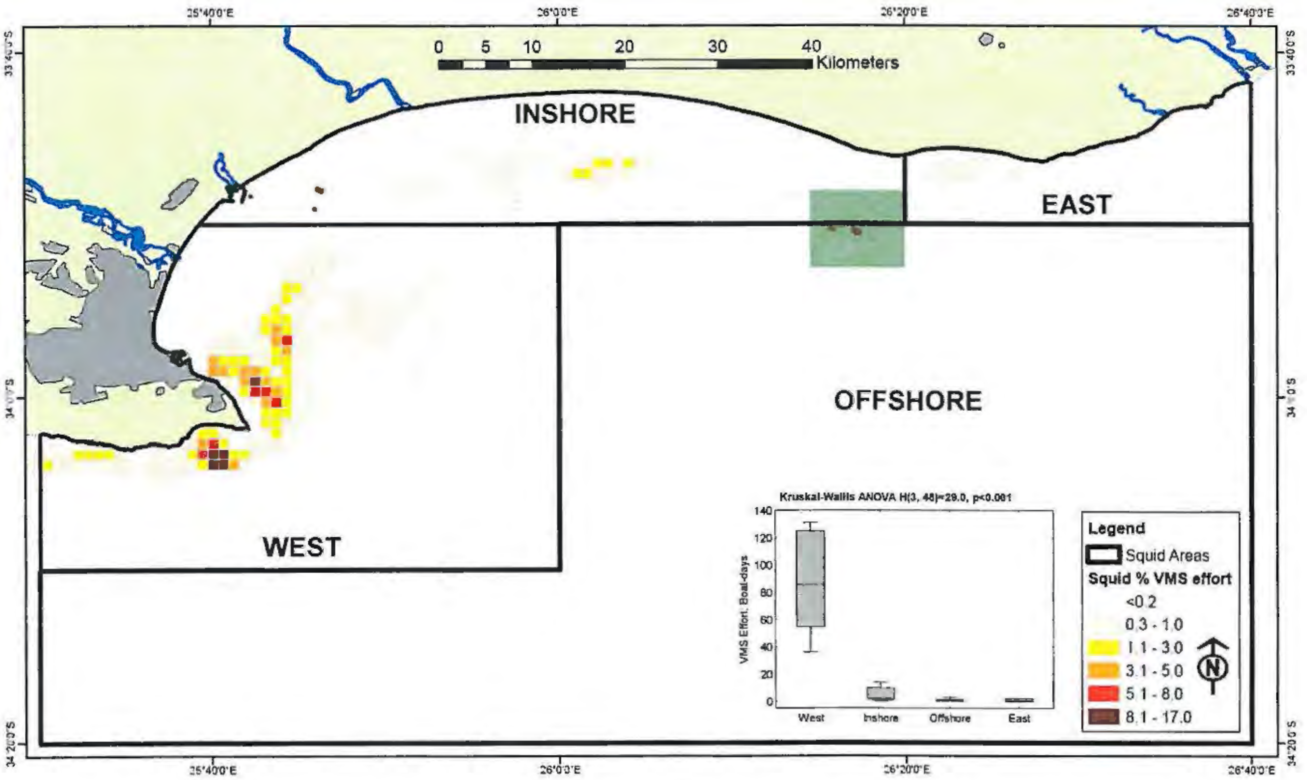


Figure 6.11. Spatial distribution of chokka-squid jig fishing effort (December 2006-November 2007) from VMS data by 1km^2 grid displayed as percent of total estimated effort in Algoa Bay, and the demarcation of four fishing areas used in further spatial analysis.

6.3.3 Small pelagic purse seine fishery

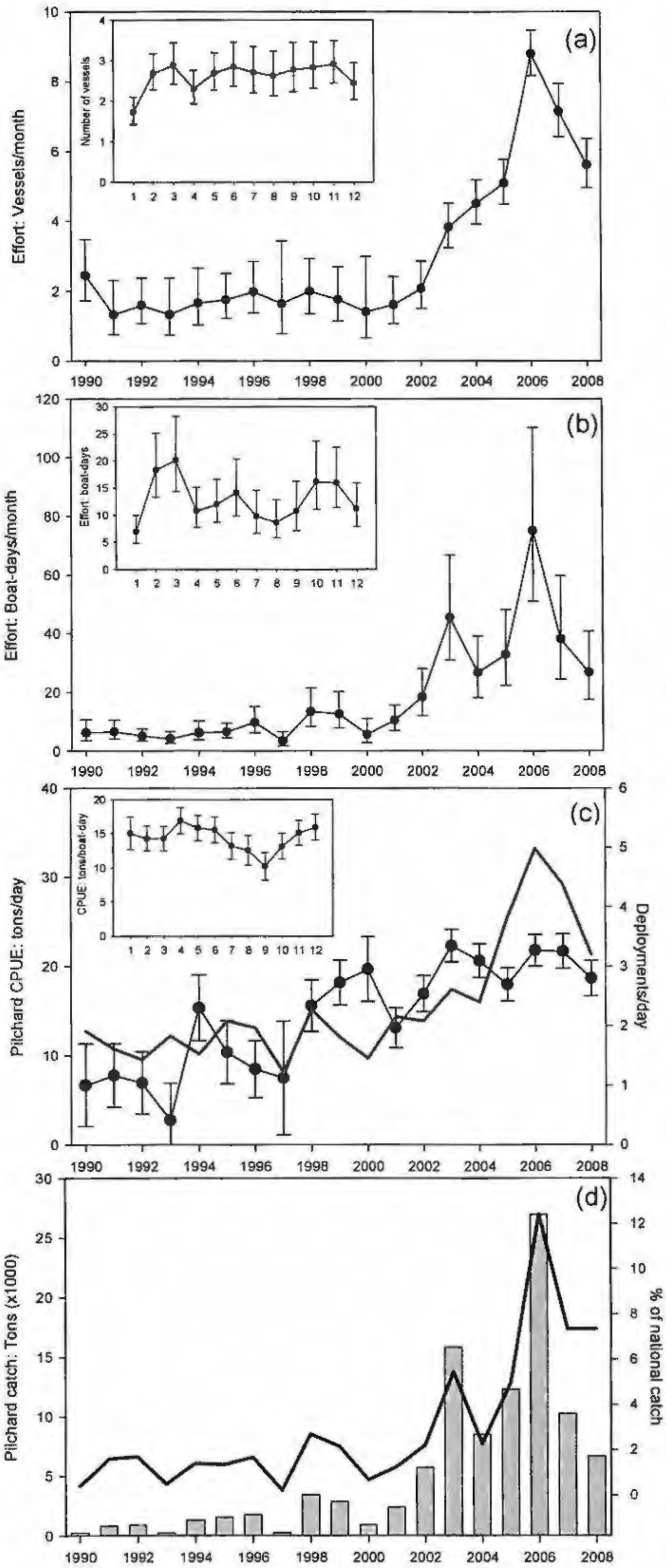


Figure 6.12. Temporal trends in small pelagic purse seine effort and catch in Algoa Bay (a) active vessels.month^{-1}, (b) boat-days.month^{-1}, (c) CPUE and number of gear sets per day (grey line), and (d) total harvest (bars) and percentage of national catch (line).

(a) Temporal trends

Vessel numbers in Algoa Bay differed significantly by year (Wald $X^2(18)=586.4$, $p<0.001$) with a increase from between 1.3 (CI: 0.8-2.3) to 2.4 (CI: 1.7-3.5) vessels from 1990-2002 to a maximum of 8.8 vessels in 2006 (Figure 6.12a). A subsequent decline to 5.6 (CI: 5.0-6.4) vessels in 2008 occurred. Fishing effort (boat-days.month^{-1}) was also influenced significantly by year (Wald $X^2(18)=294.0$, $p<0.001$) with lower effort from 1990 to 2002 ranging from 3.2 (CI: 1.7-6.5) to 18.3 (CI: 12.0-28.1) boat-days.month^{-1} (Figure 6.12b). Thereafter effort increased to a maximum of 75.0 (CI: 51.0-110.0) boat-days.month^{-1}, in 2008 with a subsequent decline to 26.7 (CI: 17.5-40.8) boat-days.month^{-1} in 2008. Vessel number (Wald $X^2(11)=63.3$, $p<0.001$) and fishing effort (boat-days.month^{-1}) (Wald $X^2(11)=34.1$, $p<0$ .001) were both influenced significantly by month with lowest effort occurring in January (1.4 vessels.month^{-1}; 7.0 boat-days-month^{-1}).

Year was a significant predictor of CPUE (Wald $X^2(18)=292.8$, $p<0.001$) with high variability between years (Figure 6.12c). CPUE was initially low but increased to approximately 20 tons.day^{-1} in 1994, but no trend is apparent subsequent to this with CPUE fluctuating between 11 and 22 tons.day^{-1}. The number of times the nets were set increased from 1.1 to 4.6 deployments.day^{-1} in 2006. CPUE (Wald $X^2(11)=112.6$, $p<0.001$) differed significantly by month with lowest catch rates occurring in early spring (10.2 tons.day^{-1}).

Sardine landings within Algoa Bay increased considerably from 2002 onwards peaking at 26 958 tons in 2006 (Figure 6.12d). The Algoa Bay landed catch accounted for 12% of the national landings in 2006 but has subsequently declined to around 7% (Figure 6.12d).

(b) Spatial distribution

Analysis of onboard observer data and VMS data revealed that the spatial distribution of fishing effort of the small pelagic purse seine fishery was highly dispersed (Figure 6.13). Due to the nature of the fishery, the influence of environmental conditions and the location of target shoals no fishing grounds were differentiated for further spatial analysis. VMS data indicated a greater amount of effort in the inshore central region of Algoa Bay in comparison to the observer data which indicated catch locations were concentrated off the harbour entrance and west of the Cape Recife headland. However, the observer dataset was small (n=78 observer days over a seven-year period) possibly accounting for the spatial discrepancy between the two data sources. Similarities in the spatial distribution of effort in close proximity to the Port Elizabeth harbour (see arrow in figure) suggest that data from the two sources corresponds and that the rules used to select VMS data portray fishery activity well. Furthermore comparison of observer and VMS records for the same days (limited to four days where records overlapped) indicated a high degree of accuracy between the observed locations of fishing activity and the positions estimated from the VMS data, with direct overlap on three of the sampling days (Figure 6.14).

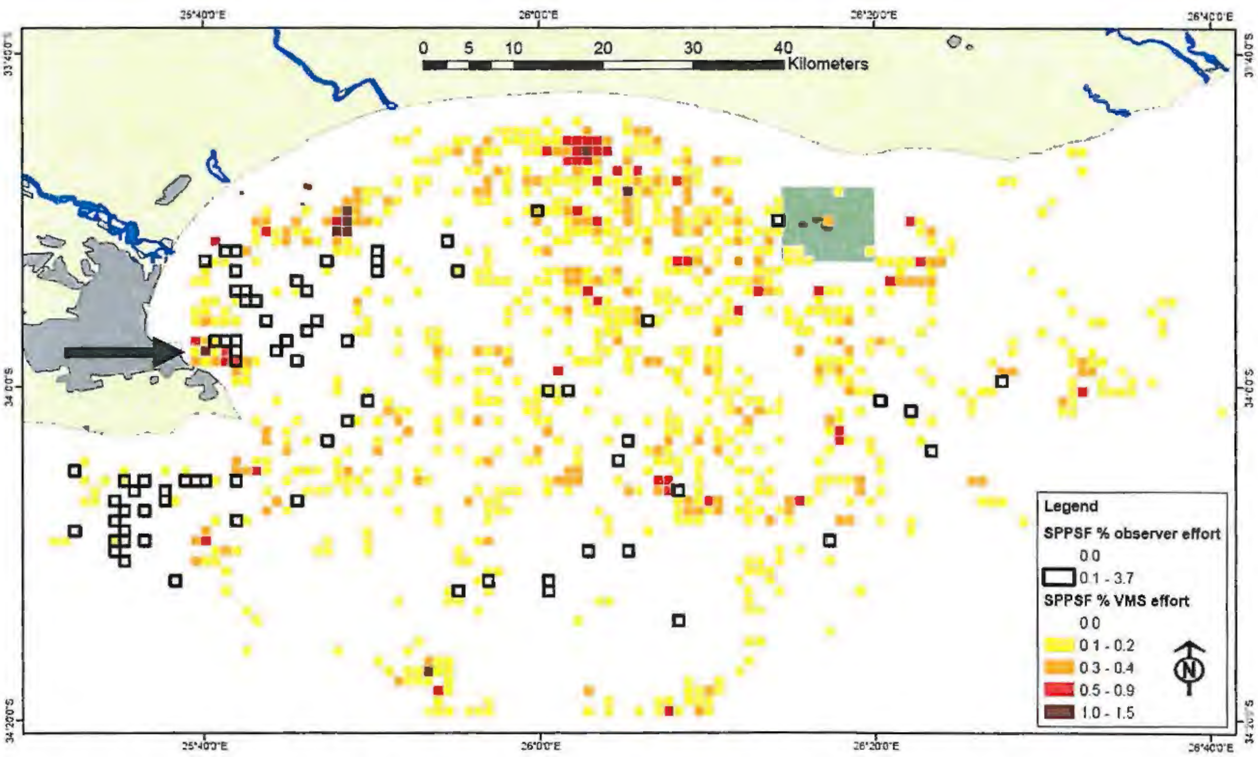


Figure 6.13. Spatial distribution of SPPSF catch locations inferred from VMS data (graduated colour) and reported by onboard observers (black border).

(c) Catch composition

The composition of the landed catch was dominated by sardine, which comprised between 42 and 100% of the landed weight by year (Figure 6.15). Mackerel (*Scomber japonicus*) was the only other species that contributed significantly to the annual landings with a maximum contribution of 49% in 1994, while Cape horse mackerel and redeye each contributed a maximum of 12 and 8% to the annual weight respectively. Anchovy accounted for less than 1% of the annual landed catch in Algoa Bay between 1990 and 2008.

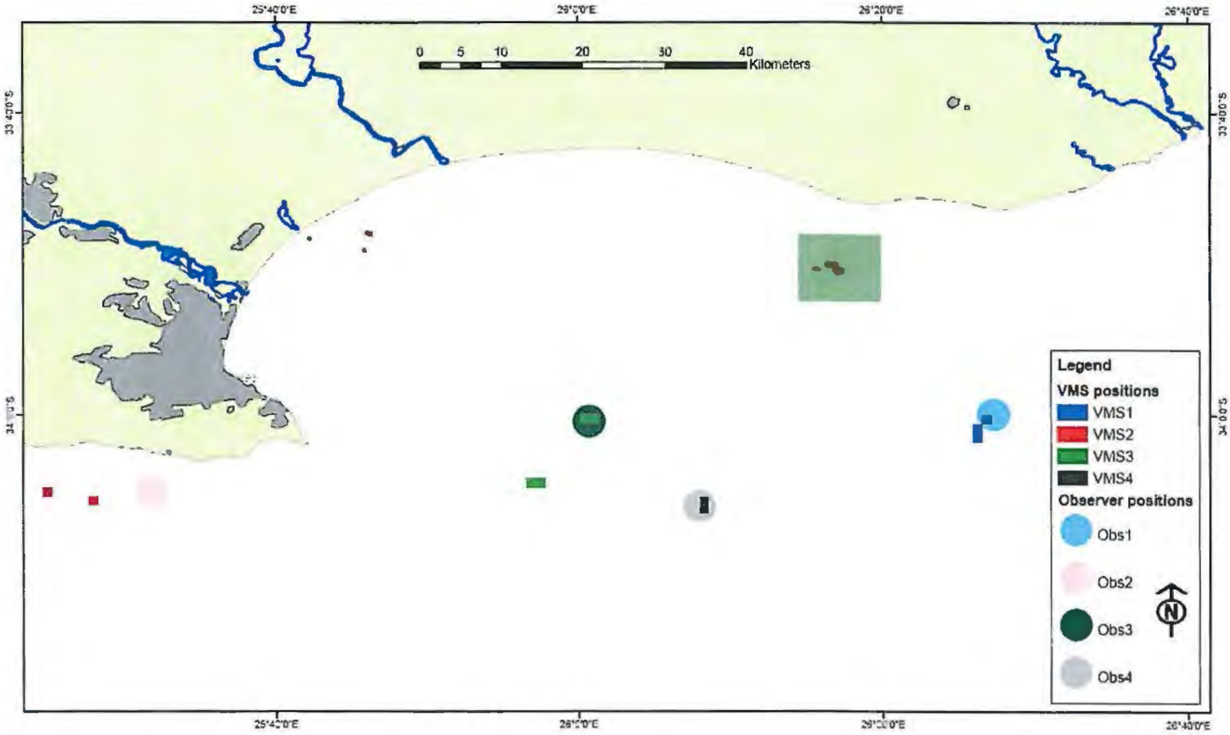


Figure 6.14. Comparison of the locations of corresponding observer and VMS data. Large circles indicate reported observer catch locations, squares indicate the catch locations inferred using VMS data for the same vessel on the same day. Colours and numbers 1-4 indicate the corresponding days from each dataset.

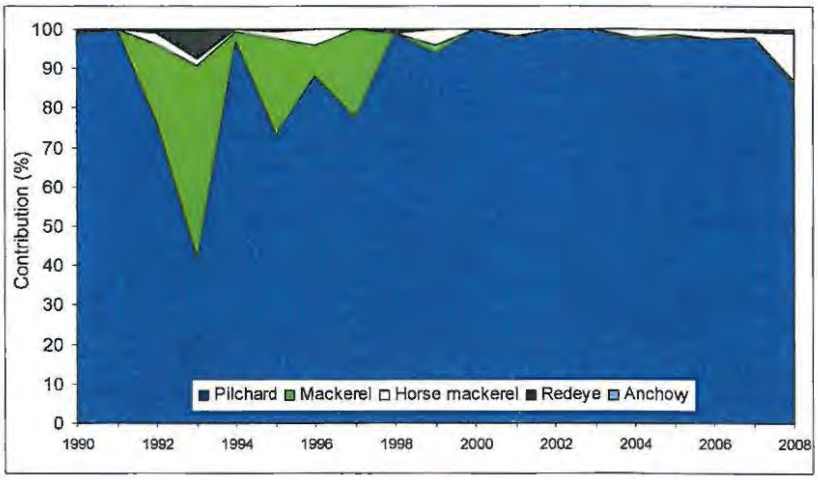


Figure 6.15. Temporal composition of the SPPSF landings in Algoa Bay.

(d) Validation

Of the 78 boat-days observed and recorded by the onboard fisheries monitor, 61 (78%) corresponding entries occurred in the submitted logbook data. There was no significant difference in the observed and reported weights where corresponding entries existed (p=0.443, n=61). However, only 53% of the catch locations were reported in the same 10' grid as recorded by the fisheries observer, indicating poor spatial accuracy of the reported logbook data.

6.3.4 Inshore demersal trawl fishery

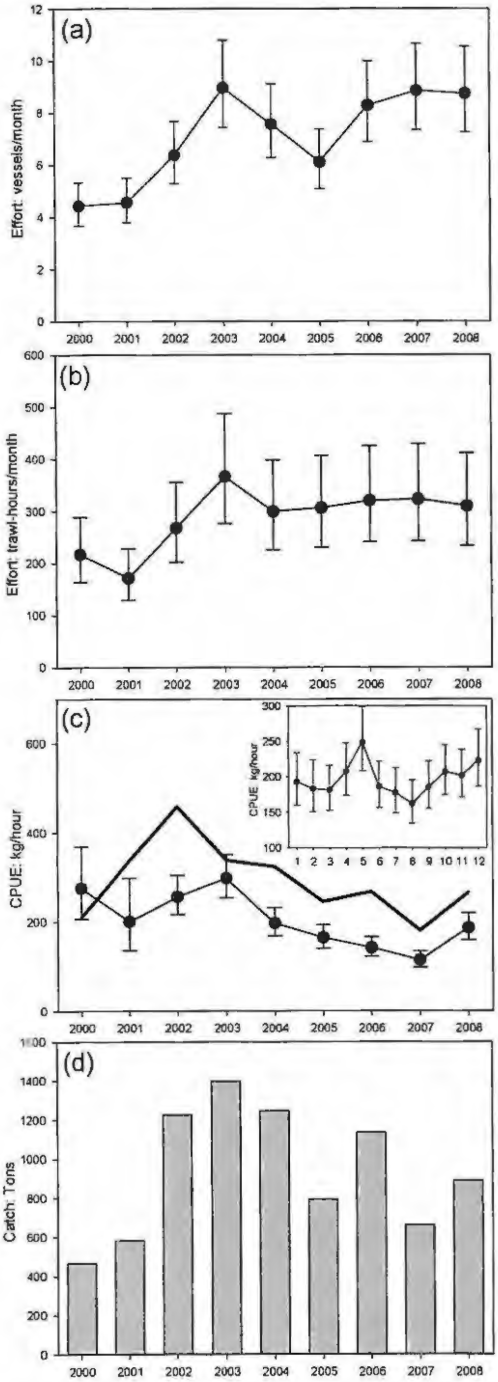


Figure 6.16. Temporal trends in effort, catch rate and harvest in the demersal trawl fishery in Algoa Bay, (a) number of active vessels, (b) effort in trawl-hours, (c) standardised and nominal (thick solid line) CPUE (insert shows monthly trends), and (d) total annual harvest.

(a) Temporal trends

Inshore demersal trawl vessel numbers differed significantly between years (Wald $X^2(8)=68.5$, $p<0.001$) increasing from 4.4 (CI: 3.7-5.3) in 2000 to 8.8 vessels.month^{-1} (CI: 7.3-10.7) in 2007 (Figure 6.16a). Monthly trends in vessel number were not significant (Wald $X^2(11)=13.5$, $p=0.230$).

Fishing effort differed significantly between years (Wald $X^2(8)=21.1$, $p=0.007$) increasing from 171.4 (CI: 128.9-227.6) in 2001 to a peak of 366.5 (CI: 275.9-486.8) trawl-hours.month^{-1} in 2003 (Figure 6.16b) but has remained consistent subsequently. Monthly trends in fishing effort were not significant (Wald $X^2(11)=17.3$, $p=0.098$).

Standardised CPUE of the total landed catch (all species) differed significantly between years (Wald $X^2(8)=270.7$, $p<0.001$) declining steadily from 298 (CI: 254-350) in 2003 to 114 (CI: 97-134) kg.trawl-hour^{-1} in 2007 (Figure 6.16c). Monthly differences in CPUE were significant (Wald $X^2(11)=40.2$, $p<0.001$) peaking in May (249 kg.trawl-hour^{-1} CI: 208-298) and being lowest during August (162 kg.trawl-hour^{-1} CI: 135-195) (Figure 6.16c insert). Nominal CPUE (thick solid line Figure 6.16c) indicated higher catch rates but showed a similar declining trend from 2002 onwards. Total annual catch landed within Algoa Bay showed a general decline from 2003 onwards decreasing from 1 398 to 889 tons in 2008 (Figure 6.16d).

Spatially commercial vessels reported landings from five 20' grid cells (Figure 6.18) within Algoa Bay. Vessel numbers fishing each grid differed significantly (Kruskal-Wallis ANOVA H(4, 540)=233.6, $p<0.001$) (Figure 6.17a). Grid 2 was fished by significantly fewer and Grid 5 by significantly more vessels than all other grids. Similarly effort in trawl-hours per month differed significantly by area (Kruskal-Wallis ANOVA H(4, 540)=342.2, $p<0.001$) with significantly higher fishing effort in the eastern region in Grid 5, followed by Grid 4 (Figure 6.17b). Standardised CPUE was significantly lower in Grid 5 in the eastern region of Algoa Bay than all other trawl grids (Wald $X^2(4)=41.9$, $p<0.001$) (Figure 6.17c). Mean annual landed catch was significantly higher in trawl grids 3 and 5 in the central and eastern sectors (Kruskal-Wallis ANOVA H(4,44)=30.4, $p<0.001$) (Figure 6.17d).

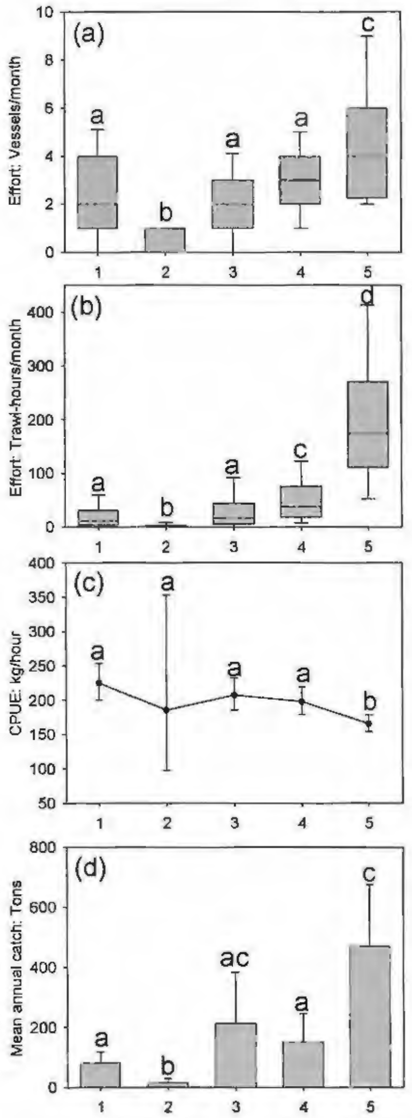


Figure 6.17. Spatial distribution (trawl grids 1 to 5) of (a) the number of active vessels, (b) effort in trawl-hours, (c) standardised CPUE, and (d) mean annual catch in the inshore demersal trawl fishery in Algoa Bay. Differing letters above error bars denote significant differences.

(b) Catch composition

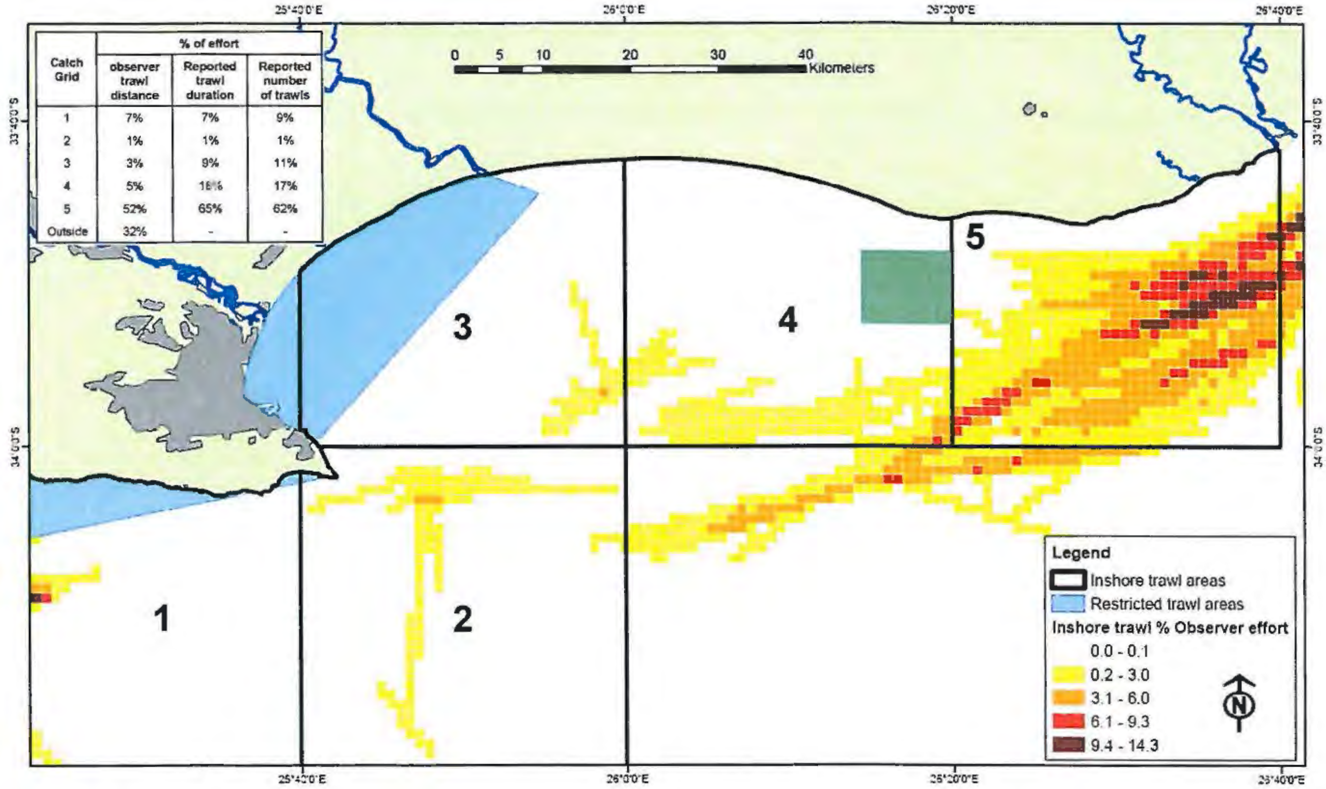
With the exception of 2001 and 2002 the hakes (Merlucciidae) dominated the inshore demersal trawl landings within Algoa Bay (Figure 6.19a) with the percentage contribution to the overall landed catch weight ranging from 24% in 2001 to 57% in 2008. Horse mackerel (Carangidae) was the second most important species in Algoa Bay contributing between 16 and 65% to the annual landed catch. Sparids accounted for 15 and 11% of the landed catch weight in 2002 and 2003 respectively, while accounting for less than 1% in all other years. The contribution of elasmobranches to the annual landed catch prior to 2004 was low (<1%) but ranged from 7-13% from 2004 onwards. The contribution of cephalopods and sciaenids ranged from 1-4% and 1-3% respectively.

Multivariate statistics indicated significant differences in community structure between trawl grids; however, the magnitude of the effect was small (ANOSIM Global R=0.099, p<0.001) (Figure 6.19b). The most significant effects were observed between trawl grids 1 and 3 (r=0.149), 3 and 5 (r=0.139) and 1 and 4 (r=0.115).

(c) Results from onboard observer data

Onboard fisheries monitors observed a total of 553 trawls within Algoa Bay between 2003 and 2008 (Table 6.2). The average trawl distance between recorded start and end points was 12.4km and the majority of effort was concentrated within the eastern region of Algoa Bay within Grid 5. Fifty-two percent of the observed effort was exerted in Grid 5, while 32% of the observed trawl distance (meters) was outside the boundaries of the grids in which catch was reported by the vessel skippers (Figure 6.18 insert table).

A comparison of the annual proportional composition of all dominant taxa caught within Algoa Bay from the observer data and commercial reported landings indicated no significant differences (Wilcoxon Matched Pairs Test, p=0.462). However, on comparison of individual taxa differences were apparent for the carangids (p=0.028), sparids (p=0.046), cephalopods (p=0.027) and other teleosts (p=0.028) (Table 6.2).



Catch Grid	% of effort		
	observer trawl distance	Reported trawl duration	Reported number of trawls
1	7%	7%	9%
2	1%	1%	1%
3	3%	9%	11%
4	5%	18%	17%
5	52%	65%	62%
Outside	32%	-	-

Figure 6.18. Spatial distribution of trawl effort per km^2 as a percentage of days fished using onboard observer data. Trawl grids 1 to 5 used in the spatial analysis are also illustrated. Insert table indicates proportional distribution of effort per 20' grid cell from observer and reported data.

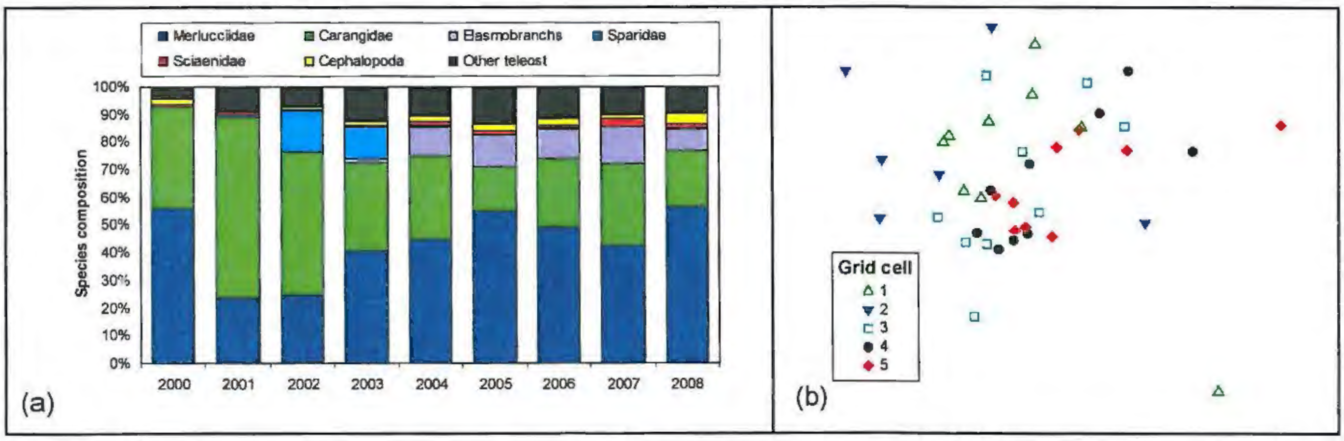


Figure 6.19. (a) Temporal and (b) spatial trends in community composition (stress 0.14) of the commercial inshore demersal trawl landings.

Table 6.2. Annual contribution (%) of major taxa to the total catch from observer data and reported landings. Cells highlighted in orange indicate significant differences between data sources.

	Observer catch data						Reported landings						Wilcoxon test
Year	2003	2004	2005	2006	2007	2008	2003	2004	2005	2006	2007	2008	p-value
Number of trawls (n)	63	51	34	105	19	178	4 741	6 622	4 797	6 474	5 057	5 761	
Chondrichthyes Total	**25**	**12**	**37**	**9**	**4**	**8**	**1**	**11**	**12**	**11**	**13**	**7**	**0.463 ns**
Dasyatidae	1	0	5	0	0	1							
Rajidae	6	3	5	3	1	2							
Squalidae	9	5	20	3	3	3							
Other chondrichthyes	9	3	7	4	0	2							
Teleostei Total	**75**	**87**	**62**	**89**	**95**	**89**	**97**	**87**	**86**	**87**	**85**	**89**	**0.917 ns**
Merlucciidae (hake)	35	46	37	68	54	56	40	45	55	49	42	57	0.753 ns
Ophidiidae (kingklip)	1	1	0	0	0	4	4	1	1	1	1	2	0.345 ns
Carangidae (horse mackerel)	4	17	6	5	9	6	32	30	16	25	30	20	0.028 *
Sciaenidae total	8	3	2	2	2	1	1	1	2	1	3	1	0.249 ns
Baardman	*1*	*0*	*0*	*0*	*0*	*0*							
Geelbek	*0*	*1*	*0*	*2*	*1*	*0*							
Argyrosomus sp.	*7*	*1*	*2*	*1*	*0*	*0*							
Soleidae	3	1	3	2	1	2	4	2	2	1	2	2	0.116 ns
Sparidae total	11	12	5	4	27	12	11	1	0	0	0	0	0.046 *
Blue Hottentot	*1*	*0*	*0*	*0*	*0*	*0*							
Panga	*6*	*12*	*5*	*3*	*27*	*11*							
Red tjor-tjor	*2*	*0*	*0*	*0*	*0*	*0*							
Carpenter	*0*	*0*	*0*	*0*	*0*	*1*							
Rhabdosargus sp.	*2*	*1*	*0*	*0*	*0*	*1*							
Triglidae	13	7	4	7	2	8	1	3	4	3	4	4	0.116 ns
Other teleost	1	1	5	1	0	2	4	4	5	6	3	2	0.028 *
Cephalopoda Total	**1**	**1**	**1**	**1**	**0**	**3**	**1**	**2**	**2**	**2**	**2**	**4**	**0.027 ***

6.3.5 Demersal shark longline fishery

Table 6.3. Annual trends in effort, catch rate and landed catch composition in the demersal longline fishery.

Year	2006		2007	
Effort: Boat-days	38		50	
Effort: Hooks	26 655		29 600	
CPUE kg.hook-hour^{-1}	0.680±3.58		0.782±3.18	
Species	**kg**	**%**	**kg**	**%**
Soupfin sharks	5 892	26	2 108	6
Smooth-hound	7 895	35	14 008	41
Copper shark	3 051	14	8 992	26
Gully shark	375	2	772	2
Hammerheads	1 112	5	4 719	14
Skates	265	1	2 383	7
Other	3 839	17	1 563	5
TOTAL LANDED CATCH	**22 428**		**34 543**	

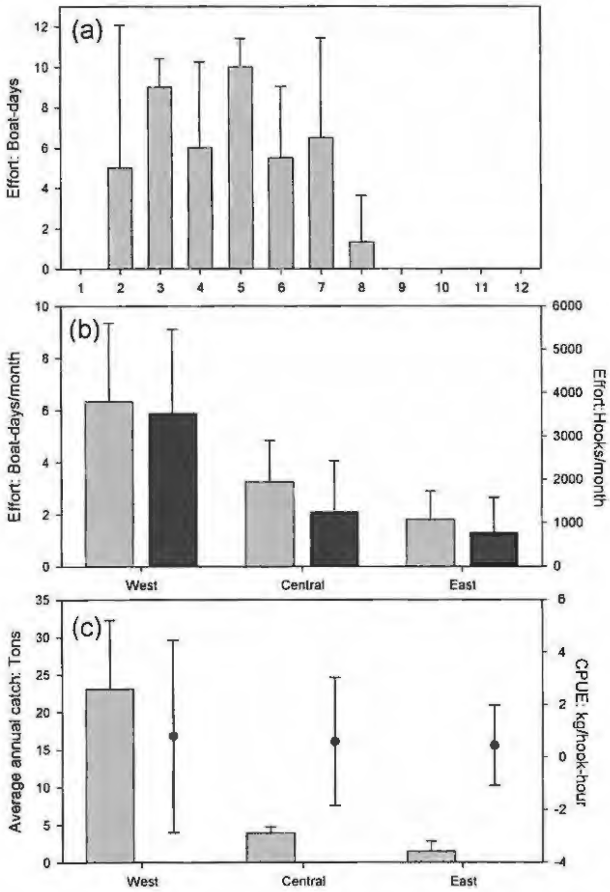


Figure 6.20. Monthly trends in fishing effort (top), spatial differences in monthly effort (middle) in boat-days (grey bar) and number of hooks (black bar), and annual catch (bar) and CPUE (bottom). Error bars denote standard deviation.

Only one vessel was active within Algoa Bay on an annual basis. Monthly fishing effort did not differ significantly between years (Mann-Whitney U tests: Boat-days p=0.387; Hooks p=0.644) with a total of 38 and 50 boat-days and 26 655 and 29 600 hooks fished in 2006 and 2007 respectively (Table 6.3). The total annual landings increased from 22 428kg in 2006 to 34 543kg in 2007 (Table 6.3).

Seasonal trends in effort were apparent with fishing occurring over the autumn and winter months with no fishing in summer (Figure 6.20a). CPUE did not differ by year (Mann-Whitney U, p=0.745) or month (Kruskal-Wallis ANOVA H(6, 529)=8.4, p=0.21) with a mean CPUE of 0.680±3.58 and 0.782±3.18 kg.hook-hour^{-1} in 2006 and 2007 respectively.

The catch composition was dominated by smooth-hound sharks (*Mustelus sp*) which comprised between 35 and 41% of the landed catch by weight. Soupfin and bronze whaler (*Carcharhinus brachyurus*) sharks were the second and third most important species in the landed catch comprising between 6-26% and 14-26% of the landed weight respectively (Table 6.3).

Spatially effort differed significantly with a higher number of boat-days.month^{-1} in the western than eastern region (Kruskal-Wallis ANOVA H(2, 25)=10.5, p=0.005) (Figure 6.20b), while a greater number of hooks were set in the western than both the central and eastern regions per month (Kruskal-Wallis ANOVA H(2, 25)=11.3, p=0.004). Fishing effort was distributed widely in depths shallower than the 50m isobath (Figure 6.21).

Average annual landings were higher in the western region (23 098±9 154kg) than the central (3 928±810kg) or eastern regions (1 460±1 398) (Figure 6.20c). However, CPUE did not differ spatially (Kruskal-Wallis ANOVA H(2, 529)=0.7, p=0.713) (Figure 6.20c). VMS polling intervals for the demersal shark longline vessel was set at six hours, precluding the use of this data for spatial analysis as vessel activity could not be distinguished at this polling frequency.

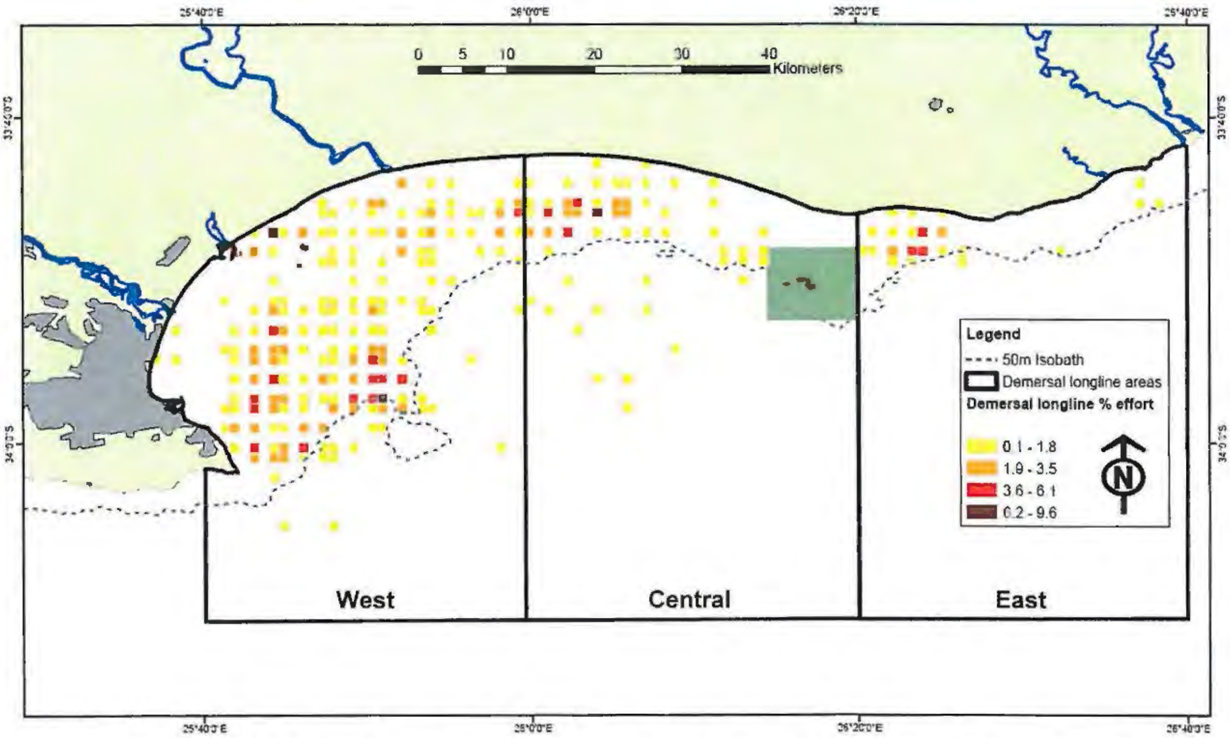


Figure 6.21. Spatial distribution of the proportion of demersal shark longline fishing effort in Algoa Bay. Dashed line indicates the 50m isobath.

6.3.6 Index of relative commercial importance (IRCI)

The IRCI integrates all commercial sectors into one spatial index which provides a valuable means to integrate fisheries data into future spatial planning and management within Algoa Bay. An important aspect of the index is that it takes into account the importance of Algoa Bay to each commercial sector based on the proportion of effort occurring locally within the bay relative to the national fishery. In doing so it awards localised fisheries that are more dependent on the resources in Algoa Bay a greater weighting than those which have access to alternative fishing grounds and fish over larger geographic areas. This is important for future planning in Algoa Bay as certain sectors are limited to fishing within close proximity of Algoa Bay due to the small sizes of vessels used or the natural distribution of the target stocks. Localised spatial closures would therefore have greater impact on these fisheries, hence the need for greater consideration of their fishing activities than that of other sectors which have access to alternative sites. It is evident from the IRCI that fishing effort is widely and heterogeneously distributed across Algoa Bay. However, certain areas of particular importance to the commercial fisheries are evident. These include the Cape Recife area, the St Croix Islands, Riy Banks, the South West Grounds and towards the east of the study area (Figure 6.22). Although these areas are important fisheries areas it is likely that they coincide with areas of biological importance and a systematic method is required to balance the requirements for conservation with those of the socio-economic requirements when planning future spatial management initiatives. This is dealt with in the following chapter.

6.3.7 Economic index of relative commercial importance

Of the commercial sectors which operate in Algoa Bay, the small pelagic trawl fishery is of greatest economic value nationally with an estimated annual value of ZAR614 million (Anon 2004) (Table 6.4). This is followed by the commercial linefishery (ZAR346 million), chokka-squid (ZAR190 million) and the demersal inshore (ZAR95 million) sectors (Anon 2004) (Table 6.4). No published catch or economic information was available for the demersal shark longline fishery nationally so the economic value of the fishery within Algoa Bay was estimated based on the average harvest (2006-2007) and average landed value per kilogram (C.Da Silva *pers. comm.*).

The relative economic importance of commercial sectors in Algoa Bay based on the participation of rights holders locally differed from the national fisheries (Table 6.4). The chokka-squid sector accounted for over half of the estimated commercial fishery economic importance when scaled by the participation of rights holders locally in Algoa Bay (Table 6.4). This was followed by the inshore demersal trawl and small pelagic trawl sectors which accounted for 20.6% and 19.1% of the estimated economic importance locally based on the participation of rights holders, respectively (Table 6.4). The relative economic importance of the commercial linefish and demersal shark longline fisheries locally was far lower accounting for only 6.9% and 0.2% respectively.

Table 6.4. National economic value of commercial fisheries operating in Algoa Bay and estimated local economic value based on participation of rights holders (not harvest) within the study area.

Sector	Landings (tons)	National economic value (ZAR'000)	Approximate number of active vessels	Ave number of vessels reporting catches in Algoa Bay (years considered)	Scaled economic importance of fisheries in Algoa Bay (ZAR'000) [4]	Relative contribution to economic importance
Chokka-squid	6 327	R189 810 [1]	≈138 (DEAT 2007c)	112 (2004-2008)	ZAR154 048	53.2
Demersal inshore trawl	10 492	R94 691 [2]	≈35 (DEAT 2005a)	22 (2000-2008)	ZAR59 520	20.6
SPPSF	454 954	R613 904 [2]	≈100 (DEAT 2005b)	9 (2000-2008)	ZAR55 251	19.1
Commercial linefish	24 103	R346 303 [2]	≈450 (DEAT 2007d)	26 (2002-2008)	ZAR20 008	6.9
Demersal shark longline	28 [3]				ZAR684	0.2
ESTIMATED VALUE OF COMMERCIAL SECTORS IN ALGOA BAY					**ZAR289 512**	

Notes: [1] Landings in 2000 (Anon 2004) multiplied by harbour landing value of ZAR30/kg (J.Tucker *pers. comm.*)
[2] Economic value reported for the sectors in 2000 (Anon 2004)
[3] No national landings or value available, calculated as average landings in Algoa Bay 2006/2007 multiplied by average price per kg (C.Da Silva *pers. comm.*)
[4] Economic importance based on the level of participation of rights holders locally in Algoa Bay relative to rights issues Nationally, and not that of the landed catch arising from Algoa Bay

These results indicate the highly skewed economic importance of fishery activities based on the magnitude of the sector nationally and level of participation locally, with the chokka-squid, inshore demersal trawl and small pelagic purse seine fishery accounting for approximately 93% of the commercial economic importance in Algoa Bay. Fisheries may receive greater consideration in future spatial planning due to their contribution to local or regional economies. As a consequence of the skewed economic importance of commercial sectors, the spatial index should not only take into account the spatial distribution effort for each sector within Algoa Bay, but also the economic importance based on participation of the rights holders in each sector. In order to account for economic importance in the spatial index, the contribution of each sector to the IRCI was scaled by the relative economic importance locally using the proportion of rights holders fishing locally to that of the number of rights issued nationally. This resulted in the production of a spatial index integrating the relative importance of spatial effort and local economic importance of each sectors and can be used to complement, or provide an alternative scenario for use in spatial planning in Algoa Bay, and engagement with fishery stakeholders.

The impact of incorporating economic importance into the spatial index is clearly evident, with the squid fishing grounds off Cape Recife standing out as of major fishery importance (Figure 6.23), followed by the inshore demersal trawl grounds to the east of the study area, as these sectors utilised spatially discrete fishing areas and were of greatest economic importance due to the high proportion of rights holders utilising these fishing grounds. Although the linefish sector also utilised spatially discrete fishing grounds, they do not appear to as important in the economic IRCI when compared to the IRCI due to the lower economic value of the fishery. Effort in the SPPSF and demersal shark sectors was more dispersed throughout the study area and the importance of these sectors in the combined indices is therefore less obvious. Both the IRCI and economic IRCI provide a valuable means for integrating fisheries data into quantitative spatial planning exercises as well as facilitating discussions with fishery stakeholders. These indices, however, represent only two possible methods for weighting and integrating all the commercial sectors into one index. Other indices could also be developed in order to reflect policy priorities for a region, and may include consideration of local of national employment, local harvest or ecological impacts of fisheries activities.

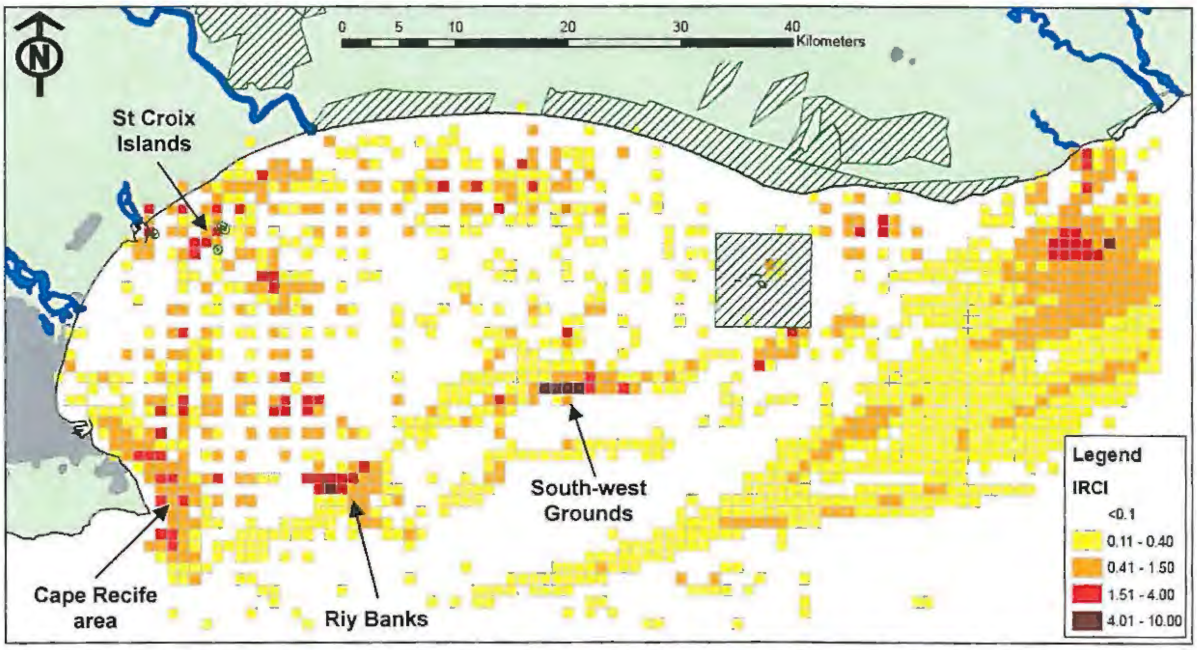


Figure 6.22. Spatial representation of cumulative commercial fishing effort in Algoa Bay based on an Index of Relative Commercial Importance (IRCI). Hashed green areas indicate existing AENP boundaries.

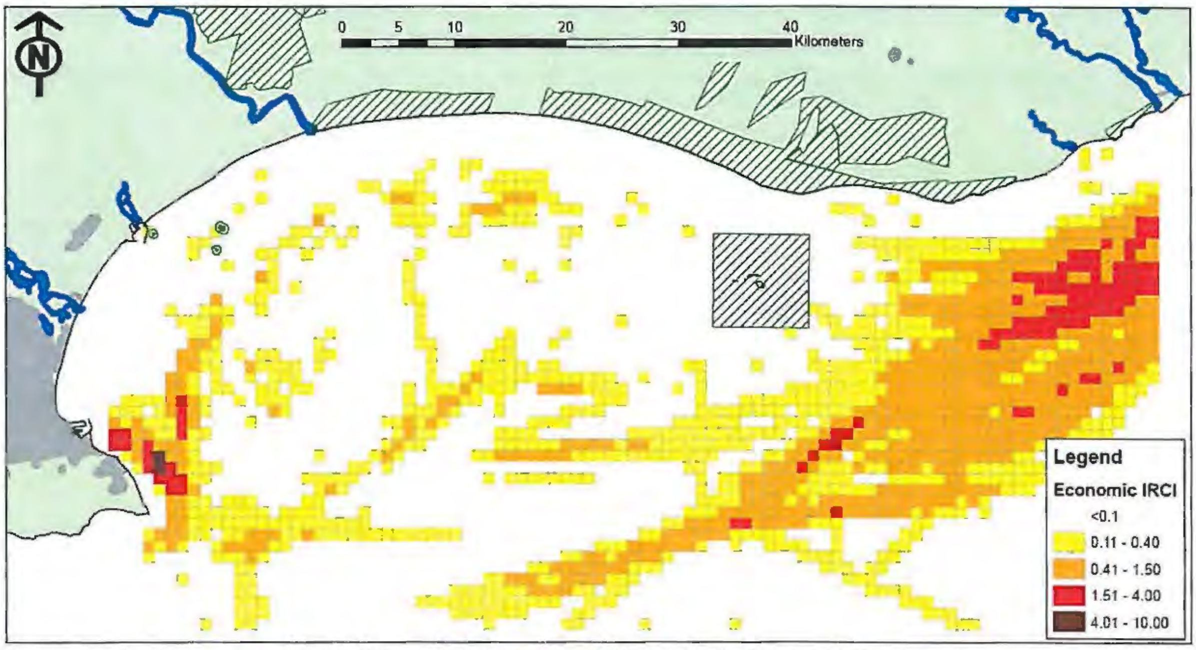


Figure 6.23. Spatial representation of cumulative commercial fishing effort in Algoa Bay based on an Economic Index of Relative Commercial Importance (Economic IRCI). Hashed green areas indicate existing AENP boundaries.

Summary of key findings

- Commercial fishing effort and catch was spatially and temporally variable within Algoa Bay
- Fishing effort and the location of fishing grounds for each sector differed considerably between commercial sectors
- The IRCI and economic IRCI were effective in identifying the important fishing grounds and provide a means to integrate commercial fisheries data into spatial planning

6.4 Discussion

6.4.1 Commercial linefishery

From the onset of the commercial linefishery in Algoa Bay (Period A, when A and B licenses were first allocated) there was an increase in participation (number of active vessels) and fishing effort (boat-days.year^{-1}) from 1985 to 1993/1994 respectively. However, from 1994 to 2003 there was a steady and progressive decline in levels of participation and fishing effort in Algoa Bay. The number of active vessels decreased from Period A to B and from Period B to C, but no change was evident from the beginning of management Period C in 2001. Similarly effort in boat-days.month^{-1} declined from Management Periods A and B to Periods C and D. This suggests that the changes in management of the commercial linefishery through the reduction in the number of rights allocated during the medium and long-term rights allocation process have been effective in reducing participation and effort in the sector locally. This supports the recent findings of Donovan (2010) for the Port Alfred commercial linefishery.

An initial increase in the number of vessels and fishing effort during the establishment of the commercial linefishery during management Period A have been reported in the Port Alfred (Donovan 2010) and KwaZulu-Natal (Penney *et al.* 1999) fisheries, mirroring the findings of this study. Most applicants were successful in obtaining commercial rights when the first management framework was implemented in 1985 (Sauer *et al.* 2003b) and effort was initially capped at this level (Penney *et al.* 1999). The progressive increase in fishing effort is therefore somewhat surprising, but the licensing system in place at the time allowed effort subsidisation. Part-time commercial fishermen (B-licence holders) utilised their commercial access rights to the fishery to supplement their income which was largely derived from other sources. They therefore sold their catch to cover their running costs rather than being dependent on their fishing rights for financial gain (Sauer *et al.* 2003b). B-licence holders were therefore essentially 'recreational' anglers legally permitted to sell their catch. This led to highly variable levels of participation in the fishery by individual rights holders. At this stage commercial rights were freely tradable and transferable (Penney *et al.* 1999) and as effort and entry into the fishery had been capped, commercial rights in the linefishery attained commercial value. Transfer and sale of rights therefore occurred freely, and inactive rights holders relinquished their rights for short-term financial gain. New entrants were typically more interested and therefore active in the fishery. A high turnover of commercial rights therefore maintained a high level of participation in the fishery between 1986 and 1997, during which time up to a third of commercial linefish rights were transferred annually (Griffiths 2000).

The transfer of linefish permits between regions was prohibited in 1994 and the subsequent decrease in effort in Algoa Bay within Management Period A cannot be due to transfer of rights to other regions. Alternatively higher operating costs and lower catch rates may have resulted in reduced profitability and lower incentive to participate in the fishery thereby reducing the participation and effort. The change in regulations in 1999, which abolished B licenses, resulted in a significant decrease in participation and effort in Algoa Bay. Elimination of less active part-time commercials (B licence holders) led to an increase in mean number of days fished per individual vessel over the same period as only more active A licence holders remained in the fishery. Under-reporting is common in the commercial linefishery (Fennessy *et al.* 2003). Catch returns were scrutinised in order to identify "true" commercial participants who were economically dependent on the fishery during the medium and long-term rights allocation process. This formal process may have increased the rights holders' awareness of the importance of submitting catch returns and created additional incentive for completing and submitting accurate returns.

Although both direct and indirect effects of the changes in the management regulations led to a reduction in commercial linefishing effort in Algoa Bay, the catch rate and overall harvest did not follow similar trends. The regulations have therefore not been effective in reducing pressure on the linefish resources locally. Standardised catch rate (for active vessels; five-year history and minimum of 24 reports per year) indicates an initial increase in CPUE from 1987 to 1989 followed by a substantial and continued annual decline from 259kg.boat-day^{-1} in 1989 to 125kg.boat-day^{-1} in 2001. This was an overall reduction of 52% in catch rate suggesting a decline in resource status. However, a subsequent increase in CPUE from 2002 onwards to higher levels than previous years (285kg.boat-day^{-1} in 2007) suggests that the standardised CPUE does not accurately portray the status of linefish resources over this period. The increase in CPUE was closely associated with a reduction in participation and fishing effort (boat-days^{-1}) (Figure 6.4a,b) in Algoa Bay, and an increase in effort per individual vessel (boat-days.vessel^{-1}.month^{-1}) (Figure 6.4b). This suggests that the increase in CPUE can rather be attributed to the elimination of less active and inexperienced skippers from the fishery primarily through the reduction of part-time B-license holders. The remaining rights holders are likely to have been highly experienced and more active in the fishery due to their history of participation in the fishery on which their allocation of long-term rights was based. The change in user profiles may have contributed to the observed increase in catch rate and may therefore discount any claims of improvement in stock status. This theory is supported through findings from earlier studies in South Africa which have indicated significant declines in linefish catch rates (Attwood and Farquhar 1999; Griffiths 2000) and changes in species composition (Donovan 2010).

The difference between the standardised catch rate of active vessels (as defined above) and all available data for the region from 1985 and 2008 supports this theory. Between 1985 and 1996 the standardised CPUE determined from all vessels was consistently lower than that of the active fleet, indicating the influence of less active vessels on lowering the catch rate. From 2002 onwards, however, the standardised CPUE from both datasets is very similar when fewer vessels were active within the fishery and the part-time commercials had been eliminated. Furthermore, due to the increasing operational costs as a result of fuel price increases, skippers are likely to have fished more selectively, expending effort only when target species were abundant in Algoa Bay, thereby avoiding poor catches and low profitability. Spatial changes in the temporal distribution of fishing effort are clearly evident and may have contributed to maintaining high catch rates. In addition technological advances (echo sounders, GPS units, monofilament lines) have increased the ability of anglers to locate and target fish, and return to the same location on subsequent outings thereby improving their fishing efficiency.

What is particularly important is that although there was an initial decrease in the total harvest following the reduction in the number of commercial permit holders, the harvest subsequently increased from 2003 to 2004 to similar levels as prior to the change in management regulations, and although harvest has subsequently fluctuated it has remained relatively high considering the extent of effort reduction. This indicates that although there are fewer active vessels in the commercial linefishery in Algoa Bay, the potential pressure on the resources remains high. This is attributable to a high level of skipper experience locally, and the improved vessel, navigation and sounding equipment available to skippers. This allows improved detection and targeting of fish stocks contributing to higher catchability which is not easily quantified or incorporated into statistical models. The reduction in number of rights holders has therefore not been as effective in reducing the pressure on the resources as was initially anticipated.

A change in the species composition has occurred since the onset of the fishery. In the early 1990s sparids contributed greatly to the overall landed weight, followed by the sciaenids. However, the contribution of sparids declined considerable over the following decade, with increasing contributions of carangids, scombrids and more recently elasmobranches. This suggests a reduction in the relative abundance of sparids reducing availability to the commercial linefishery in Algoa Bay through localised stock depletion. This is confirmed by the declining trends in the proportion and catch rates of four targeted sparids (carpenter, panga, dageraad and roman) since the mid 1990s. There was also a concomitant increase in the proportion of geelbek, giant yellowtail, santer, elf and elasmobranches in the landings suggesting selective targeting of alternative species. This may explain the temporal shift in the spatial distribution of fishing effort as species are selectively targeted in specific areas and during certain seasons. The results suggest that a change in species composition has maintained the high overall CPUE rates in recent times (discussed above); a sign of serial overfishing. This is not surprising as sparids are typically long-lived, slow growing, late to mature, and highly resident with several species undergoing a sex reversal during the life cycle, all of which contribute to their susceptibility to overexploitation. While the sciaenids may not be as susceptible to overfishing, and their large scale migratory patterns contribute to local variations in their availability to the fishery, silver kob are considered to be heavily overexploited and depleted throughout their distributional range (Griffiths 1997b). Although elasmobranches are contributing increasingly to the annual harvest they are particularly susceptible to fishing pressure due to their small brood sizes and late maturation (Stevens *et al.* 2000) and a similar decrease in their CPUE may therefore be anticipated under the current levels of fishing effort. Furthermore sharks are selectively targeted in Algoa Bay by the demersal shark longline fishery (see below) which, in addition to unselective fisheries such as the demersal trawl fishery, place high levels of pressure on this group.

The NMLS provides the most comprehensive means for assessment of commercial linefisheries in South Africa, yet the accuracy of the data is questionable due to the reliance on submission of catch returns from rights holders with little means for independent verification. In this study comparison with observer data indicated that the submitted catch returns compared well with those of independently monitored landings and highlights the importance of such programmes for future monitoring and management of the sector. Improved spatial monitoring of the linefishery is now possible through the implementation of VMS in the sector. This study, however, has revealed that the estimation of fishing effort from VMS data differs greatly from the NMLS, indicating either over-reporting on catch returns or under estimation of effort from VMS. It is unlikely that over-reporting of catch and effort occurs due to the potential implications this may have on the rights holders through further catch and effort restrictions and increased levies. It is therefore most probable that effort is under estimated from VMS data. This can either occur as a result of faulty units or due to willing non-compliance by rights holders through deactivation of units. Nonetheless both data sources confirmed similar spatial trends in fishing effort (based on proportion of total estimated effort) indicating that the VMS data can be used efficiently to obtain highly accurate spatial information on fishing activities and should be used for future monitoring of changes in the spatial distribution of fishing effort. To improve the validity and effectiveness of the use of VMS data an improved system for monitoring compliance with the regulations pertaining to VMS needs to be implemented. This can either be done through frequent spot checks when vessels are observed on the water by compliance vessels or alternatively through regular comparison with accurate launch records and the implementation penalties for non-compliance with VMS permit requirements. In the case of Algoa Bay, where most vessels launch through a national port, accurate harbour entry and exit logs are maintained which can be used for independent verification of the accuracy of VMS recording systems in the future. Problems with VMS unit reliability have been reported by the sector (C.Wilke *pers. comm.*) and will also need to be overcome to improve the potential future use of these systems for monitoring.

6.4.2 Chokka-squid jig fishery

Effort trends in the chokka-squid sector indicate a progressive increase in participation within Algoa Bay from the establishment of the fishery in the mid 1980s to a peak in effort in 2005. Several changes in management have occurred during the history of the fishery which need to be taken into consideration. Prior to the early 1980s the majority of chokka-squid was landed as bycatch in the foreign and domestic trawl sectors (Augustyn *et al.* 1992). However, the development of an international market and the high commercial value of squid led to the rapid development of the chokka-squid jig sector from an early experimental fishery to a full commercial sector, which now accounts for approximately 80% of the annual landed catch (Augustyn *et al.* 1992). Absence of regulations for exploitation of squid led to many linefish vessels targeting this species in the Jeffrey's Bay and St Francis Bay areas. This resulted in an oversupply of poor quality squid to local markets with considerable wastage (Augustyn and Roel 1998). The first management measures for the chokka-squid jig fishery were developed to prevent future oversaturation of the market, which included a daily bag limit for the public fishery, a reduction in effort through elimination of non-active vessels, a three-year moratorium on the transfer of licenses and implementation of a formal closed season over the breeding period (Augustyn *et al.* 1992; Augustyn and Roel 1998).

More recent changes in the chokka-squid fishery have seen the development and use of larger vessels with freezing capabilities to improve the quality of the product, the use of lights to attract squid at night to increase catch rates, as well as the use of drogue anchors which allows parachute fishing in deeper waters at night. The number of vessels active within the fishery has therefore decreased through the amalgamation of the crew complement (TAE) from several smaller vessels into fewer larger vessels capable of spending longer periods at sea and supplying a higher quality product. Effort in the fishery has been regulated from the late 1980s and the observed increase in participation through increased vessel numbers in Algoa Bay up until 2005 is likely due to a spatial shift in fishing effort. Historically most fishing effort occurred on the main fishing grounds situated west of Algoa Bay near Jeffrey's and St Francis bays. As the fishery developed, new spawning sites were identified further east in the vicinity of Port Elizabeth. This resulted in a shift in effort toward the east with increasing numbers of vessels fishing in Algoa Bay. Squid are also highly abundant on the inshore spawning grounds during summer (Sauer *et al.* 1992), particularly in the Algoa Bay region, and as the fishery identified squid spawning aggregations to the eastern side of its distributional range in summer, more vessels targeted squid aggregations in these areas, contributing to the increase in effort.

Although effort within Algoa Bay has increased progressively through the history of the fishery, the CPUE has declined slowly from 1993 to 2008, possibly suggesting local depletion of squid stocks due to the disruption of spawning behaviour and high seasonal pressure on the stocks in these areas. No clear trends were apparent in the total landed catch within Algoa Bay and inter-annual variability was high. Nonetheless the contribution of the landings from Algoa Bay to the national landings has remained relatively consistent from 1991 onwards (15-24%) despite the concomitant increase in fishing effort in Algoa Bay over the same period. This may possibly indicate that the fishery is focusing more effort on the fishing grounds to the east of the distributional range of the species in order to sustain high catch rates in light of declining abundances on the historical fishing grounds to the west of Algoa Bay.

Temporal changes in the spatial distribution of fishing effort within Algoa Bay are also apparent. Effort in the western region of Algoa Bay decreased during the 1990s, while effort in the inshore region increased. This may again be due to a combination of decreasing catch rates on the spawning aggregations in the western region of the fishery and the identification of new productive spawning sites in the central inshore region of Algoa Bay. The progressive shift in effort from the western side of the fishery (Jeffrey's and St Francis bays) and resulting increase in effort in Algoa Bay, primarily in the western region of Algoa Bay, may have placed exceedingly high pressures on the squid stocks on the known spawning aggregations in these areas. This may have contributed to decreasing catch rates in the west of Algoa Bay resulting in the movement of vessels and fishing effort to alternative spawning sites in the inshore region, or alternatively the identification of new sites during the 1990s. The advent of improved sounding and GPS equipment would have facilitated the location of new spawning sites which were previously unknown. Furthermore squid are short-lived and stock status is highly dependent on recruitment into the fishery from the previous year (Rosenberg *et al.* 1990; Pierce and Guerra 1994). Unpredictable movement patterns and poor recruitment can therefore influence availability of the stock to the fishery (Pierce and Guerra 1994). This leads to a highly variable fishery, which may account for spatial changes in fishing effort based on recruitment and movement patterns.

Spatial trends in reported effort for the period 2002-2008 from commercial catch returns indicate that higher levels of effort have recently been occurring in the western region of Algoa Bay. This was supported by the spatial analysis of VMS data for the 2006/7 season, which confirmed the high levels of effort off the Cape Recife point. Surprisingly CPUE was highest in the eastern region, although effort in this area was lowest, while CPUE was lowest in the offshore and western regions of Algoa Bay. Nonetheless due to the high levels of effort in the western region this area accounts for the highest annual harvest within Algoa Bay. This may be due to the spatial and temporal predictability of spawning aggregations in this region compared to other areas further east.

Seasonally effort was higher from June to December than during the first half of the year. A noticeable decrease in effort is evident during November which is due to the closed season (approximately 23 days during November) which is enforced over this period. CPUE during the last week of November was highest which coincides with the peak in activity on the spawning grounds over this period. This indicates that the closed season is effective in reducing fishing pressure when the stocks are densely aggregated on the spawning grounds and therefore particularly sensitive to capture.

Although the chokka-squid fishery is a national fishery and vessels are permitted to move throughout the coastal waters, it is primarily based in the Eastern Cape due to the natural distribution of the squid stocks. The sector contributes significantly to the economy of the Eastern Cape and in particular to the coastal towns and cities where harbour and processing facilities are located (Britz *et al.* 2001). The estimated value of the fishery ranges from ZAR108 million to over ZAR180 million (Augustyn *et al.* 1992; Sauer *et al.* 2003b) and creates employment for approximately 3 500 – 5 000 people locally (Roel 1998; Roel and Butterworth 2000; Dorfler 2006). The sector is of particular importance to the Eastern Cape and Algoa Bay (Sauer *et al.* 2003a) due to both the location of spawning aggregations and the development of local industry, which is dependent on sustainable use of the resource. Although annual landings in the jig fishery are relatively low in comparison to other fisheries it is one of South Africa's most valuable sectors as most landings are exported to European markets (Sauer *et al.* 2003b).

6.4.3 Small pelagic purse seine fishery

There has been a notable increase in participation of SPPSF vessels in the Algoa Bay region since the early 2000s, as well as an increase in fishing effort over the same period. These increases are a likely result of changes in the spatial dynamics of the distribution of the target stocks. The bulk of sardine biomass was traditionally distributed on the south-west coast of South Africa; however, a major shift in distribution of stocks occurred during the 1990s with an increasingly larger proportion of stock biomass located on the south-east coast (van der Lingen *et al.* 2005; Coetzee *et al.* 2008). This change in distribution has seen an increase in fishing effort along the south-east coast with movement of fishing effort from the traditional fishing grounds to the west of Cape Agulhas (van der Lingen *et al.* 2005; Fairweather *et al.* 2006b) as was observed in the current study. Although daily catch rates in Algoa Bay have remained relatively stable from 2003 onwards, the number of gear sets per day have increased over this period. This has accounted for increasing annual landings in Algoa Bay and the increasing contribution and importance of the Algoa Bay landings to the national fishery during the early and mid 2000s. Sardine dominated the landings in Algoa Bay with mackerel being the only other species which contributed significantly to the total landed catch.

A decline in effort in 2007 and 2008 occurred with Algoa Bay landings decreasing and accounting for a lower proportion of the national landings. The spatial shift in fishing effort is likely to be due to the inability of the fishery to land the annual TAC during the mid to late 2000s (Coetzee *et al.* 2008) and greater emphasis placed on fishing the less traditional grounds. Seasonal trends indicate lower catch rates in Algoa Bay during spring possibly accounting for lower effort during this time. Stock size and recruitment of small short-lived pelagic fish is highly variable (Schwartzlose *et al.* 1999) with environmental conditions playing a major role in determining the large-scale distribution of stocks (Armstrong *et al.* 1987) and recruitment success (Cole 1999; Daskalov *et al.* 2003; Yatsu *et al.* 2005). These factors contribute to the dynamic nature of the fishery.

Observer data indicated that catch locations were highly dispersed, occurring throughout Algoa Bay, although a concentration of effort close to Cape Recife and Port Elizabeth was evident. This is likely due to the frequency with which vessels traverse these areas when entering and exiting the harbour and moving to fishing grounds or harbour facilities to the south-west increasing their likelihood of intercepting pelagic shoals in the process. Verification between catch return and observer data indicated that reported landings were accurate. However, the accuracy of the spatial information reported by skippers was poor, and should therefore not be relied upon for future spatial planning. Although observer data provides a more accurate means for spatial assessment of fishery activities, there was a low frequency of monitoring by onboard fisheries observers of vessel activity within Algoa Bay (0-3% of fishing days per year). Using VMS data to supplement the observer data is highly beneficial as it provides accurate spatial information from which fishing activity can be inferred. Verification of catch locations indentified through analysis of VMS data with locations provided by onboard observer records indicated that the decision rules used in the analysis of VMS data were relatively effective in identifying fishing positions. Although additional catch sites were identified through the VMS data, these sites still represent areas in which the SPPSF were actively engaged in searching for pelagic shoals and contribute to the overall effort in the fishery. The spatial distribution of small pelagic species is highly influenced by environmental conditions (Armstrong *et al.* 1987) and is therefore highly variable spatially and temporally. Taking searching effort into account provides a conservative assessment of the fishery activities and will not negatively influence further analysis.

6.4.4 Inshore demersal trawl fishery

Temporal trends in the demersal trawl fishery indicate an increase in participation and trawl effort in Algoa Bay from 2000 to 2003 but have subsequently remained relatively stable. From 2003 onwards up to 77% of the inshore fleet fished within Algoa Bay annually indicating the importance of this area to the national fishery. Landings from Algoa Bay, however, did not contribute significantly to the national landed catch, ranging from 6 to 11% between 2002 and 2006 (Anon 2004; Anon 2006; Anon 2007).

Both nominal and standardised CPUE indicate a progressive decline in catch rate from 2003 onwards suggesting decreasing stocks. Due to the declining catch rates total annual harvest also showed a general decline from a peak in 2003 to landed catches in 2008 which were approximately 36% lower despite a constant level of fishing effort during this period. These trends in effort and catch rate suggest that the historic and current levels of fishing pressure are exceeding the sustainable harvest levels of the demersal resources within Algoa Bay. Despite declining CPUE, the reported landed catch composition indicates relatively stable contribution of shallow-water hake, the main target species, to the annual landings from 2003 onwards. The proportion of horse mackerel decreased considerably over the assessment period; however, these contributions are higher than the contribution of horse mackerel to the national landings which ranged from 6-8% (2005-2006) (Anon 2007) compared to 16-25% in Algoa Bay over the same period. Algoa Bay is therefore an important area for horse mackerel landings which is used by the inshore fleet. Changes in the proportion of catches are most likely due to the pelagic and short-lived life history characteristics leading to high spatial and temporal variability in the distribution of horse mackerel stocks. Due to these characteristics specific targeting of horse mackerel is limited to periods when they are locally abundant (Sauer *et al.* 2003b) and the majority of inshore trawl effort is concentrated on demersal species which are more predictable in their spatial and temporal patterns.

Although the shallow-water hake is the main target species of the inshore demersal fleet, the landed catch composition has been reported to be influenced by the quota allocations of rights holders, with the proportion of bycatch increasing with decreasing hake quota (Sauer *et al.* 2003b). Rights holders with smaller hake quotas tend to be less 'selective' in their fishing in order to increase their overall harvest and economic return prior to fulfilling their hake quota. The proportion of bycatch for the three largest hake quota holders in the inshore fleet is reported to be less than 10% but as high as 38% for the remaining smaller quota holders (Sauer *et al.* 2003b). The declining catch rates evident in Algoa Bay may be due to selective targeting and high fishing pressure on bycatch species which are more sensitive to fishing pressure due to the life history characteristics (slow growth, late maturity etc.). Reported catch composition of landings indicated that with the exception of years 2002 and 2003 (14% and 11%) sparids contributed less than 1% to the annual harvest in Algoa Bay. Contrarily the observer data from Algoa Bay indicated that sparids may account for as much as 27% of the catch weight, suggesting significant inaccuracies in the reported landings by skippers. Furthermore, observer data showed that up to 25% of the catch weight consisted of elasmobranches in 2003 while reported landings indicated that elasmobranches accounted for only 1%. Although the hake stocks are typically well managed in South Africa, the linefish resources in contrast, particularly the sparids and sciaenids, are overexploited and several stocks are considered collapsed. High levels of trawling effort may therefore contribute to local depletion of these species as they are unable to withstand intensive fishing pressure. Comparison of the observer catch composition and the reported data confirmed differences in the proportion for certain taxa, suggesting inaccurate or misreporting by vessel skippers with only small landings of non-target species reported, if any at all. Although it is possible that the low frequency of observer trips of the demersal inshore fleet (3-12% of boat-days in Algoa Bay annually between 2003 and 2008) may account for these differences, it highlights the need for increased monitoring of catches in Algoa Bay and the potential impact inshore demersal trawling may be having on non-target species.

Reported effort was concentrated in the eastern region of Algoa Bay. The observer trawl path data confirms the accuracy of the reported data, and provides further high resolution spatial data indicating spatial heterogeneity in fishing effort and the high frequency at which some areas are repeatedly trawled within Algoa Bay (Figure 6.22). The spatial distribution of substrates suitable for bottom trawling is likely to be the most important factor influencing the location of trawling effort. Repeated trawling of an area is not only likely to influence and reduce the local abundance of target species, leading to localised depletion, but is also likely to impact on the benthic communities, reducing their ability to support demersal fish communities. Although effort was highest in the eastern region of Algoa Bay, catch rate was lowest, suggesting that this may indeed be the case. Research trawls in this region confirm the dominance of horse mackerel and shallow-water hake in the demersal communities in this area (Chapter 4) but indicate lower proportions than reported by the fishery. Lower proportions of elasmobranches and sparids were reported by the fishery suggesting effective targeting of shallow-water hake and horse mackerel, or alternatively that under-reporting of bycatch is occurring.

Demersal trawling is highly destructive, altering benthic community structure, species richness, biomass and productivity (Thrush *et al.* 1998; Kaiser *et al.* 1998; Thrush and Dayton 2002; Blyth *et al.*

2004; Hiddink *et al.* 2006; Hinz *et al.* 2009). These organisms form the base of the food web for many higher trophic level demersal species and may therefore influence the distribution of demersal ichthyofaunal assemblages. A strong positive correlation has been shown to exist between benthic macrofaunal abundance and commercial demersal fish abundance (Moran and Stephenson 2000). Little benthic macrofauna is typically retained in the nets of commercial vessels leading to the belief that the trawl nets have little impact on the communities and that the soft benthic substrates support sparse communities. However, research has shown macrofauna are abundant on trawl grounds and that only approximately 4% of the larger benthic organisms detached from the substrate are retained in the net and landed on deck (Moran and Stephenson 2000). Each trawl pass over the substrate has also been estimated to reduce benthic macrofaunal density by up to 20% (Moran and Stephenson 2000; Pitcher *et al.* 2000). This highlights the potential impact of repeated high density trawling effort on the macrobenthic assemblages. Little information pertaining to benthic macro invertebrate communities and the interdependencies of demersal ichthyofauna on these communities on the inshore trawl grounds on the east coast of South Africa is currently available.

Although demersal trawling can be highly destructive, it is the most important commercial fishery sector in South Africa, accounting for up to 50% of the income generated from the living marine resources (Sauer *et al.* 2003b). The sector also provides the highest levels of permanent employment in fisheries. However, due to its long-term impact on demersal habitats through homogenisation and modification of communities and substrates, ecologically sensitive areas need to be identified and protected. Due to the economic importance of the fishery, considerable research has been conducted on the hakes and the hake fisheries (Payne *et al.* 1985; Payne *et al.* 1987; Badenhorst 1988; Punt 1994; Pillar and Barange 1997; Wilkinson and Japp 2005; Fairweather *et al.* 2006a) as well as the east coast sole (Le Clus *et al.* 1994; Le Clus *et al.* 1996). However, the quantification and management of bycatch remains a problem. Spatial management of the inshore trawl fleet through exclusion from areas which support diverse assemblages of non-target species is likely to be the most effective means for managing unselective targeting, bycatch problems and habitat degradation in the future. Due to the inaccuracies indentified in the reported landings data this requires dedicated fisheries independent assessments and the use of high resolution observer monitoring data. Increased effort should be placed on monitoring trawling activities and catch composition in sensitive inshore areas and independent non-destructive techniques for evaluating the demersal ichthyofauna and macrobenthic communities should be investigated.

6.4.5 Demersal shark longline fishery

Due to the poor performance in the fishery effort was reduced from 30 shark longline permits in the past to 11 permits in 2004. Further changes in the fishery occurred in 2006 following the long-term rights allocation process with the shark longline fishery split into pelagic and demersal sectors, and six permits allocated to the demersal fishery (Da Silva and Bürgener 2007; DEAT 2007b). Only one of these vessels was active within Algoa Bay during 2006 and 2007 and the fishing effort was low (38-50 boat-days.year^{-1}) in comparison to other commercial fisheries assessed in this study.

Strong seasonal trends in effort were apparent with fishing occurring over the autumn and winter months and no fishing during late spring and summer. No seasonal restrictions apply to the fishery and the limited effort during spring and summer has been attributed to participation in multiple sectors by the rights holder (DEAT 2007b). Shark fishing is therefore only conducted when activity in other sectors is low and sharks are abundant locally during winter (DEAT 2007b). Almost all fishing effort was in depths less than 50m with effort concentrated in the western region of Algoa Bay. This indicates selective targeting of inshore species in the sheltered waters. Catch rate did not vary spatially; however, the variability around the mean was high as is typical for shark catches in longline fisheries (Beerkircher *et al.* 2002).

Demersal shark longline landings in South Africa declined from 24.4 in 2001 to 5.4 tons (gutted and headed) in 2005, far lower than the landings in Algoa Bay in 2006 and 2007. This drop in landings was attributed to a decrease in effort over this period rather than declining catch rates (Da Silva and Bürgener 2007). In comparison the reported shark landings from the traditional linefish sector ranged from 328.8 to 174.3 tons between 2000 and 2005 (Da Silva and Bürgener 2007) indicating a significantly larger contribution to the annual demersal shark landings than the longline fishery. Elasmobranchs have low fecundity, are slow growing, become sexually mature at a late age and undertake complex migrations (Stevens *et al.* 2000). There is also limited biological data for many species and the current status of stocks in South Africa is uncertain (DEAT 2007b). These factors are further confounded by poor and unreliable fisheries data, which contributes to a poor understanding of the dynamics of the stocks, increasing their vulnerability and susceptibility to overexploitation. Furthermore the demersal shark longline fishery is only regulated through TAE, with no limitation on catch landed, potentially allowing for overexploitation.

This analysis was based on catch return data submitted by the rights holder as no data were available from onboard observers or access point monitoring. Furthermore the VMS polling interval on the tracking unit was set to six hours, which proved insufficient for spatial analysis. Both catch and spatial data for this sector is therefore questionable as there was no independent means of validation, and poor data veracity and monitoring has been highlighted as problems in the demersal shark fisheries previously (Da Silva and Bürgener 2007). Recent stock assessments conducted on the soupfin shark (McCord 2005) and the smooth-hound shark (Da Silva 2007) have indicated that these species are overexploited suggesting that an improved system for monitoring catch and effort in the fishery is required in order to evaluate its long-term sustainability. Improved spatial management on a local scale within Algoa Bay will contribute to the protection of several demersal shark species targeted locally through a spatial reduction in fishing effort.

6.4.6 Conclusions

Five commercial sectors which actively fish within Algoa Bay were identified and assessed in this study. Long-term temporal trends evident in some sectors were influenced by changes in management regulations. All sectors displayed strong spatial and seasonal trends in fishing effort. Spatial indices of relative effort across Algoa Bay were developed for each sector using VMS and onboard observed data where possible. The sector indices were weighted based on the relative importance of fishing grounds in Algoa Bay to each sector prior to being integrated into a spatial index of commercial fisheries to depict areas of greatest importance for commercial fishing effort in Algoa Bay. This was undertaken to account for the lower importance of fishing grounds in Algoa Bay to sectors which fish over larger geographic areas and have access to alternative fishing sites, while increasing the importance for those sectors which are more reliant on the fishery resources locally within Algoa Bay. This index confirmed the wide scale and heterogeneous distribution of commercial fishing effort in Algoa Bay and contributed to the identification of key areas of importance. The estimated economic value of each sector in Algoa Bay was used to produce a spatial index of economic importance for commercial fisheries. This highlighted the economic importance of the chokka-squid and demersal trawl fishing grounds locally. These indices form the basis for integrating commercial fisheries activities into spatial planning in Algoa Bay, and can be used to initiate engagement with fishery stakeholders (Table 6.5).

This study has highlighted the data sources available for monitoring future trends in commercial fishery activities in Algoa Bay and the limitations of each (Table 6.5). This will contribute to improved data quality and evaluation in the future through the establishment of systems for validating reported logbook data using fishery independent sources, and through engagement with fishery stakeholders on a local level. Ongoing evaluation will allow the quantification of effort displacement as a result of spatial closures in Algoa Bay and evaluation of changes in harvest and catch composition.

Table 6.5. Contribution of chapter results to spatial planning and monitoring in Algoa Bay.

Chapter 7: Systematic conservation planning	Chapter 8: Monitoring and evaluation
1. Spatial indices of relative effort for each commercial sectors 2. Integrated spatial indices of relative commercial effort and relative economic importance	1. Long-term fishing effort and catch composition baseline data for future comparative assessments 2. Identification of key factors influencing spatial and temporal dynamics in each fishery sector 3. Identification of data sources for future monitoring, and limitations thereof

CHAPTER 7
BALANCING ECOLOGICAL AND SOCIO-ECONOMIC OBJECTIVES IN ALGOA BAY THROUGH SYSTEMATIC CONSERVATION PLANNING

7.1 Introduction

Marine ecosystems are increasingly threatened through human activities, including exploitation, pollution, coastal development, mining and shipping amongst others. The escalation of industrial fisheries has led to the wide scale depletion of target populations (Roberts 2007) and contributed to the degradation of marine communities and habitats through the use of destructive and unselective fishing techniques (Moran and Stephenson 2000; Pitcher *et al.* 2000). Although sector and species specific fisheries regulations have been in place in many regions for some time, the cumulative effects and growth of multi-sectoral fisheries has often been overlooked leading to unsustainable levels of utilisation globally (Pauly *et al.* 2002). Unregulated and illegal fisheries activities remain problematic (le Gallic and Cox 2006; Moolla 2008) and contribute to the current poor state of marine resources, which is evident through declining annual global harvests of wild capture marine fisheries (FAO 2010). Historically fisheries management has focused on determining optimal harvest levels of target species through single-species management approaches with little regard for non-target bycatch organisms, or the biological significance of extractive use on ecosystem functionality. Increasing recognition of the collective effects of all fisheries activities on community structure and ecosystem health prompted the development of ecosystem based management principles for marine fisheries, which aim to balance the socio-economic requirements of society with the protection of the natural environment through an integrated approach (Garcia *et al.* 2003). Although the principles of EAF are not new, practical implementation has been poor in the absence of clear guidelines for execution (Paterson and Petersen 2010).

Spatial planning is one of the key tools available which can be used to facilitate the implementation of EBM in marine ecosystems (Douvere 2008) through the protection of natural habitats in MPAs in which human activities are strictly regulated or excluded. Marine ecosystems are open-access commons with management restrictions often limited to territorial waters and specific activities which are poorly regulated due to limited capacity. Formal management and zoning strategies are therefore required if sustainable use is to be achieved in the long-term (Stewart *et al.* 2003). MPAs are a central component for the conservation and management of marine ecosystems, playing a role in both the protection of biodiversity and ecosystem processes (Agardy 1994; Worm *et al.* 2006; Tognelli *et al.* 2009; Koldewey *et al.* 2010) as well as supporting traditional fisheries management through the protection of target stocks (Mosquera *et al.* 2000; Murawski *et al.* 2000; Roberts *et al.* 2001; Gell and Roberts 2003a; Gell and Roberts 2003b). No-take MPAs have the added advantage that they are spatially explicit allowing for strict regulation of all activities within a defined area. In some instances this may provide a more effective means for fisheries management in regions where multi-species and sectoral fisheries are highly dispersed and the capacity for enforcement and monitoring of traditional regulations is poor.

Although the benefits of MPAs are acknowledged, the effectiveness of individual, and networks of MPAs in achieving their objectives is dependent on their location, size and design (Gaines *et al.* 2010). Historically MPAs have often been established on an *ad hoc* basis and for a variety of purposes (Roberts 2000). Large reserves are more beneficial for the conservation of biodiversity, while several smaller reserves which have a large boundary length are better suited to fisheries management as they allow greater export to adjacent fisheries across the MPA borders (Hastings and Botsford 2003). MPAs have also often been located in areas which present opportunities for protection rather than in sites which optimise the benefits for the conservation of marine biodiversity and contribute to regional conservation objectives (Hockey and Branch 1997). This has frequently resulted in poor and inadequate representation of many of the biophysical features and species which they aim to protect, leading to a false sense of security in conservation and management efforts.

Systematic conservation planning (SCP) has been developed to overcome these problems and identify priority areas which best represent the biodiversity and ecological processes within the planning area (Margules and Pressey 2000). Protection of priority areas identified through SCP promotes the long-term persistence of species and ecosystem services. In order to effectively identify these areas comprehensive spatial information on habitat and species distributions, diversity patterns and biophysical interactions is required. Surrogate biophysical features are often selected to represent the overall biodiversity of a region in areas where limited spatial data are available and are thereby used for the identification of priority areas for conservation investment (Margules and Pressey 2000). Broad societal or political goals form the basis for spatial planning initiatives which are interpreted into quantitative targets which define the level of protection required for each biophysical feature (Rondinini and Chiozza 2010). Conservation targets not only provide a quantifiable means to monitor and evaluate progress towards management objectives, but also provide a transparent and defensible decision support system with which to guide the design process for the selection of reserve networks (Margules and Pressey 2000). In order to improve regional representivity of biophysical features within MPA networks, the contribution of existing MPAs to the conservation objectives need to be taken into consideration and used as the basis for future expansion (Pressey *et al.* 1993). This will ensure that newly proclaimed MPAs complement existing reserves to create comprehensive networks which are fully representative of regional biodiversity (Stewart *et al.* 2003).

SCP has been widely and successfully used in numerous terrestrial programmes for the identification of priority areas for protection and the development of conservation strategies and reserve systems. However, they have been less widely, and only relatively recently applied to marine ecosystems (e.g. Beck and Odaya 2001; Airamé *et al.* 2003; Stewart *et al.* 2003; Stewart and Possingham 2005; Lombard *et al.* 2007; Klein *et al.* 2008a; Klein *et al.* 2008b; Ban and Klein 2009; Klein *et al.* 2010; Giakoumi *et al.* 2011; Grantham *et al.* 2011). The absence of spatially explicit biophysical data have been a major constraint to the application of SCP in marine ecosystems (Carr *et al.* 2003). Logistical and technical difficulties of working in marine subtidal environments as well as financial constraints have often limited the acquisition of the required data to conduct effective SCP. In addition, past research emphasis has focused on charismatic species, or specific areas of interest such as existing MPAs (Stewart *et al.* 2003) with little effort on broader spatial patterns.

Although the overall goal of SCP is the conservation of biodiversity through the protection of representative areas, it can also take socio-economic considerations into account thereby contributing to the goals of ecosystem based approaches. This is achieved through solving the "minimum set coverage problem" where the objective is to determine the best spatial solution for achieving the conservation targets at the least cost (Moilanen *et al.* 2009). Cost can simply be incorporated into the analysis as the amount of area so that the smallest possible reserve system is identified which meets all conservation targets. Alternatively cost minimisation can be based on spatially explicit economic or opportunity costs using the economic value of the land required, or potential losses of revenue or access in the minimisation process. As with baseline biophysical data, a paucity of spatially explicit socio-economic data is typical for most marine ecosystems. Incorporating spatially explicit cost information into conservation planning not only reduces the socio-economic impacts of establishing new protected areas (Klein *et al.* 2008b; Ban and Klein 2009; Klein *et al.* 2010) but also increases public support and contributes to improved compliance. The dearth of spatially explicit biophysical and socio-economic data are a major challenge to marine conservation planning. Planning initiatives need to overcome these problems if meaningful spatial conservation actions are to be identified and implemented.

A national assessment of biodiversity within the existing South African MPA network indicated poor representation and highlighted the need for future expansion in order to attain the desired targets for protection (Lombard *et al.* 2004). A more detailed study was conducted within the Agulhas Bioregion along the south-east coast of South Africa which confirmed the inadequacy of the current MPA network in achieving conservation targets for selected biophysical features (Clark and Lombard 2007). Furthermore bay environments along the South African coastline were identified during stakeholder workshops as being poorly represented, and of ecological importance due the nursery function they serve for many marine species (Clark and Lombard 2007). The initial proposals for the development of the AENP MPA originated in the mid 1990s with preliminary footprint designs based on expert opinion (Kerley and Boshoff 1997). Although several refinements and alternative footprint designs were considered during the strategic environmental assessment for the AENP (Newman and Klages 2001), the overall footprint has remained largely unchanged. Recently no-take areas (See Appendix 1) within the AENP footprint have been defined in order to design a multi-use MPA within Algoa Bay. Both the footprint and no-take areas have largely been defined based on expert opinion and there has been no quantitative appraisal of the contribution that the AENP footprint would make to local and regional conservation objectives, nor an evaluation of the conservation benefits of the proposed no-take areas within the proposed AENP footprint. Furthermore there has been no attempt to quantify the level of impact to fishery activities in the area.

The overall aim of this chapter was to evaluate the MPA design currently proposed and develop alternative no-take reserve design options. This was achieved by quantifying the ecological benefits and impacts to fisheries using biophysical and socio-economic baseline data presented in previous chapters using a quantitative systematic conservation planning framework (Figure 7.1).

Systematic conservation planning analyses were conducted to meet the following objectives:

1. to identify priority areas for conservation investment;
2. to determine the optimal strategy for expansion of existing no-take MPAs in Algoa Bay;
3. to investigate alternative reserve designs taking the spatial distribution of fisheries activities into account in order to minimise impacts; and
4. to evaluate the conservation potential of the proposed AENP MPA footprint and no-take areas and provide recommendations for additional expansion.

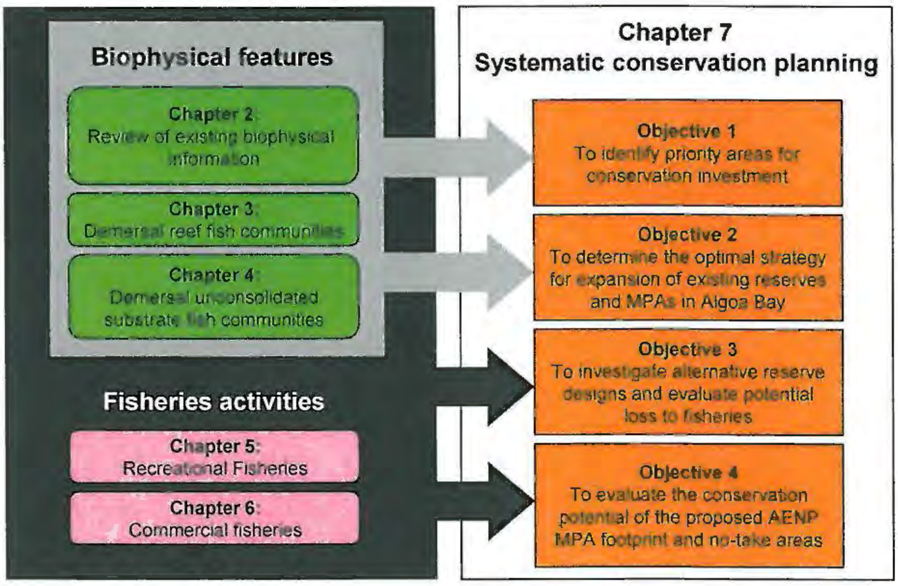


Figure 7.1. Contribution and integration of different thesis chapters to achieve conservation planning objectives.

7.2 Methods

7.2.1 Planning domain and planning units for SCP

The planning domain for Algoa Bay was defined as the area from the approximate location of the 100m isobath offshore of Cape Recife point eastwards to the Bushmans River Mouth. The study area was divided into 1km^2 (100ha) planning units (PUs) using a square grid which resulted in a total of 4 124 PUs. Conservation feature (Table 7.1) abundance data were populated for each planning unit and different cost layers were used based on the objectives of each scenario (Table 7.2).

7.2.2 Conservation features and targets

Biophysical features deemed important for conservation in Algoa Bay were identified through a review of existing baseline information (Chapter 2) and the assessment of demersal ichthyofaunal communities (Chapter 3 and 4). Three categories of features were distinguished for planning purposes, namely habitats, process areas and surrogate species (Table 7.1). Habitat features included intertidal (Clark and Lombard 2007) and subtidal (SANHO 1939; SANHO 1950; SANHO 1952; Bremner 1978; Buxton 1987; Koornhof 2000; Sampson 2002; Rubidge 2006; Boshoff and Boshoff 2008) substrate types as well as depth categories (SANHO 1975) obtained from available information sources (Table 7.1).

Process areas are areas which serve important ecological functions. These include aggregation, nursery and feeding areas, and migration routes. Process areas identified in Algoa Bay included a juvenile silver kob nursery area around the St Croix islands, suggested by Smale (1984), which was confirmed through the controlled angling survey in this study (Chapter 3). Additionally, a nursery area was identified in the shallow sheltered western region of Algoa Bay for other demersal ichthyofaunal species (Chapter 4). Several marine teleosts are estuarine dependent, utilising estuarine habitats during certain phases of their life-cycle (Harrison 2003). Estuary mouths and the surrounding nearshore regions are therefore important areas for these species as they form a link between the estuarine and near and offshore marine environments, creating a corridor through which these species move on a regular basis (Turpie *et al.* 2002). In order to accommodate the importance of this feature into the reserve design, a 2km estuarine linkage buffer around estuarine mouths was developed. The islands of Algoa Bay serve an important ecological function supporting important bird colonies and nesting sites (Crawford *et al.* 1983; Shelton *et al.* 1984; Randall and Randall 1986; Klages *et al.* 1992; Crawford *et al.* 1995; Pichegru and Ryan 2008; Pichegru *et al.* 2010). These colonies in turn export nutrients to the surrounding marine environment and shallow subtidal and island surrounds creating a highly productive environment which supports a rich biodiversity of marine organisms and high abundances of species (Anderson and Stegenga 1989; Anderson 2003). All these features were included into the analysis as process areas (Table 7.1).

Several species were selected as surrogates to represent important nearshore and offshore areas for marine communities (Table 7.1). Three nearshore and estuarine associated species, dusky kob (*Argyrosomus japonicus*), white steenbras (*Lithognathus lithognathus*) and leervis (*Lichia amia*) were selected as surrogate species as they are important target species in the recreational shore fishery, and are considered to be overexploited (Bennett 1993; Griffiths 1997a; Mann 2000; Smith 2008). Spatial distribution maps were developed during a specialist workshop with research scientists from the South African Institute for Aquatic Biodiversity (SAIAB) who are currently engaged in passive tagging and acoustic telemetry studies to investigate nearshore fish movement and behaviour of these species in Algoa Bay. As these three species were not representative of nearshore resident reef dependent communities (Mann 2000; Cowley *et al.* 2001; Watt-Pringle 2009), the intertidal rocky shoreline was buffered by 500m on the seaward boundary to simulate the distribution on inshore reef associated species.

The distribution of two offshore migrant species, geelbek (*Atractoscion aequidens*) and silver kob (*Argyrosomus inodorus*), which are commonly encountered within Algoa Bay and are important target species for the recreational and commercial skiboat fisheries (Smale and Buxton 1985; Hecht and Tilney 1989; Brouwer and Buxton 2002) was determined through the analysis of research trawl data which were converted to a spatial layer for incorporation into the analysis (Chapter 4). Two subtidal reef fish community types were identified in Algoa Bay using multivariate analysis (Chapter 3) and all reef localities identified during this project were designated as either community group 1 or 2 based on their location within Algoa Bay (Table 7.1).

Table 7.1. Biophysical conservation features identified for inclusion in the systematic conservation planning process, their data source, extent within the study area and the target for representation within no-take marine-protected areas.

Feature category	Conservation feature	ID	Source	Unit	Amount within study area	Target (%)
Habitats	Intertidal rock	1	Clark and Lombard 2007	km	12	25
	Intertidal rock above sand	2			23	25
	Intertidal rock and sand	3			4	25
	Intertidal sand	4			95	25
	Intertidal sand above rock	5			11	25
	Subtidal gravel	6	Bremner 1978; Sampson 2002	km^2	317	25
	Subtidal mud	7			30	25
	Subtidal muddy-sand	8			723	25
	Subtidal sand	9			2629	25
	Subtidal sandy-mud	10			228	25
	Subtidal reef – confirmed	11	Mapping conducted in Chapter 3; Buxton 1987; SANHO[12] 1939, 1950 and 1952; Koornhof 2000; Rubidge 2006; Boshoff and Boshoff 2008; Waypoints from stakeholders[13]	Presence per $1km^2$	174	50
	Subtidal reef – potential	12			398	30
	Shallow	13	South African Nautical Charts, SANHO 1975	km^2	572	15
	Medium	14			722	15
	Deep	15			1325	15
	Very deep	16			1305	15
Process areas	Kob nursery	17	Smale 1984; Chapter 3	km^2	30	20
	Estuary link	18	Estuary mouths buffered 2km either side	km^2	23	70
	Demersal nursery	19	Chapter 4	km^2	760	10
	Island – terrestrial	20	Shoreline boundaries of islands mapped in Chapter 3; Intertidal area buffered 1km; Island surrounds buffered 3km	km^2	1	100
	Island – shallow subtidal	21			20	100
	Island – surrounds	22			80	30
Species distributions	Dusky kob	23	Specialist input[14]	km^2	389	15
	White steenbras	24			165	15
	Leervis	25			123	15
	Intertidal reef fish	26	Clark and Lombard 2007; shoreline reef buffered 500m	km^2	27	20
	Geelbek	27	Chapter 4	km^2	1863	10
	Silver kob	28			1076	10
	Subtidal reef fish Group 1	29	Chapter 3	km^2	31	25
	Subtidal reef fish Group 2	30			118	25
	Abalone distribution	31	SFRI 1986; Tarr 1987; Godfrey 2000; Chalmers, unpublished data	km^2	28	75
	A. australis distribution	32	Talbot 1988a; Talbot 1988b	km^2	44	15
	Penguin foraging BI (a)	33	Pichegru *et al.* 2010, foraging ranges prior to 2008 (a) and after 2009 (b) for Bird Island (BI) and St Croix (StC) colonies	km^2	218	15
	Penguin foraging BI (b)	34			240	15
	Penguin foraging StC (a)	35			819	15
	Penguin foraging StC (b)	36			360	15

[12] South African Navy Hydrographic Office
[13] Interviews were conducted with tourism operators and recreational and commercial anglers who provided waypoints for reef locations but wished to remain anonymous
[14] Dr Paul Cowley – South African Institute for Aquatic Biodiversity (SAIAB); Ms Amber Childs – SAIAB; Mr Rhett Bennett – SAIAB

Shallow subtidal reef sites where abalone are known to occur were identified through past research projects (SFRI 1986; Tarr and Anderson 1987; Wood 1993; Britz *et al.* 2002; Godfrey 2003; Raemaekers and Britz 2009) as well as through numerous exploratory dives during the establishment of abalone monitoring sites as a component of the AENP MPA project (Chalmers, unpublished data). The spatial extent of abalone distribution was defined as extending from the shoreline to the 15m isobath in all shallow subtidal reef areas where they have been recorded.

The Sundays surf zone is known to support dense aggregations of the surf zone diatom, *Anaulus australis*, due to the seepage of nutrient rich water from aquifers along this region of coastline (Talbot and Bate 1987b; Talbot and Bate 1988a; Campbell and Bate 1998). These diatom aggregations are an important feature in Algoa Bay having a major contribution to the primary production in the area. The spatial distribution of *A. australis* blooms was delimited using a 1km buffer from the shoreline for the area between the Sundays Estuary Mouth and the Woody Cape Headland. The foraging ranges of African penguins was obtained from recent research publications based on GPS tracking of individuals from the St Croix and Bird Island colonies in Algoa Bay (Pichegru and Ryan 2008; Pichegru *et al.* 2010).

Conservation targets provide the quantitative means with which to select representative areas for protection by specifying how much of each feature must be protected, and provide accountable and defensible reasons for establishing protected areas. They are selected based on the interpretation of broad conservation goals determined through policy directives, expert opinion, stakeholder interaction or combinations of these (Pressey *et al.* 2003; Rondinini and Chiozza 2010). As a result, target values have varied considerably between areas and projects with much ongoing debate over how much of a feature is required in order to ensure representivity and long-term persistence (Svancara *et al.* 2005). Often targets are set as an equal percentage of the amount of all conservation features which are present within the planning domain based on recommendations from international conventions or guidelines (e.g. Convention of Biological Diversity; World Parks Congress IUCN). Such targets are often used on a political level to specify how much of a country or region is required to be conserved in order to meet international agreements; however, they have been widely criticised as they often lack biological justification (Svancara *et al.* 2005). Nonetheless, such conservation targets are often used as the basis for guiding conservation assessments in the absence of sufficient scientific knowledge or expertise.

A widely used target is that recommended at the World Parks Congress (2003) which requires 20-30% of marine conservation features to be protected within protected area networks by 2012 (Anon 2003). In South Africa the National Spatial Biodiversity Assessment (NSBA) provides a broad appraisal of the country's conservation efforts, providing a basis for identifying where further local level investigation is required. The NSBA, completed in 2004, is currently under revision and the marine assessment is utilising conservation targets of 15% and 20% for feature representation in no-take zones and MPAs (both take and no-take) respectively (S.D.Holness *pers. comm.*)[15]. In order to contribute to achieving these national targets, target values for local MPA design need to be higher, with the goal of improving representation of features currently poorly represented in MPAs and to reflect the local management

[15] Senior Manager: Strategic Parks Planning and Development, Conservation Services, South African National Parks

objectives based on specific concerns or issues. Conservation targets can also vary between features in order to reflect differing importance and greater need for improved protection. In this study target levels were therefore based these considerations.

Target levels of 25% were selected for each intertidal and subtidal habitat class, and 15% of each depth category within the planning domain to ensure protection of representative areas of all habitat features (Table 7.1). The stocks of numerous linefish species are considered depleted or collapsed (Mann 2000) which led to the declaration of an emergency in the linefishery in 2000 (DEAT 2000). Enhanced protection of these species in the future is therefore a critical management objective for the South African linefishery, and one of the main management objectives for the development of the AENP MPA. In order to enhance the protection of depleted and heavily targeted reef fish stocks, higher target levels were set for the conservation of reef areas, with 50% and 30% targets set for confirmed[16] reef and unconfirmed reef areas respectively. These reef areas support fish communities which are long-lived and highly resident and are therefore susceptible to overexploitation and in need of higher levels of protection in line with the conservation objectives of the MPA.

Areas adjacent to estuary mouths were identified as important process areas due to the migratory corridor they provide between estuarine and marine environments, and the importance of estuarine habitats to many marine species (Harrison 2003). Numerous species are therefore more abundant and susceptible to fishing pressure adjacent to estuary mouths, few of which currently receive formal protection in the MPA network. Few estuaries are present within Algoa Bay, but both of the permanently open estuaries within the planning domain are highly regarded in terms of their ecological value, with the Swartkops and Sundays estuaries ranked as 11th and 39th in terms of their conservation importance out of the 250 estuaries along the South African coastline (Turpie *et al.* 2002). In order to capture their ecological importance in the conservation planning process and contribute to improved protection of species occurring in these areas, a target level of 70% was set for the estuary associated nearshore process areas (Table 7.1). Lower targets of 20% and 10% were set for the silver kob and demersal nursery areas respectively. This was due to the lower spatial accuracy with which these areas were defined. Due to the ecological importance and sensitivity of the islands and immediate subtidal habitats (Newman and Klages 2001), and the limited number of these habitats along the east coast of southern Africa, targets of 100% were set for these features to ensure complete incorporation in no-take zones. The wider island surrounds also support diverse and abundant communities and are important foraging areas for nesting birds during the breeding season. In order to enhance the protection of these surrounding waters, while not forcing total inclusion, a target of 30% was set for this feature.

Due to historical overexploitation and poaching of abalone in South Africa (Steinberg 2005; Edwards and Plagányi 2008; Raemaekers and Britz 2009) and the limited distribution of suitable habitat within Algoa Bay, the protection of representative areas to safeguard this species was seen as a critical

[16] Confirmed reefs refers to reef complexes identified through side scan sonar data, mapping conducted for this project, SANHO substrate information, reef locations published in books, or a combination of these, while unconfirmed or potential reefs are locations which were provided by stakeholders but which have not been validated by alternate data sources.

management objective for the AENP. Targets were therefore set high at 75% of the available shallow (<15m) reef habitat in areas where they are known to occur (Table 7.1). Although protection of resident subtidal and intertidal reef fish communities was also considered an important management objective, the spatial resolution of the data were not very precise, hence lower targets of 25% and 20% were set respectively. The targets for nearshore linefish species and penguin foraging areas were set at 15%, while those for wide ranging and migratory offshore linefish species was set at 10% due to the lower spatial resolution of the data.

7.2.3 Systematic conservation planning analyses

Systematic conservation planning analyses were conducted with Marxan (Ball and Possingham 2000; Possingham *et al.* 2000) using the CLUZ interface for ArcView 3.2 (Smith 2005a). Marxan is designed to solve the minimum set reserve design problem, which aims to achieve the defined conservation targets at minimum cost[17] within the study area (Ball *et al.* 2009; Moilanen *et al.* 2009). The programme's optimisation algorithm uses simulated annealing and iterative improvement to identify multiple near-optimal solutions to the minimum set reserve design problem. Incorporating cost involves minimisation of a combination of spatially variable socio-economic data across the study area, and the boundary length of the reserve systems. Incorporation of spatial cost data aims to reduce the size of the reserve design, while minimising boundary length aims to increase the compactness of the reserve design. These factors therefore aid in designing reserves which are practical to implement and manage.

Marxan outputs include the "Best" solution, which is the spatial output of the run which achieved all targets at the lowest cost from each analysis, and the "selection frequency", which is the number of times an individual PU is selected out of the number of runs in the analysis and is therefore indicative of the conservation importance of a PU in achieving the defined targets (Stewart *et al.* 2007; Grantham *et al.* 2011). As the selection frequency is based on the results of multiple runs it provides a better indication of the conservation importance of a PU in attaining the targets than that of the single best solution (Grantham *et al.* 2011). An iterative or stepwise approach using the selection frequency output was used to systematically select the most important 5% of PUs for achieving the conservation targets. The status of the top 5% of PUs was changed from "available" to "conserved" so that they were included in the reserve design from the onset of the next Marxan analysis. Marxan was then re-run to identify the next most important 5% of PUs for achieving the conservation targets of which the status was again changed to "conserved". Marxan was run 100 times with simulated annealing followed by iterative improvement in each analysis in this stepwise procedure until all conservation targets were achieved. Once all targets were met using the 5% stepwise procedure, PUs with the lowest selection frequency were sequentially removed to determine the absolute minimum set required to achieve all conservation targets thereby reducing the area. This stepwise procedure resulted in identification of a reserve design based on the incremental importance of PUs in achieving the conservation targets.

[17] Cost does not specifically refer to economic value but can either mean an opportunity cost or area cost which is used to minimise the size of the reserve design

One of the key objectives of this study was to identify different reserve design options which meet conservation objectives, and evaluate the implications for fisheries activities. Marxan minimises the cost of reserve design based on a single spatial layer incorporated into the model. Several different cost layers were therefore utilised to develop a range of reserve design scenarios which could be evaluated in terms of their impacts to fisheries (Table 7.2). The cost data therefore differed per scenario based on specific objectives (Table 7.2), but initial values were set equal to the area of the PU in order to discount the spatial effects of fisheries on reserve design. Species penalty factor (SPF) values were set high at one million to ensure greater importance of selecting PUs to meet conservation targets over minimising costs or boundary length. The influence of boundary length modifier (BLM) values on reserve design was initially investigated in Scenarios 1 and 2 (Section 7.2.4) and the value which produced the optimal spatial output in respect of reserve compactness was used for all further scenario analyses. The compactness of reserve designs were assessed based on the ratio of the boundary length from the reserve design to the circumference of a circle of the same area (Possingham *et al.* 2000).

7.2.4 Spatial costs layers and planning scenarios

Eight scenarios were used to investigate the influence of different objectives and opportunity costs on reserve design. Marxan takes into account spatially variable cost data so as to avoid areas of high opportunity cost where conservation features are present in alternative sites, thereby reducing the overall socio-economic impact of spatial restrictions on area closures. The primary objective of the study was to identify the most suitable locations for the establishment of no-take zones in Algoa Bay for the protection of marine biodiversity and to ensure long-term persistence of populations. Non-consumptive recreational and tourism activities are unlikely to be affected through spatial zoning of consumptive use and were therefore not considered in the development of cost layers for this study. Impacts to fisheries were evaluated in terms of effort displacement per sector.

(a) Scenario 1: Biodiversity priority areas ("starting from scratch")

Scenario 1 was used to identify a set of priority areas for conservation of biophysical features irrespective of existing reserves and MPAs, or opportunity costs. All PUs were considered equal with their costs equivalent to their area and the status of all PUs was set as "available" for the analysis (Figure 7.2a).

(b) Scenario 2: Optimal expansion (expanding the existing reserves)

Scenario 2 was used to investigate the optimal spatial expansion of the current St Croix reserves and Bird Island MPA required for achieving the conservation targets for Algoa Bay. All PUs which were partially (>25%) within an existing reserve or MPA area were set as "conserved" and therefore included in the reserve design from the onset of the analysis with additional PUs selected to complement the reserve design. As in Scenario 1 opportunity costs were not considered and the cost of each PU was set equal to the area (1km^2) (Figure 7.2a). Scenario 2 is representative of the best expansion strategy for no-take areas and is used as the status quo against which alternative designs in which spatially explicit opportunity costs were considered are compared.

(c) Scenarios 3: Minimisation of recreational opportunity costs (expansion while minimising impact to recreational fishery activities)

The influence of spatially explicit recreational opportunity costs on reserve design was investigated using the index of relative recreational importance (IRRI) (Figure 7.2b) developed in Chapter 5. The IRRI was scaled to a maximum value of 9, and the area of each PU (1km^2) was added to create an opportunity cost layer with a range from one to ten. As in the status quo, the existing reserve areas were "locked" in the analysis as their status is unlikely to change in the future irrespective of the outcomes of the conservation planning exercise.

(d) Scenario 4: Minimisation of commercial opportunity costs (expansion while minimising impact to commercial fishery activities)

The influence of spatially explicit commercial opportunity costs on reserve design was investigated using the index of relative commercial importance (IRCI) (Figure 7.2c) developed in Chapter 6. The relative contribution of each sector was scaled based on national importance of Algoa Bay to the fishery sector (see Chapter 6). As above in Scenario 3 the IRCI was scaled to a maximum value of 9, and the area of each PU (1km^2) was added to create an opportunity cost layer with a range from one to ten. As in the status quo, the existing reserve areas were "locked" in the analysis.

(e) Scenario 5: Minimisation of combined opportunity costs (expansion while minimising impact to both recreational and commercial fisheries activities)

The scaled IRRI and IRCI were combined to create an Index of Total Importance (ITI) and a total opportunity cost layer with a range from one to ten (Figure 7.2d). As in the status quo, the existing reserve areas were "locked" in the analysis.

(f) Scenario 6: Economic value (expansion while minimising additional economic impact)

The economic value of each commercial and recreational sector varies considerably based on the target species, capital investment and size of the industry. In Scenario 6, the influence of the economic importance of each sector on the reserve design outputs was investigated. The economic indices of relative importance developed for the recreational and commercial sectors in Chapters 5 and 6 respectively were integrated into a single spatial economic index. The economic value for each commercial sector was scaled by the proportion of the average number of rights holders (2002-2008) reporting catches in Algoa Bay to the number of rights issued nationally per sector (see Chapter 6). The direct economic value of the recreational shore and skiboat fisheries in Algoa Bay was estimated based on the number of fishing days calculated per annum (Chapter 5) multiplied by the average daily expenditure of participants. The economic values of the recreational fisheries were not scaled as only the Algoa Bay fishery was considered in the estimates. The total economic value of the Algoa Bay fisheries was calculated as the sum of all fishery sectors, and the relative proportion of each sector to the total economic value of Algoa Bay was used to scale the proportion of effort per PU for each fishery sector. These values were then added across all sectors and scaled to an economic cost layer with a maximum range of one to ten (Figure 7.2e) as described above to create an Index of Economic Importance (IEI). Existing reserves and proposed no-take areas were "locked" into the analysis.

(g) Scenario 7: Evaluation of proposed AENP footprint

The proposed AENP MPA footprint was originally defined based on specialist input taking into consideration the coastal extent of the existing terrestrial park so as to create linkages between the terrestrial and offshore marine environments. As no quantitative assessments were conducted when the AENP MPA footprint was defined it may not adequately represent the biophysical features within Algoa Bay. Scenario 7 aimed to investigate the effectiveness of the proposed footprint in achieving the conservation objectives for Algoa Bay as well as to determine the optimal design of no-take areas within the footprint. Conservation features which did not occur within the footprint were removed for this analysis, and where feature targets could not be achieved within the footprint, the target amount was reduced to equal 100% of the feature abundance within the footprint. All PUs outside of the proposed AENP MPA footprint were set as "unavailable" and were therefore excluded from the analysis, forcing the selection of PUs from within the footprint boundary. Existing reserves and proposed no-take areas were "locked" into the analysis and the ITI cost layer was used.

(h) Scenario 8: Evaluation of proposed no-take areas and recommendations for expansion

Within the proposed AENP MPA footprint no-take areas have been recommended based on expert opinion. The contribution of these proposed no-take areas to the conservation targets for Algoa Bay were assessed in Scenario 8 and options for expansion to fulfil the conservation objectives were investigated. Existing reserves and proposed no-take areas were "locked" into the analysis and the ITI cost layer was used.

Table 7.2. Scenarios considered for designing a reserve network in Algoa Bay.

Scenario	Description	Objective
1 Priority areas	Existing reserves/MPAs not considered Cost equals area	Objective 1: To identify priority areas for conservation
2 Optimal expansion	Existing reserves locked into analysis Cost equals area	Objective 2: To determine optimal strategy for expansion of existing reserves and MPAs
3 Recreational costs [18]	Existing reserves locked into analysis Recreational costs only	Objective 3: To determine optimal reserve design taking opportunity costs and economic value into consideration
4 Commercial costs [19]	Existing reserves locked into analysis Commercial costs only	
5 Total Costs [20]	Existing reserves locked into analysis Total costs	
6 Economic costs [21]	Existing reserves locked into analysis Cost layer weighted by economic value of each sector	
7 Evaluation of AENP MPA footprint	Existing reserves locked into analysis PUs outside footprint excluded Total costs	Objective 4: To evaluate the effectiveness of the proposed footprint and no-take zones
8 Evaluation on AENP MPA no-take areas	Existing reserves and proposed no-take areas locked into analysis Total costs	

[18] Cost implies opportunity cost based on index of relative recreational effort (IRRI)
[19] Cost implies opportunity cost based in index of relative commercial effort (IRCI)
[20] Cost implies opportunity cost based on integrated index of relative recreational and commercial effort
[21] Cost based on relative economic value of fisheries locally in Algoa Bay

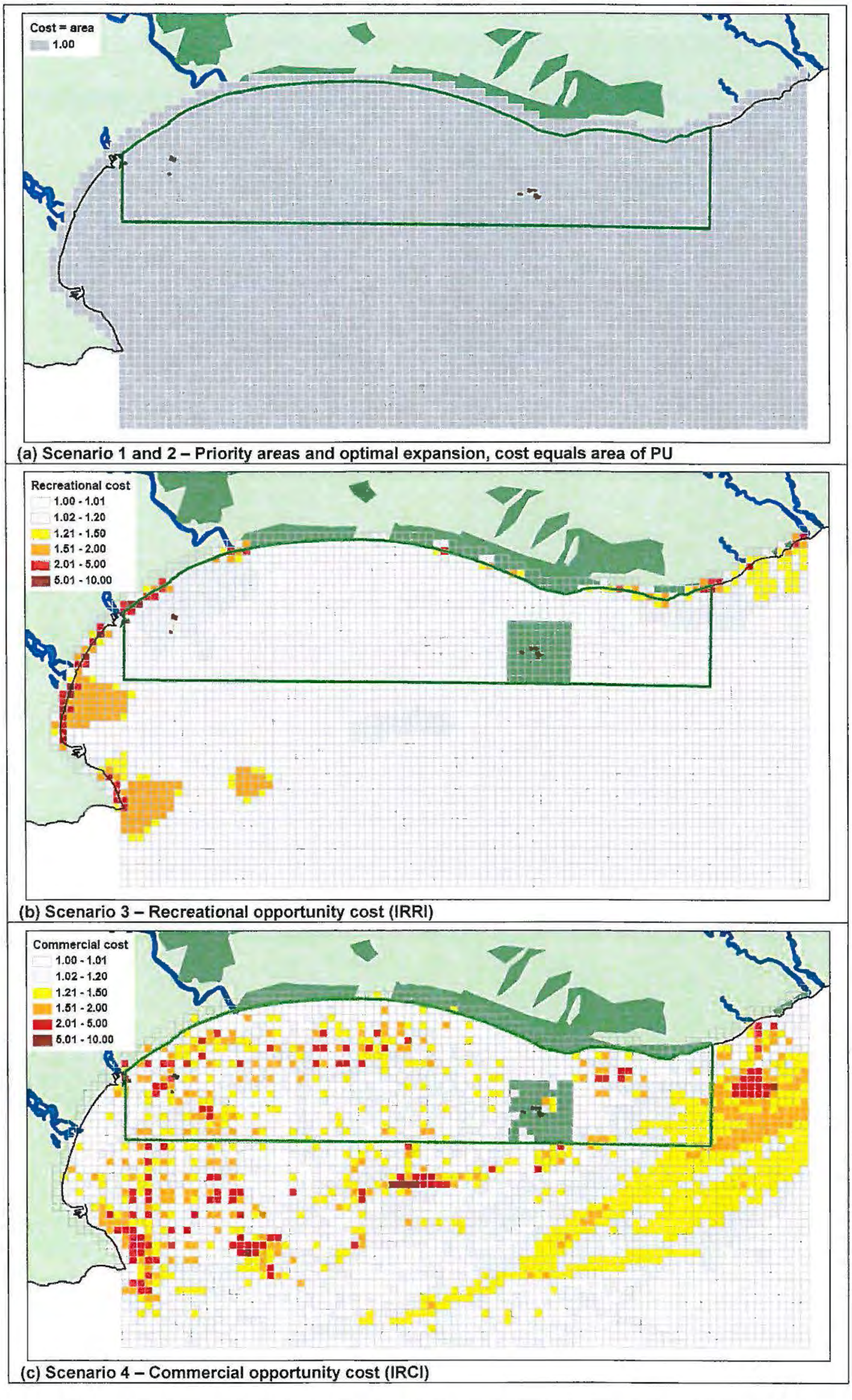
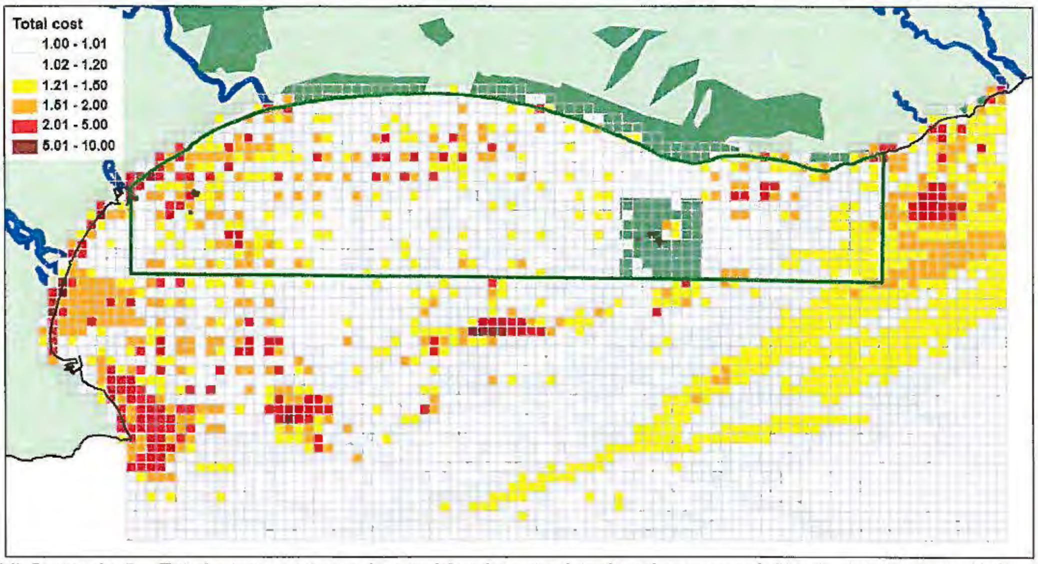
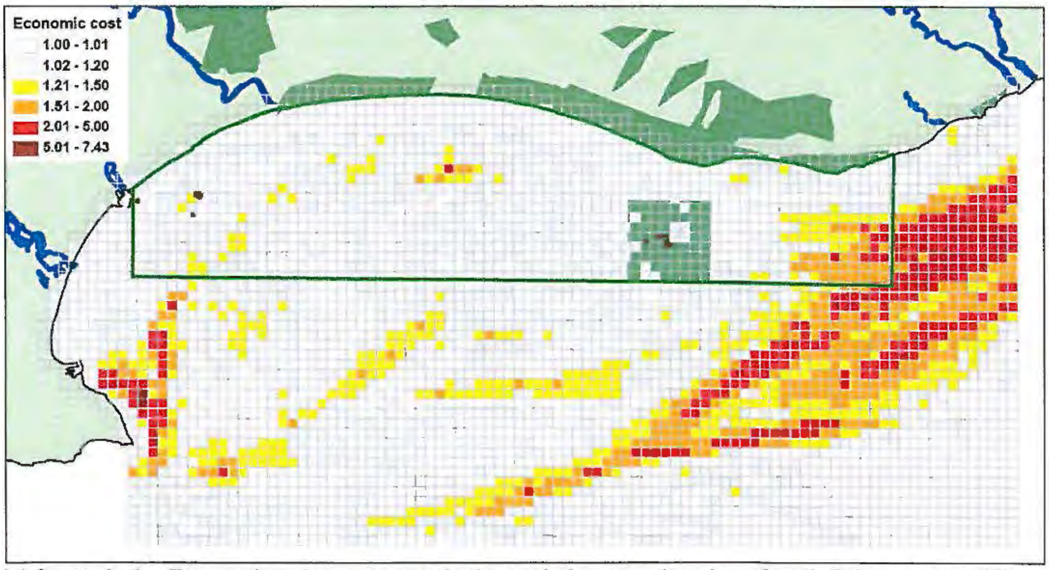


Figure 7.2. Cost layers used in SCP analyses. (a) Cost equals area (Scenario 1 and 2); (b) Cost equals recreational effort (Scenario 3); and (c) Cost equals commercial effort (Scenario 4). Dark green shaded areas indicate existing AENP, green line indicates proposed AENP MPA footprint.



(d) Scenario 5 – Total cost, cost equals combined recreational and commercial opportunity costs (ITI)



(e) Scenario 6 – Economic value, cost equals the scaled economic value of each fishery sector (IEI)

Figure 7.2 Cont. Cost layers used in SCP analyses. (d) Cost equals combined recreational and commercial effort (Scenario 5); and (e) Cost equals economic value (Scenario 6). Dark green shaded areas indicate existing AENP, green line indicates proposed AENP MPA footprint.

7.2.5 Contribution of Algoa Bay and the proposed AENP MPA to regional conservation

The contribution of the Algoa Bay, the proposed AENP footprint and proposed no-take zones were evaluated in terms of the representivity of key offshore and coastal features used in the 2011 NBA (S.D.Holness *pers. comm.*). Thirty-five coastal, subtidal and pelagic features used in the NBA occur within the Agulhas Bioregion and were used for the assessment. Feature abundance data of all features within the bioregion were populated for each PU in Algoa Bay using ArcView 3.2. The contribution of Algoa Bay to regional conservation was assessed through increases in the percent of targets achieved for different planning scenarios.

7.3 Results

7.3.1 Priority areas for conservation and optimal reserve design

Scenarios 1 and 2 resulted in the selection of similar areas producing comparable spatial reserve designs in order to achieve the conservation targets for Algoa Bay (Figure 7.4). The boundary length modifier (BLM) value had a considerable effect on the compactness of the areas selected for protection with BLM values less that 0.5 leading to highly fragmented reserve designs (Figure 7.3; Figure 7.4). For both scenarios 1 and 2 (all BLM values) between 25 and 27% of PUs were required to meet the conservation objectives for Algoa Bay representing between 1 017 and 1 132 of the PUs available for selection. A greater area was required to meet the conservation targets with lower BLM values as less emphasis was placed on selecting neighbouring Pus, leading to lower selection frequencies of PUs overall over a wider spatial scale.

In Scenario 1, where no reserve areas were "locked" into the analysis, PUs in the St Croix and Bird Island areas were selected most frequently and are therefore areas of greatest priority. Additional areas of greatest importance include the Cape Recife point, the estuarine mouth areas of Swartkops, Sundays and Boknes estuaries, and the inshore regions of the Woody Cape and Cape Padrone headlands. In Scenario 2 the existing reserves held 25 of the 36 conservation features; however, conservation targets were only fully met for six features out of 36 (Table 7.3). Similar spatial selection of high priority areas to Scenario 1 were identified when additional PUs were selected for reserve expansion. However, due to the representation of several features within the Bird Island MPA less extensive areas were selected at Cape Recife and around the Woody Cape and Cape Padrone headlands. The existing reserve network would need to be expanded by a minimum of 14 times in order to achieve the desired conservation objectives for Algoa Bay.

Scenario 2 with a BLM value of one required the least amount of PUs (1 017) to meet the conservation targets and also resulted in a compact design from which practical options for reserve implementation could be identified (Figure 7.3). As a result a BLM value of one was used for all further SCP analyses.

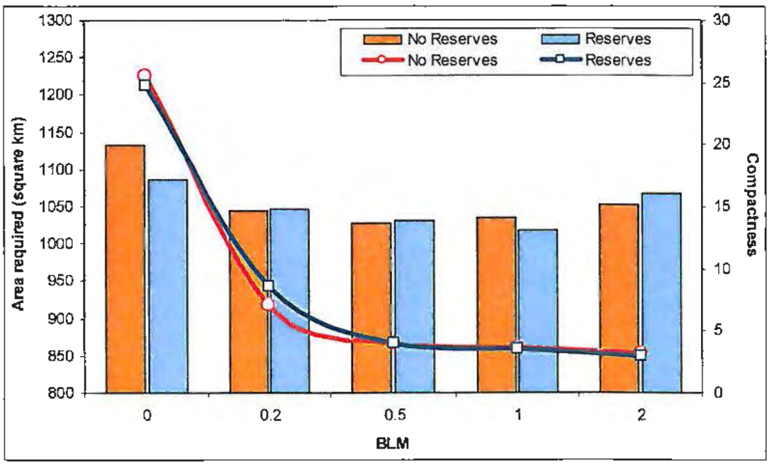


Figure 7.3. Area required (bars) and the compactness (lines) of the reserve designs under differing BLM values for Scenario 1 and 2.

7.3.2 Incorporating fisheries costs into reserve design

The incorporation of spatially explicit opportunity cost data into the SCP analysis resulted in an overall reduction in the associated impacts to all commercial and recreational fisheries while still achieving all conservation targets with only modest increases in the amount of area required (Figure 7.5; Figure 7.6). The efficiency (area required) of each scenario differed based on the different cost layers. However, there was a maximum difference of only 2% in the area required between the least (Scenario 6: Economic value) and most efficient (Scenario 4: Commercial costs) scenarios (Figure 7.5). Although Scenario 6 was the least efficient it produced the most compact reserve design, while Scenario 4 and 5 were the least compact. The magnitude in difference in compactness, however, was relatively small (Figure 7.5).

Scenarios 1 and 2, in which no spatially explicit opportunity cost data were considered, resulted in the greatest overall impact on fishery activities, with a 41 and 36% displacement of effort respectively (Figure 7.6). Recreational fisheries were most affected (53% for each scenario) with a displacement of 33 and 38% of shore fisheries effort, and 73 and 68% of skiboat fisheries effort in Scenario 1 and Scenario 2 respectively. Commercial fisheries were affected by a 36 and 30% displacement of effort in Scenario 1 and Scenario 2 respectively, with the chokka-squid fishery influenced most (58 and 43% respectively), followed by the shark longline (41 and 32% respectively), traditional linefishery (36 and 34% respectively), SPPSF (25 and 22%) and lastly the demersal trawl sector (21% for each scenario).

The incorporation of recreational opportunity cost data (IRRI Chapter 5) in Scenario 3 resulted, with the exception of the demersal trawl fishery, in reduced impact to all fisheries when compared to the status quo (Scenario 2) (Figure 7.6; Table 7.4). The estimated loss to all fisheries was below 30% of current levels of utilisation. The overall impact was reduced by 17%, while the recreational and commercial sectors benefited from a 36% and 9% reduction in impact due to spatial displacement of effort (Table 7.4). The recreational skiboat fishery benefitted the most with a 46% reduction in effort displacement from the status quo, followed by the commercial chokka-squid and recreational shore fisheries with 32% and 25% reductions in displacement respectively. Spatially similar priority areas were selected to the status quo; however, the reduced impact on fisheries can largely be attributed to the selection of fewer PUs around Cape Recife.

Table 7.3. Conservation feature targets achieved by the existing reserve network in Algoa Bay (St Croix reserves and Bird Island MPA). Features highlighted in grey represent targets which were achieved with the current reserve design. Features are listed in descending order based on the percentage of target achieved.

Conservation feature	ID	Feature category	Amount conserved	% of target achieved
Intertidal reef fish	26	S	8	149
Island - surrounds	22	P	34	142
Kob nursery	17	P	8	133
Penguin foraging BI (b)	34	S	45	124
Subtidal reef fish Group 2	30	S	34	114
Island - terrestrial	20	P	1	100
Subtidal gravel	6	H	71	90
Penguin foraging BI (a)	33	S	28	85
Island - intertidal	21	P	17	85
Subtidal reef fish Group 1	29	S	6	81
Shallow	13	H	50	58
Subtidal reef - potential	12	H	64	54
Abalone distribution	31	S	11	53
Subtidal reef - confirmed	11	H	45	52
Silver kob	28	S	41	38
Medium	14	H	26	24
Geelbek	27	S	41	22
Dusky kob	23	S	9	15
Demersal nursery	19	P	9	11
Penguin foraging StC (b)	36	S	5	9
Leervis	25	S	1	7
Deep	15	H	11	5
Subtidal sand	9	H	17	3
Penguin foraging StC (a)	35	S	3	3
White steenbras	24	S	0	2
Intertidal rock	1	H	0	0
Intertidal rock above sand	2	H	0	0
Intertidal rock and sand	3	H	0	0
Intertidal sand	4	H	0	0
Intertidal sand above rock	5	H	0	0
Subtidal mud	7	H	0	0
Subtidal muddy-sand	8	H	0	0
Subtidal sandy-mud	10	H	0	0
Very deep	16	H	0	0
Estuary link	18	P	0	0
A. australis distribution	32	S	0	0

Feature categories: H=habitat, P=process areas, S= species distributions

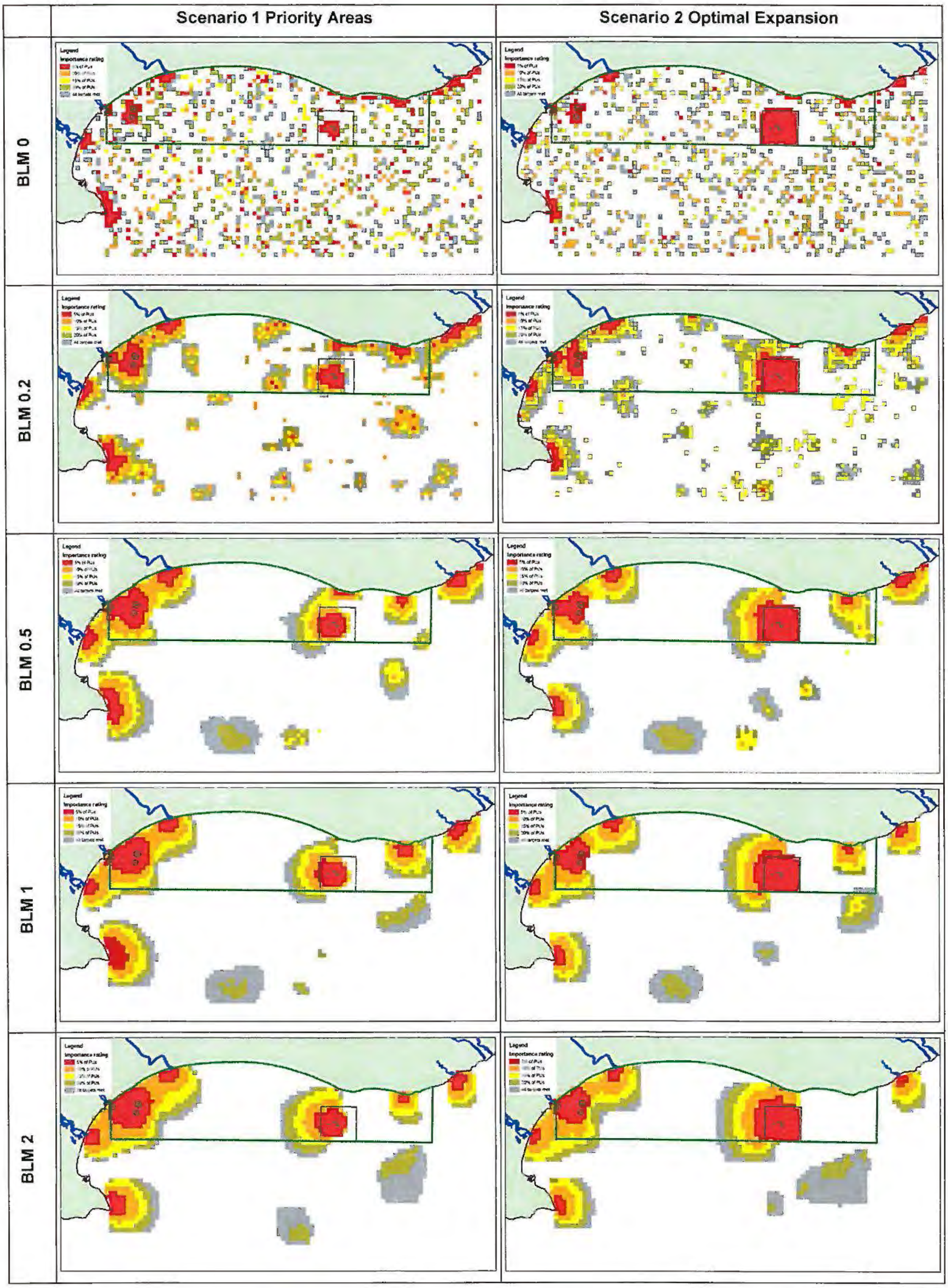


Figure 7.4. Priority conservation areas identified in Algoa Bay to meet feature targets with different BLM values for Scenario 1 (No Reserves) (left) and Scenario 2 (Reserves locked-in) (right). Graded colours represent the importance of planning units in meeting feature targets based on selection frequency. Dashed black line indicates existing reserve boundaries, and green line indicates the proposed AENP MPA footprint.

Legend	
Importance rating	
	5% of PUs
	10% of PUs
	15% of PUs
	20% of PUs
	All targets met

Scenario 4 substituted recreational with commercial opportunity cost data (IRCI Chapter 6) and when compared to the status quo resulted in a reduction in the impact on all fisheries combined (18%), as well as the commercial (18%) and recreational (17%) sectors (Table 7.4). Although the impact on recreational fisheries was less than in the status quo, they remained relatively high with a 35 and 37% displacement of effort in the recreational shore and skiboat fisheries respectively (Figure 7.6). The impact on all commercial fisheries was below 23% of the current level of utilisation (Figure 7.6), and with the exception of the SPPSF (increase of 1%), there was a reduction in the amount of impact for all fisheries from the status quo (Table 7.4). Spatially the PUs selected to meet conservation targets were similar to Scenario 3, with few PUs selected around Cape Recife. The greater reduction in impact for the commercial than the recreational fisheries was due to a reduction in the number of PUs selected in the eastern region of the proposed AENP MPA footprint and around the Sundays Estuary mouth.

A combination of the IRRI and IRCI opportunity cost data in Scenario 5 resulted in the lowest overall impact on all fisheries with a total displacement of 14% (Figure 7.5), which constituted an improvement of 23% from the status quo (Table 7.4). Impacts to the commercial and recreational sectors were 12 and 18% respectively (Figure 7.6), representing a 19 and 35% improvement from the status quo for each sector (Table 7.4). The spatial selection of PUs was similar to that in Scenario 4; however, more area was required to meet the targets and the design was the least compact (Figure 7.5).

Incorporating the economic value of each fishery sector into the SCP analysis resulted in the least efficient design in terms of area required, yet most compact reserve design (Figure 7.5; Figure 7.6). The overall impact to all sectors resulting from the design was a 27% displacement of effort, which was a 10% improvement from the status quo, yet 13% higher than Scenario 5 (combined recreational and commercial opportunity costs). The impact to the commercial and recreational sectors was 22 and 40% respectively (Figure 7.6), which was a 9 and 10% improvement from the status quo for each sector (Table 7.4).

7.3.3 Evaluation of the proposed AENP MPA footprint and no-take areas

The proposed AENP MPA footprint represents 32% of the study area, covering a total area of 1 302km^2. The footprint was unable to meet targets for seven of the conservation features identified in Algoa Bay, with two features (muddy substrata and very deep depths) not present at all (Table 7.5). These were therefore excluded during further analyses investigating potential no-take design options for within the footprint boundaries. Target levels of the five features (Table 7.5) that were present in insufficient quantities were reduced to equal 100% of the feature abundance within the footprint area. Upon reduction of the conservation targets for these five features a total of 908 PUs (70% of the footprint area) were required to achieve the conservation objectives using the combined recreational and commercial cost layer (Scenario 5 ITI) (Figure 7.7). Although not directly comparable to previous scenarios as the conservation targets differ, this reserve design resulted in an overall displacement of 16% of fishing effort, and displacement of 11 and 18% of the recreational and commercial sectors respectively (Figure 7.7).

The proposed AENP no-take zones (638 PUs) represent 49% and 15% of the proposed AENP footprint and Algoa Bay area considered in this assessment respectively (Figure 7.7). Overall targets for 21 of the 34 conservation features were achieved within the proposed no-take areas (Table 7.6). The proposed no-take areas resulted in a displacement of 11% of the overall fishing effort, with a 7% and 13% displacement of recreational and commercial effort respectively (Figure 7.7). The proposed no-take areas had the greatest impact on the demersal shark longline fishery (23%), followed by the traditional linefish (15%), SPPSF (14%) and the recreational skiboat (11%) fisheries, with effort displacement to all other fisheries below 10%. Potential designs for expansion of the proposed AENP no-take areas to achieve the conservation targets (revised) for Algoa Bay are illustrated in Figure 7.8.

Table 7.4. Change in effort displacement to fishery sectors based on results of Scenario 2 Optimal expansion with the inclusion of different spatial cost layers. Negative values indicate a reduction in effort displacement from Scenario 2, positive values an increased displacement of fisheries effort.

Fishery sector	Cost layer used in analysis			
	Scenario 3 Recreational	Scenario 4 Commercial	Scenario 5 Combined	Scenario 6 Economic
Commercial fisheries				
Shark longline	-3	-20	-22	-1
Demersal trawl	3	-13	-14	-18
SPPSF	-2	1	0	8
Chokka-squid	-32	-38	-38	-31
Traditional linefish	-13	-21	-19	-2
Total Commercial	-9	-18	-19	-9
Recreational fisheries				
Shore	-25	-3	-20	-5
Skiboat	-46	-31	-50	-20
Total Recreational	-36	-17	-35	-12
TOTAL	-17	-18	-23	-10

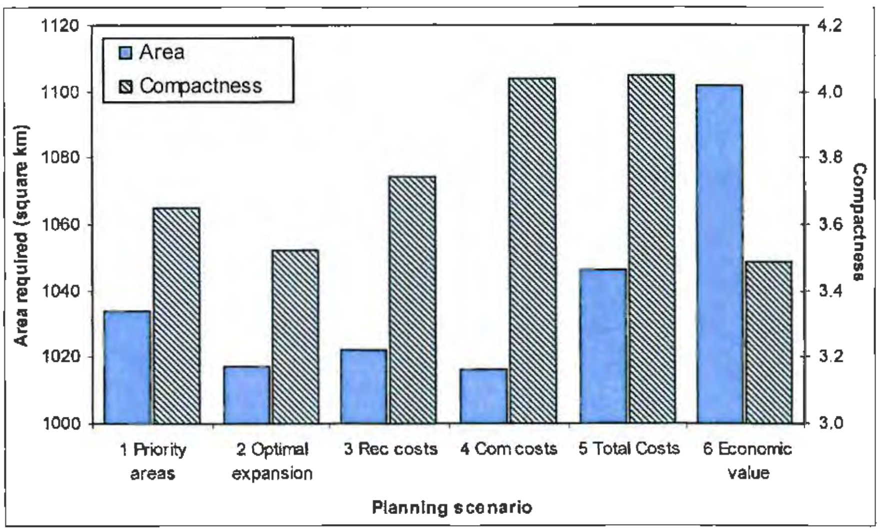


Figure 7.5. The efficiency with which each scenario achieves the conservation targets in terms of area required (blue solid bars) and the compactness of the reserve design (green striped bars). BLM values of 1 were used for each scenario.

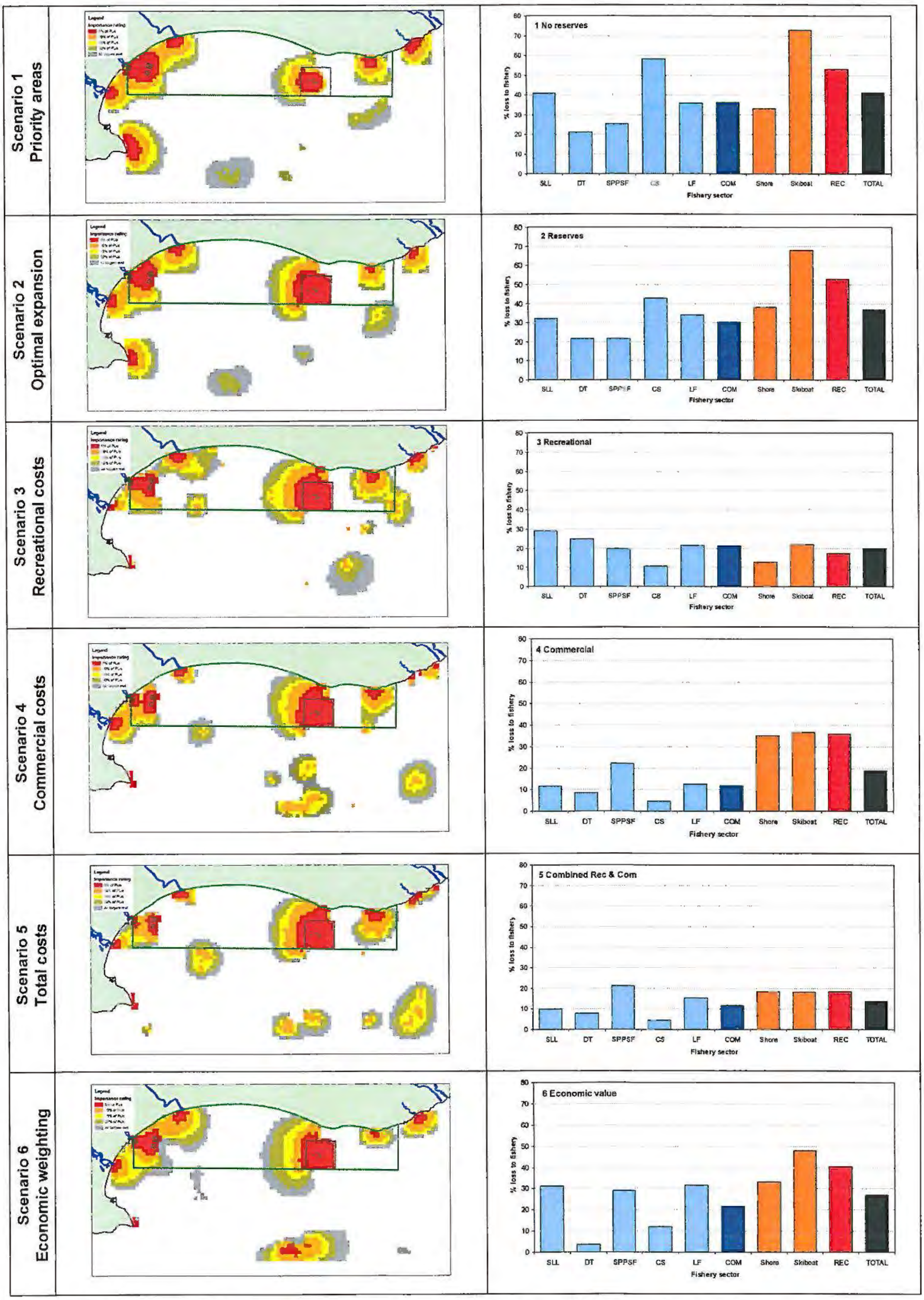


Figure 7.6. Design outputs for reserve networks in Algoa Bay based on different cost layers (left) and amount of fishing effort displaced per sector and cumulatively (right). Total effort displacement determined as equal weighting between sectors. SLL=shark longline; DT=demersal trawl; SPPSF=small pelagic purse seine fishery; CS=chokka-squid; LF=commercial linefish; COM=Total commercial; Shore=recreational shore; Skiboat=recreational skiboat; REC=Total recreational; TOTAL=Total of all sectors.

Legend
Importance rating
5% of PUs
10% of PUs
15% of PUs
20% of PUs
All targets met

Table 7.5. Amount and percentage of the feature target available within the proposed AENP MPA. Conservation features not meeting the targets within the footprint highlighted in bold and revised targets based on 100% of available feature within the footprint calculated. Conservation features excluded from further analysis highlighted in grey. Features listed in descending order based on percentage of target achievable.

Conservation feature	ID	Amount within Algoa Bay study area (km or km^2)	Original target (%)	Original target amount (km or km^2)	Amount within AENP MPA proposed footprint (km or km^2)	% of original target available	New target %
A. australis distribution	32	44	15	7	44	667	15
Penguin foraging BI (b)	34	240	15	36	207	576	15
Demersal nursery	19	760	10	76	406	534	10
Kob nursery	17	30	20	6	30	500	20
Dusky kob	23	389	15	58	281	482	15
Penguin foraging BI (a)	33	218	15	33	152	466	15
Shallow	13	572	15	86	377	439	15
Medium	14	722	15	108	463	427	15
White steenbras	24	165	15	25	103	416	15
Intertidal rock above sand	2	3	25	1	3	400	25
Intertidal rock and sand	3	4	25	1	4	353	25
Leervis	25	123	15	18	63	342	15
Intertidal reef fish	26	27	20	5	17	320	20
Island - surrounds	22	80	30	24	76	317	30
Silver kob	28	1076	10	108	324	301	10
Penguin foraging StC (a)	35	819	15	123	368	300	15
Intertidal sand	4	95	25	24	67	284	25
Geelbek	27	1863	10	186	449	241	10
Subtidal reef fish Group2	30	118	25	29	66	226	25
Subtidal gravel	6	317	25	79	171	216	25
Subtidal reef - potential	12	398	30	119	243	204	30
Intertidal rock	1	12	25	3	6	194	25
Subtidal reef fish Group1	29	31	25	8	15	192	25
Penguin foraging StC (b)	36	360	15	54	97	180	15
Deep	15	1324	15	199	334	168	15
Subtidal sand	9	2629	25	657	858	131	25
Subtidal reef - confirmed	11	174	50	87	97	111	50
Island - terrestrial	20	1	100	1	1	100	100
Island - intertidal	21	20	100	20	20	100	100
Abalone distribution	**31**	**28**	**75**	**21**	**17**	**80**	**75**
Subtidal muddy-sand	**8**	**723**	**25**	**181**	**116**	**64**	**16**
Intertidal sand above rock	**5**	**11**	**25**	**3**	**2**	**61**	**15**
Subtidal sandy-mud	**10**	**228**	**25**	**57**	**30**	**52**	**13**
Estuary link	**18**	**23**	**70**	**16**	**7**	**44**	**31**
Subtidal mud	***7***	***30***	***25***	***7***	***0***	***0***	***Excluded***
Very deep	***16***	***1305***	***15***	***196***	***0***	***0***	***Excluded***

Table 7.6. Amount and percentage of targets achieved for each conservation feature present within the proposed AENP MPA no-take areas. Features not meeting targets highlighted in grey. Features listed in descending order based on the percentage of target achieved.

Conservation feature	ID	Amount within Algoa Bay study area (km or km^2)	Original target amount (km or km^2)	Amount within AENP MPA proposed no-take (km or km^2)	% of original target achieved
Penguin foraging BI (b)	34	240	36	184	512
Kob nursery	17	30	6	30	494
A. australis distribution	32	44	7	30	460
Penguin foraging BI (a)	33	218	33	132	404
Intertidal rock above sand	2	3	1	2	334
Island - surrounds	22	80	24	74	309
Dusky kob	23	389	58	172	296
Shallow	13	572	86	229	267
Demersal nursery	19	760	76	195	257
Medium	14	722	108	222	205
Intertidal reef fish	26	27	5	10	195
Subtidal gravel	6	317	79	152	192
Subtidal reef fish Group 1	29	31	8	14	183
Subtidal reef fish Group 2	30	118	29	54	183
Leervis	25	123	18	34	182
White steenbras	24	165	25	37	151
Subtidal reef - potential	12	398	119	160	134
Intertidal sand	4	95	24	31	132
Penguin foraging StC (b)	36	360	54	67	123
Island - terrestrial	20	1	1	1	100
Island - intertidal	**21**	**20**	**20**	**20**	**99**
Subtidal reef - confirmed	**11**	**174**	**87**	**80**	**92**
Silver kob	**28**	**1076**	**108**	**89**	**82**
Geelbek	**27**	**1863**	**186**	**150**	**81**
Deep	**15**	**1324**	**199**	**135**	**68**
Intertidal rock and sand	**3**	**4**	**1**	**1**	**63**
Abalone distribution	**31**	**28**	**21**	**12**	**59**
Penguin foraging StC (a)	**35**	**819**	**123**	**73**	**59**
Subtidal muddy-sand	**8**	**723**	**181**	**101**	**56**
Subtidal sand	**9**	**2629**	**657**	**327**	**50**
Intertidal rock	**1**	**12**	**3**	**1**	**37**
Subtidal sandy-mud	**10**	**228**	**57**	**10**	**18**
Estuary link	**18**	**23**	**16**	**1**	**9**
Intertidal sand above rock	**5**	**11**	**3**	**0**	**0**
Subtidal mud	**7**	**30**	**7**	**0**	**0**
Very deep	**16**	**1305**	**196**	**0**	**0**

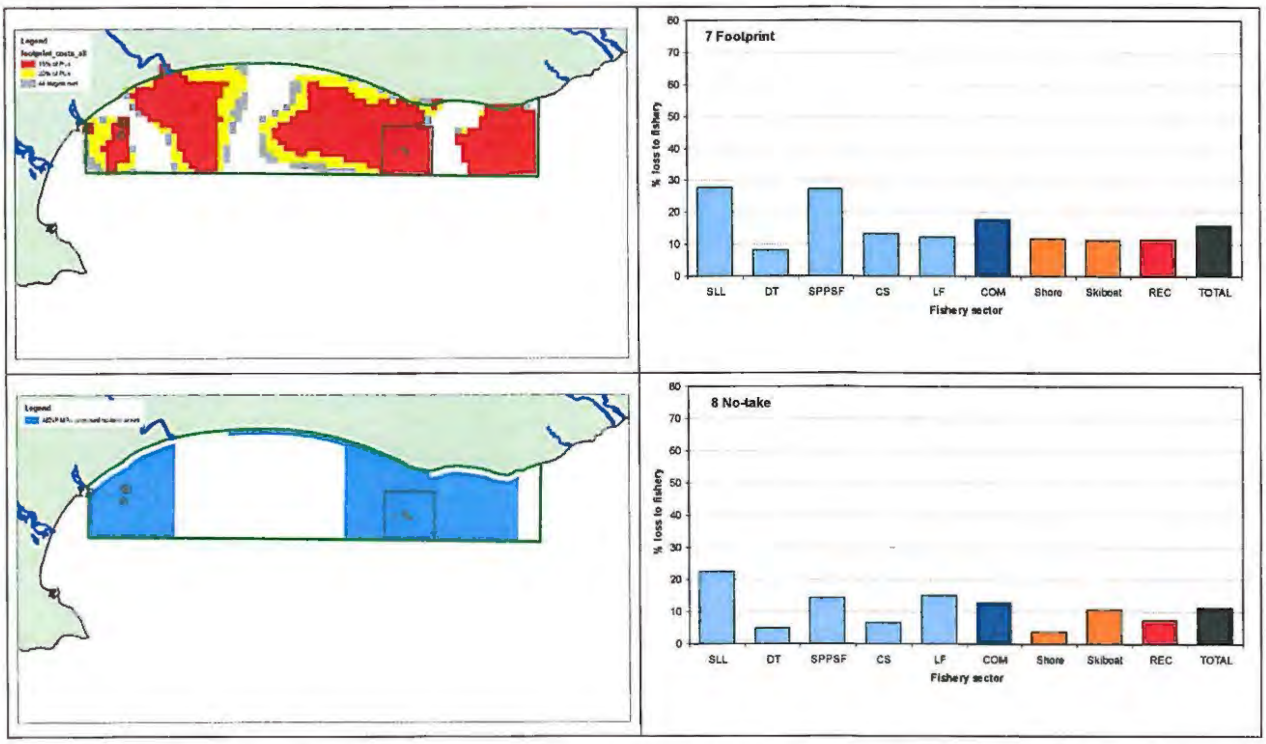


Figure 7.7. Optimal reserve design limited to the selection of PUs within the proposed AENP footprint and evaluation of fishing effort displacement (top). AENP proposed no-take areas and the estimated displacement of fishing effort in Algoa Bay per sector (bottom). Colour scale red-yellow-grey indicated areas of decreasing importance to meet targets.

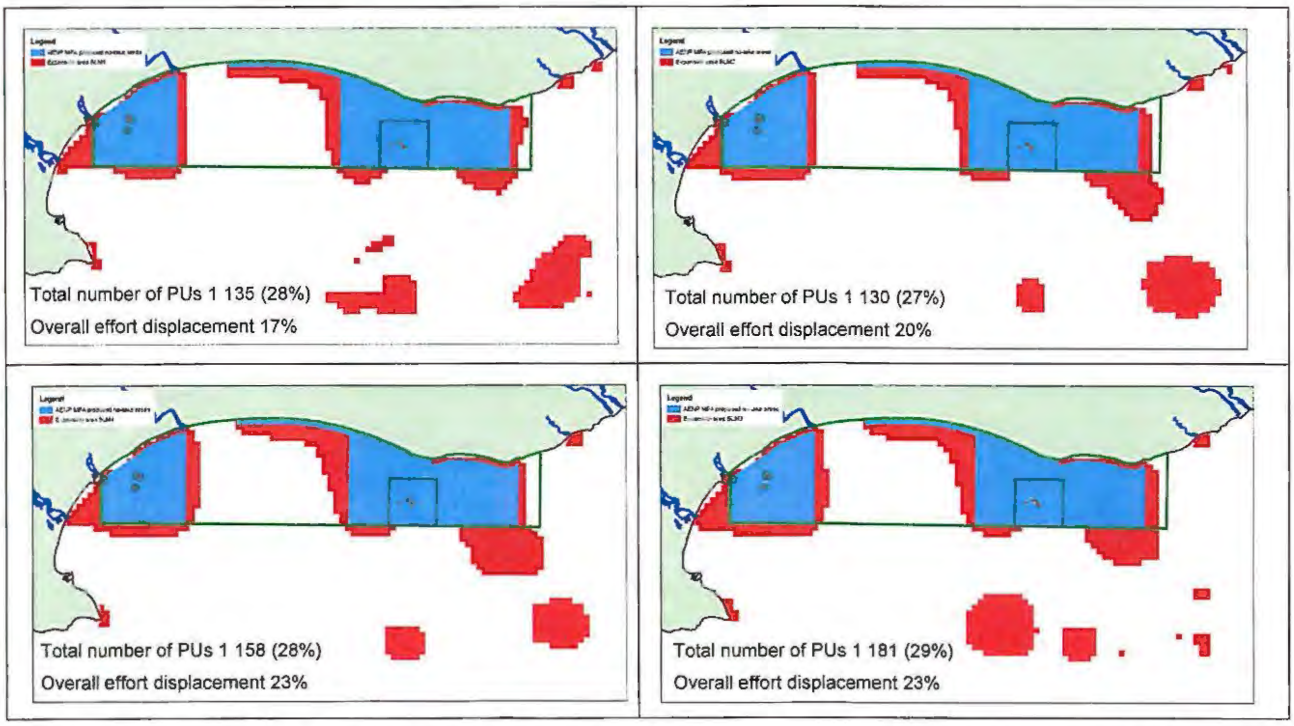


Figure 7.8. Optional designs for increasing the proposed AENP no-take areas to achieve all conservation targets for Algoa Bay using different BLM values (BLM 1-4 clockwise direction).

7.3.4 Contribution of Algoa Bay and the proposed AENP MPA to regional conservation

Within the Agulhas Bioregion only six of the 35 conservation feature[22] target levels are achieved by the current MPA network (Table 7.7). Twenty-two of the 35 conservation features were present within Algoa Bay (Table 7.7 and 7.8). The broader Algoa Bay study area contributed to increasing the representivity of 18 of these conservation features by more than 10% indicating the high conservation value of Algoa Bay to the bioregion (Table 7.8). Furthermore Algoa Bay contributed to achieving the regional MPA targets for an additional seven features (estuarine shore; intermediate sandy coast; sheltered and very exposed rocky coast; sandy inshore; island; mixed inner shelf sediment), increasing the number of regional conservation targets achieved from six to 13 (Table 7.7). The proposed AENP footprint increased the regional representivity of five features sufficiently to contribute to greater than 10% improvement in attaining regional targets (Table 7.8). Furthermore three features (estuarine shore; sandy inshore; island) were present in sufficient quantities within the proposed AENP footprint to successfully contribute to achieving regional targets, increasing the number of conservation targets successfully achieved from six to nine (Table 7.7).

Exposed rocky coast was the only feature for which both MPA and no-take targets were achieved within the bioregion prior to the inclusion of areas within Algoa Bay (Table 7.7). The Algoa Bay study area and proposed AENP footprint contributed to 17 and 3% increases in the representation of exposed rock coast within the bioregion respectively (Table 7.8). Regional no-take targets for exposed rocky coast were increased by <1, 4 and 9% for the proposed AENP no-take zones and the no-take areas identified in Scenario 5 and 6 respectively (Table 7.8).

With the exception of one feature, island, no additional regional no-take targets were achieved by the proposed AENP no-take zones and Scenario 5 and 6 no-take areas (Table 7.7). Island representivity was increased by 240, 157 and 246% by the proposed AENP no-take and scenario 5 and 6 areas respectively (Table 7.8). Despite not achieving regional no-take targets for numerous features, the proposed AENP no-take zones and scenario 5 and 6 areas contributed to 57, 51 and 63% increases in the representation of dissipative intermediate sandy coastline respectively (Table 7.8), increasing its representivity to between 87 and 99% of target values (Table 7.7). Scenario 5 and 6 contributed to 33 and 36% increases in the representivity of intermediate sandy coastline (Table 7.8) increasing representivity to 58 and 61% of target values respectively (Table 7.7). Both scenario 5 and 6 also contributed to 30 and 53% increases in the representivity of sandy inshore habitat (Table 7.8) resulting in achieving 65 and 88% of regional target levels (Table 7.7), while the proposed AENP no-take zones contributed 17%, increasing regional representivity to 52% of target values. The proposed AENP no-take zones made a significant contribution to the representivity of mixed inner shelf sediment increasing from 0 to 34% of target values, while scenario 5 and 6 each contributed to a 23% increase in achieving target values (Table 7.7 and Table 7.8). Scenario 5 also contributed to a 43% increase in representivity of very exposed rocky coast increasing overall representivity to 88% of target levels, while scenario 6 increased representivity to 64% of target levels (Table 7.7 and Table 7.8). Scenario 5 and 6 also resulted in moderate (>10%) increases in the representivity of estuarine and mixed shore, and muddy outer shelf habitats, but not the proposed AENP no-take zones (Table 7.8).

[22] 35 marine conservation features were identified in the Agulhas Bioregion and mapped for the 2011 National Spatial Biodiversity Assessment (NSBA)

Table Key

Protection level categories

Not protected	No formal protection
Hardly protected	Under 5% of Biodiversity ta met in MPA*
Poorly protected	5% - under 50% of Biodive target met in MPA
Moderately protected	50-under 100% of Biodive target met in MPA
Well protected	100% targets met in MPA not enough no-take
Well protected and no-take targets achieved	Targets fully met and suffi no-take

No-take MPA protection level

Enough	No-take targets met
Not enough	No-take targets not me
None	No representation

es in italics indicate no-change (nc) to regional targets.

% of MPA	Comparison of different AENP no-take options as % of no-take target		
ootprint d no-take 3	Proposed AENP4	Scenario 5 5	Scenario 6 6
	nc	*nc*	*nc*
	nc	*nc*	*nc*
	92.4	86.6	99.0
	45.8	58.1	57.6
	116.6	119.6	124.9
	24.7	57.6	60.8
	62.1	72.1	73.1
	nc	*nc*	*nc*
	0.0	0.0	0.1
	45.0	88.2	63.5
	nc	*nc*	*nc*
	nc	*nc*	*nc*
	62.0	67.9	62.0
	52.3	65.3	88.0
	294.3	211.5	300.2
	nc	*nc*	*nc*
	17.2	17.2	17.2
	0.0	0.0	0.0
	33.5	22.9	22.6
	0.0	0.0	0.0
	9.5	14.8	14.2
	nc	*nc*	*nc*
	3.6	3.6	3.6
	0.0	0.0	0.8
	0.0	0.0	0.0
	0.0	17.6	32.1
	nc	*nc*	*nc*
	0.0	5.2	2.0
	nc	*nc*	*nc*
	nc	*nc*	*nc*
	nc	*nc*	*nc*
	nc	*nc*	*nc*
	nc	*nc*	*nc*
	4.4	7.4	7.7
	0.0	4.0	0.5

Table 7.8. Percentage increase in conservation feature target achievement for MPA design option in Algoa Bay. nc = no change to representivity of conservation features, colour key below table.

Habitat category	Habitat type	As MPA		As no-take		
		Algoa Bay as take MPA	AENP footprint as take MPA	Proposed AENP no-take	Scenario 5	Scenario 6
Shoreline	Boulder Shore	nc	nc	nc	nc	nc
	Dissipative Sandy Coast	nc	nc	nc	nc	nc
	Dissipative-Intermediate Sandy Coast	128	91	57	51	63
	Estuarine Shore	[illegible]	8	nc	12	[illegible]
	Exposed Rocky Coast	17	3	1	4	9
	Intermediate Sandy Coast	[illegible]	nc	nc	33	36
	Mixed Shore	[illegible]	[illegible]	4	14	15
	Reflective Sandy Coast	nc	nc	nc	nc	nc
	Sheltered Rocky Coast	301	nc	nc	nc	nc
	Very Exposed Rocky Coast	[illegible]	nc	nc	43	[illegible]
Inshore	Inshore Gravel	nc	nc	nc	nc	nc
	Inshore Hard Grounds	nc	nc	nc	nc	nc
	Inshore Reef	55	nc	nc	6	nc
	Sandy Inshore	75	46	17	30	52
Island	Island	264	185	240	157	246
Inner shelf	Gravel Inner Shelf	nc	-	nc	nc	nc
	Hard Inner Shelf	[illegible]	4	nc	nc	nc
	Inner Shelf Reef	[illegible]	nc	nc	nc	nc
	Mixed Sediment Inner Shelf	155	57	34	23	23
	Muddy Inner Shelf	6	nc	nc	nc	nc
	Sandy Inner Shelf	[illegible]	9	nc	8	8
Outer shelf	Canyon	nc	nc	nc	nc	nc
	Gravel Outer Shelf	2	nc	nc	nc	nc
	Hard Outer Shelf	2	nc	nc	nc	1
	Mixed Sediment Outer Shelf	[illegible]	nc	nc	nc	nc
	Muddy Outer Shelf	73	nc	nc	18	32
	Outer Shelf Reef	nc	nc	nc	nc	nc
	Sandy Outer Shelf	11	nc	nc	5	2
Shelf edge	Gravel Shelf Edge	nc	nc	nc	nc	nc
	Hard Shelf Edge	nc	nc	nc	nc	nc
	Muddy Shelf Edge	nc	nc	nc	nc	nc
	Sandy Shelf Edge	nc	nc	nc	nc	nc
	Shelf Edge Reef	nc	nc	nc	nc	nc
Pelagic	Pelagic 1	[illegible]	4.1	1.3	4.4	4.6
	Pelagic 45	4.8	nc	nc	4.0	0.5

% improvement in feature representation
Zero improvement in achieving target
0-10% improvement
>10-20% improvement
>20-50% improvement
>50-100% improvement
>100% improvement

7.4 Discussion

The use of quantitative SCP methods for evaluating the conservation efficiency of existing MPA networks, identifying priority areas for conservation investment and the design of new MPAs has increased rapidly in recent years. Through its quantitative approach SCP provides a transparent, repeatable and defensible method to support decision-making as well as facilitate stakeholder engagement (Margules and Pressey 2000). Maps and basic statistics such as area required, number of targets achieved and socio-economic costs of various design options are easily understood and interpreted by all parties. This not only provides a basis for discussion with stakeholder groups, but also equips managers and decision-makers with the necessary alternatives to evaluate options for successful implementation and enforcement. The conservation benefits of SCP have been shown to outweigh those of opportunistic reserve design and selection procedures, but limited data availability for marine ecosystems and the costs of data collation and preparation have presented challenges in the past (Ban 2009; Hansen *et al.* 2011). Recent technological advances have facilitated the acquisition of the marine biophysical data required for SCP, and in conjunction with increased commitment to marine conservation worldwide have led to SCP becoming a widely accepted and preferred tool for marine conservation planning and the implementation of integrated ecosystem-based management. This study therefore applied a quantitative systematic planning approach to evaluate and investigate reserve network designs in Algoa Bay in line with current best practices.

The complexities of integrating ecological and socio-economic requirements have hindered the implementation of ecosystem-based management globally. This comes from a conflict of interests which arises when the management of stocks for long-term sustainability competes with income generation in the short-term. Advances in SCP approaches and software, and improved data availability have contributed to the successful integration of socio-economic considerations into conservation planning processes whereby reserve areas are selected to achieve the desired ecological objectives using explicit conservation targets while having the least possible impact on the socio-economic environment. SCP is therefore a key tool which can be used as a starting point for the integration of ecological and socio-economic requirements thereby contributing to the development of management interventions necessary for improved governance in marine ecosystems.

7.4.1 Identification of priority areas for conservation and reserve design considerations

The first two objectives of this study were i) to identify priority areas for conservation investment in Algoa Bay, and ii) to evaluate the spatial compatibility of the existing marine reserves with the priority areas identified. Based on the selection frequency of PUs several key sites for conservation investment were identified within Algoa Bay. However, at low BLM values the majority of PUs required to meet the conservation targets had low selection frequencies and were widely dispersed across the study area leading to highly fragmented reserve designs. Reserve designs at low BLM values are therefore not feasible as they are impractical to manage and are of limited ecological value. The high level of flexibility in the selection of PUs is due to the wide spatial scale over which many of the conservation features were distributed in combination with low targets values for some features. This allows for numerous selection permutations for meeting conservation targets leading to low selection frequency for all but the most important PUs which represent features of limited spatial extent. Reduced fragmentation can be achieved through the imposition of a BLM parameter which aims to reduce the boundary length of the resulting design thereby increasing the level of compactness (Ardron *et al.* 2008), or by incorporating spatially explicit cost data into the analysis so that the least costly PUs are preferentially selected (Ban 2008; Ban *et al.* 2009).

Increasing the BLM parameter produced increasingly compact reserve designs which are more likely to support viable populations of marine species in the long-term and allow more effective enforcement. Furthermore, compact reserve designs allow for identification of discrete areas in which conservation investment can be directed as well as the demarcation of discrete areas in which reserves can be implemented and enforced more easily on the ground (Götz *et al.* 2009a). The outputs from this study indicate that the spatial resolution of each feature, the number of conservation features identified, and targets selected for each feature were sufficient for Marxan to identify priority areas for the protection of marine biodiversity in Algoa Bay. Furthermore the addition of opportunity costs reduced the overall socio-economic impact of the reserve design process.

The existing reserves within Algoa Bay overlapped well with the priority areas identified, indicating that they are well sited for the protection of biodiversity. However, due to the small area of the existing reserves they are inadequate to meet the conservation objectives set for Algoa Bay and an expansion of the reserve network is required. Areas identified for the expansion of the reserve network also overlapped well with the priority areas identified, indicating the compatibility of the existing reserves with future conservation objectives for Algoa Bay. A large proportion of the priority areas identified also occurred within the AENP MPA boundary recommended by specialists, indicating that local scientific knowledge may be sufficient to aid reserve design but lacks the quantitative data required to justify decisions regarding boundary design to fishery sectors which will be affected in the long-term. Expanding and building on the existing reserve boundaries and development of no-take zones within the proposed footprint is therefore a feasible option for advancement towards meeting the conservation objectives for Algoa Bay.

7.4.2 Incorporation of socio-economic data into reserve design

The third and most important objective of this chapter was to design MPA network options in Algoa Bay which met specific conservation targets for the protection of marine biodiversity while minimising negative socio-economic impacts associated with area closures on fishery activities. This is the foundation for adoption of an ecosystem-based management approach which calls for the concomitant consideration of ecological and socio-economic requirements and is advocated as the way forward in the future management of marine fisheries (FAO 2003; Garcia and Cochrane 2005; FAO 2005). Inclusion of socio-economic data into SCP aids in the efficient design of reserve networks which protect representative levels of biodiversity at least opportunity cost to society (Naidoo and Adamowicz 2006). In instances where there is a high degree of flexibility in potential reserve design, spatially explicit socio-economic data can improve the design of MPA networks as opposed to using biophysical data in isolation (Ban *et al.* 2009). Despite regulatory departments often having fishery information which can be developed into a cost layer for SCP, the spatial resolution or accuracy of the data are often poor and is therefore a major constraint to conducting effective SCP in marine ecosystems (Ban *et al.* 2009; Giakoumi *et al.* 2011). In order to overcome these problems in the current study, gaps in the available information and sources of data were identified (Chapter 2). Random stratified surveys were designed to develop an opportunity cost spatial layer for the recreational sectors (Chapter 5) for which no high resolution spatial data existed, and spatial data from fishery independent sources were obtained for commercial sectors for the development of a commercial spatial opportunity cost layer (Chapter 6). These layers and various combinations of each were used to aid in the design of a reserve network in Algoa Bay and resultant potential losses to fisheries were evaluated to determine optimal reserve design solutions.

The incorporation of spatially explicit costs led to the selection of PUs with lower cost within similar areas and the spatial patterns of areas selected did not differ greatly. Although similar spatial patterns were maintained, the overall impact to all fisheries was greatly reduced while still achieving all the conservation objectives. These findings support those of previous studies which have documented the important role that spatial socio-economic data plays in minimising the impacts to resource users when designing marine reserve networks using SCP (Stewart and Possingham 2005; Richardson *et al.* 2006; Naidoo *et al.* 2006; Klein *et al.* 2008b; Ban *et al.* 2009; Ban and Klein 2009; Klein *et al.* 2010; Weeks *et al.* 2010; Giakoumi *et al.* 2011). The primary goal of SCP is the protection of the biophysical environment. The conservation targets are therefore always achieved even if some conservation features are only present in PUs which have a high opportunity cost associated with them. However, when a feature is present in alternative PUs, those which have the lowest opportunity cost will be preferentially selected prior to the incorporation of more costly PUs until all conservation targets are achieved. In this manner the potential conflicts between ecological requirements and societal expectations are reduced through minimising the spatial overlap between area closures and fisheries activities (Ban and Klein 2009). However, in doing so the conservation objectives are not compromised and the resulting reserve design will be effective for conserving biophysical features provided realistic targets based on sound ecological principles and best available scientific information are used (Tear *et al.* 2005).

The combined recreational and commercial opportunity cost layer (ITI) produced the reserve design which resulted in least overall displacement of effort to fisheries, despite requiring a greater number of PUs than designs based on the recreational or commercial indices alone. The spatial selection of priority areas for protection, however, showed strong overlap for all spatially explicit socio-economic cost layers considered, but differed slightly from those in which the area of PUs was used as the index of cost. The homogenous (cost=area) cost layer resulted in numerous PUs being selected around Cape Recife; however, when socio-economic costs were included in the analyses far fewer PUs were selected in this region due to the high opportunity cost associated with this area. Conservation targets were therefore achieved through the selection of PUs which adequately represented the conservation features but had a lower opportunity cost from elsewhere in the planning domain. Using spatially explicit cost data did not significantly increase the amount of area which was required; however, the compactness of the reserve designs decreased slightly. The resulting designs were, however, sufficiently clustered to allow for the identification of key areas which would support practical implementation of conservation action.

The importance of including socio-economic data for activities which are likely to be impacted upon in the proclamation of new protected areas is highlighted through this and other studies (Klein *et al.* 2008a; Ban *et al.* 2009; Ban and Klein 2009). The current study only considered fisheries activities in the compilation of opportunity cost layers for SCP. This was due to the proposed AENP MPA only including two categories for zonation, "controlled" and "no-take" (See Appendix 1 for regulations). Within the controlled areas all current activities will be permitted but strictly regulated. Extractive resource use will be prohibited within the no-take zones, while non-consumptive activities will remain unaffected. Although non-consumptive tourism activities contribute to the local economy and can be of major importance to the generation of revenue in MPAs (Merino *et al.* 2009), the spatial distribution of these activities was not considered in this study as they are unlikely to be affected through the overall MPA design process at this stage. Nonetheless understanding spatial and temporal patterns of non-consumptive recreational activities is essential for sustainable management (Smallwood *et al.* 2011) and can facilitate zoning to reduce user conflicts and habitat degradation resulting from user numbers exceeding the ecological carrying capacity. Future research should therefore take non-consumptive activities into consideration in the development of more refined zonation plans within the MPA footprint to minimise spatial conflict between competing user groups. This will facilitate the establishment of core nodes in which similar tourism activities can be promoted in a sustainable manner through the provision of additional services.

Although numerous conservation planning assessments have been conducted in marine ecosystems, few have resulted in the successful implementation of conservation action. The establishment of new MPAs is often met with resistance from fishermen due to the potential loss of access to traditional fishing grounds which often delays or halts the implementation process. The support of public stakeholders is therefore critical if meaningful conservation actions are to implemented successfully to achieve the desired ecological benefits (Samoilys *et al.* 2007; Weeks *et al.* 2010). Stakeholders should therefore be included in the planning process from an early stage, and may provide valuable input into the design process. A lack of support from local communities and fishermen leads to poor compliance which not only threatens the ecological and fishery benefits of establishing protected areas (Samoilys *et al.* 2007), but also requires higher levels of enforcement and monitoring, increasing operational and management costs (Kelleher 1999). Inclusion of socio-political aspects into the design of MPAs and involving local stakeholders in management can improve compliance with regulations (Kelleher 1999; White and Vogt 2000; Walmsley and White 2003). This can be facilitated through the engagement and inclusion of stakeholders in the SCP design and planning process. Public consultation for the AENP was initiated prior to the commencement of this project in order to present the proposed AENP MPA design, which was based on expert judgement, and to obtain stakeholder support. However, the process was halted after the first round of engagement as there was widespread opposition to the proposed design as Park staff could not adequately justify the reasons for the proposed boundaries, and had limited knowledge on the extent of marine based fisheries activities in the area and the level to which they would be impacted. Numerous recreational and commercial anglers were engaged during the data collection phase of this project and informed of the process to follow. The completion of the spatial assessment presented in this chapter now presents the Park officials with the required data to re-initiate the public engagement process and present different reserve design options which are based on quantifiable data and from which impacts to fisheries activities can be assessed. This will aid in fostering stakeholder support and contribute to the development of an acceptable compromise for reserve design.

7.4.3 Importance of spatially explicit cost layers

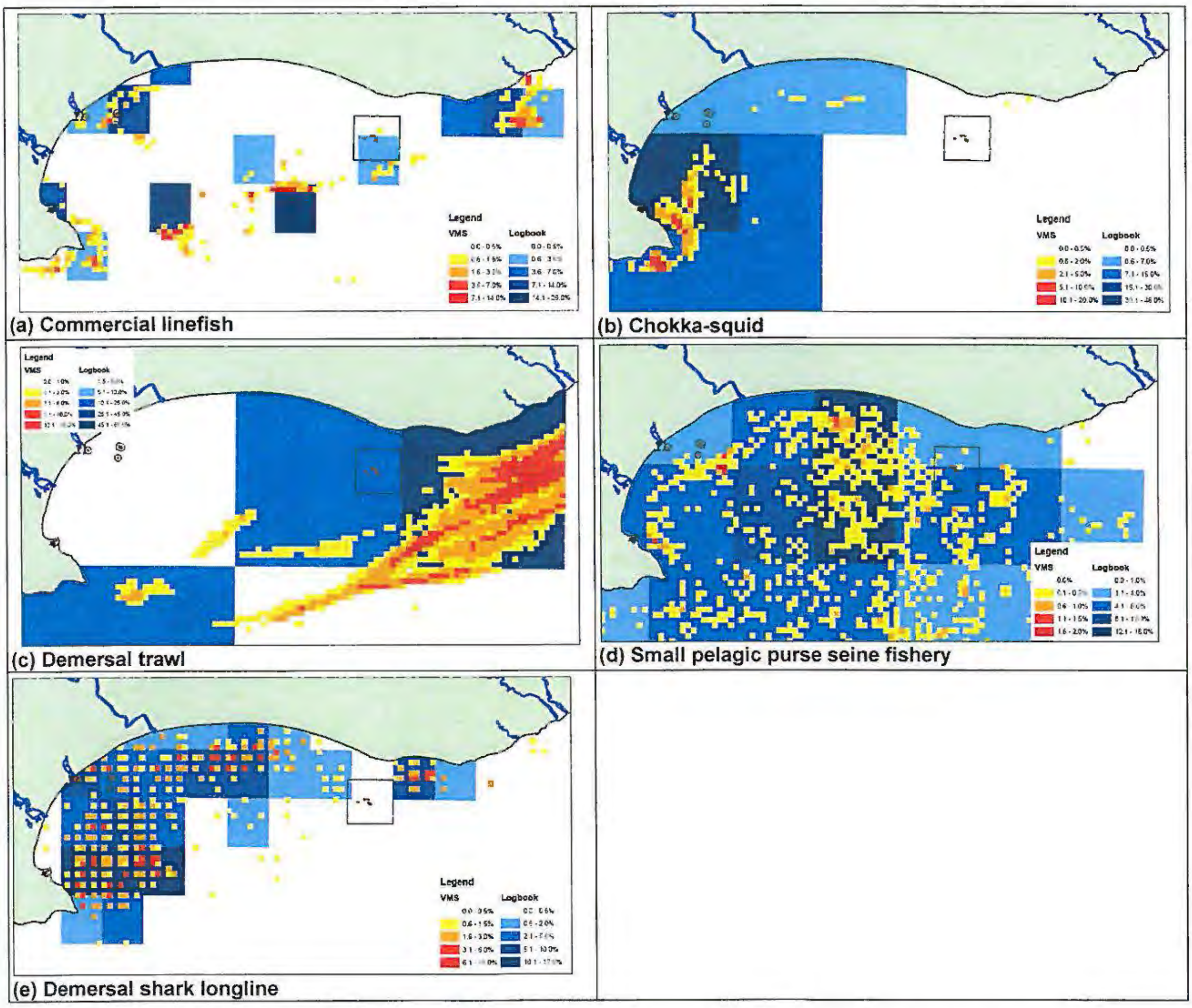
This study made use of the best available spatial data for incorporation of socio-economic considerations into the planning process. For recreational fisheries this involved intensive on-site interview techniques and validation using aerial surveys. These methods required considerable commitments of manpower and financial resources in order to obtain the required data which may not be possible for many projects. However, the importance of undertaking such surveys is highlighted through the spatial disparity in the distribution of recreational fishing effort, catch compositions and catch rates (Chapter 5) which all have important implications for MPA design. The use of spatial indices of recreational fishing effort in SCP contributed to reducing the impacts of reserve design on both the shore and skiboat fisheries, highlighting the importance and value of acquiring and using recreational data for future systematic planning studies.

The spatial distribution of commercial fisheries effort was assessed using VMS and onboard observer data, which were used to develop a spatial index of relative fishing effort. This data are the best possible spatial data available for the assessment of commercial fishing effort, being independent of reporting bias. However, it is not readily available to researchers and planners due to the confidentially of fishing locations utilised by rights holders. Furthermore, analysis of the VMS data requires a considerable investment of time to clean and compile data into a format in which it can be used for SCP. In contrast logbook data are fairly easily accessible for most commercial fisheries in South Africa but is recorded on a courser spatial scale according to 5', 10' or 20' grids depending on the fishery (Figure 7.8), and may be influenced by deliberate misreporting. The use of VMS/observer data improves the spatial accuracy of data but may be biased by underreporting as a result of malfunctioning units or in instances where the units are intentionally switched off. Nonetheless the use of VMS units is a legal requirement for commercial fishing vessels and improved compliance in the use of VMS can be achieved by regular monitoring by regulatory authorities. VMS data should therefore be an important component for any future SCP exercise in marine ecosystems due to the high spatial accuracy of the data from which fishing patterns can be inferred. Due to the improved spatial resolution of the data it is more preferable than traditional logbook data for evaluating the effects of reserve proclamation and expansion and can contribute to increased stakeholder support through the minimisation of spatially explicit socio-economic costs.

7.4.4 Evaluation of proposed AENP MPA design options

The proposed spatial design of the AENP MPA footprint was based on input from coastal specialists during a strategic environmental assessment process (Newman and Klages 2001) and no quantitative data were used to determine the optimal boundary location for the conservation of marine biodiversity in Algoa Bay. This study is the first quantitative assessment to evaluate the contribution of the proposed footprint to the conservation objectives of the broader Algoa Bay area, and the region as a whole. The proposed footprint is well sited with 29 of the 36 features represented at sufficient levels to meet the conservation objectives. Only two features were not represented at all, very deep habitats greater than 100m in depth and subtidal muddy substratum. The lack of representation of deepwater habitats and muddy substratum is not seen as a major constraint to the footprint design as the main goal of the MPA is to protect coastal inshore biodiversity and a separate planning initiative is currently underway to design an offshore MPA network which will protect deepwater habitats and the associated biodiversity (Sink and Attwood 2008).

A further five features were not well represented and could not meet the desired target levels within the footprint. Three of these features were intertidal (sand above rock) and subtidal (muddy-sand and sandy-mud) substrate types for which a target value of 25% was set. However, only between 13 and 16% of the features were present within the footprint. The majority (56%) of the intertidal sand above rock conservation feature is located at Cape Recife, well outside the footprint area, while scattered patches occur along the coastline in the eastern region of the study area. An eastern expansion of the footprint by approximately 3km would be required to meet the conservation target for this feature.



(a) Commercial linefish
(b) Chokka-squid
(c) Demersal trawl
(d) Small pelagic purse seine fishery
(e) Demersal shark longline

Figure 7.9. Comparison of the spatial resolution of data between VMS and commercial logbook sources. (a) commercial linefish, (b) chokka-squid, (c) Demersal trawl, (d) small pelagic purse seine fishery, and (e) demersal shark longline.

Subtidal sandy-mud and muddy-sand substrate is distributed predominantly at depths below 100m in Algoa Bay extending into the shallower waters in the eastern region of the study area and with few isolated patches of muddy-sand in the western region. Substrate type plays an important role in the distribution and abundances of demersal species targeted by the trawl sector in South Africa (Le Clus *et al.* 1994; Le Clus *et al.* 1996; Fairweather 2001; Sampson 2002) and inclusion of representative areas of different habitat types is therefore required for the effective protection of the associated species. In order to meet the conservation targets for these features a seaward expansion on the eastern end of the proposed footprint would be required. This would contribute to the protection of demersal ichthyofaunal communities occurring on these substrate types. Such an expansion would, however, led to a considerable increase in the level of impact on the demersal trawl fishery as this is the main area which it utilises in Algoa Bay (Chapter 6; Figure 7.9). An important consideration to be taken into account is the spatial accuracy of the subtidal substrate data layer which is based on extrapolation from samples collected thirty years previously. The lower confidence in this data source may outweigh the motivation for a seaward expansion until additional studies have confirmed the distribution of substrate types and the ecological value as it serves as a proxy for biological importance.

Only 31% of the two-kilometer buffer area around estuary mouth was present within the footprint, less than half the target value. Estuary mouths play a critical role in the connectivity between freshwater, estuarine and marine environments allowing the recruitment of juveniles into estuaries and the transient movement of sub-adults and adults of several marine species. Juvenile marine fish use estuaries as nursery areas and they are important feeding areas for sub-adult and adult fish (Baird *et al.* 1996; Whitfield 1998). Due to the importance of estuaries and the periodic movement of fish between marine and estuarine waters, the density of many species is often greater in close proximity to estuary mouths. It is also in these areas that they are heavily targeted by anglers (Chapter 5) and therefore in need of additional protection over and above species specific regulations. Only one estuary mouth, the Sundays Estuary mouth, is present within the proposed footprint. Protection of this estuary mouth through the proclamation of the AENP MPA is therefore regarded as a high priority. Additional protection measures for the nearshore waters around the permanently open Swartkops and Bushmans estuaries should be considered to enhance the level of protection in these ecologically important areas.

Target levels for abalone habitat were also not achieved within the footprint with only 60% of the available habitat within Algoa Bay represented. The conservation target for abalone habitat was set high (75%) due to the poor status of the stocks and its vulnerability to overexploitation as a result of its sedentary nature, slow growth rate and limited distribution (<15m depth). Furthermore, a large illegal abalone fishery is active within Algoa Bay with little deterrence due to the poor capacity for enforcement (Raemaekers and Britz 2009; Raemaekers 2009). Protecting a large proportion of the abalone habitat was therefore seen as a key objective for the establishment of the AENP MPA as it would allow for improved enforcement within no-take zones. However, despite the declaration of the Bird Island MPA in 2004 (DEAT 2004) poaching activities continued to increase within the no-take MPA due to a continued lack of enforcement capacity (Raemaekers and Britz 2009; Raemaekers 2009). It is only recently that a dedicated enforcement team and vessel have become established within the Bird Island MPA, resulting in a reduction in poaching activity (R.Fox *pers. comm.*; A.Padayachee *pers. comm.*). In order to increase the representivity of abalone habitat additional areas at Cape Recife would need to be incorporated into the footprint design. This would create practical management difficulties due to the distance apart and would likely require additional capacity for enforcement and monitoring. The Woody Cape headland may contribute to the protection of abalone but was not included as an abalone area in this analysis due to a lack of supporting data indicating the presence of abalone on these reefs. Unlike the Cape Recife, Bird Island and Cape Padrone areas where abalone research surveys have been conducted (SFRI 1986; Tarr and Anderson 1987; Godfrey 2003) and poaching activity has been observed (R.Fox *pers. comm.*) confirming the presence of abalone in these areas, no such evidence exists for the Woody Cape area. A dedicated survey would be required to establish whether this area contributes to the protection of abalone stocks.

In order to meet the conservation targets (revised where necessary for the footprint) within the footprint 70% of the area would be required to be designated as no-take. While this represents a large proportion of the footprint area, the impact on fishing effort for both recreational and commercial sectors would be a reduction of less than 20% of the current levels in Algoa Bay. This assumes that current fishing within the proposed no-take zones would be eliminated; however, a displacement of fishing effort is likely to occur rather than an overall reduction (Charles and Wilson 2009) and the economic impact is therefore likely to be less than anticipated. In addition benefits from long-term spillover and larval export from MPAs may enhance local fisheries (Hastings and Botsford 1999; Halpern *et al.* 2004) offsetting the costs of displacement. Furthermore, the proposed footprint does not incorporate areas immediately adjacent to the main launch sites used by recreational or commercial vessels (Port Elizabeth harbour, Boknes and Kenton), and with the exception of the Sundays Estuary mouth, only includes coastal areas which are less accessible to shore anglers. Travel times and travel distances to the most easily accessible sites would therefore not be affected and would not result in increased financial cost to the anglers in order to reach their fishing destinations. However, the fishing quality at sites closer to access points may be poorer than at sites further afield. The displacement of fishing effort may also lead to further aggregation of anglers in the remaining sites and result in localised stock depletion (Charles and Wilson 2009), particularly when slow growing resident reef species are targeted. Local depletion may, however, be offset by the dispersal of larvae and spillover of juveniles and adults from protected no-take zones into adjacent fishing grounds (Roberts *et al.* 2001; Halpern *et al.* 2004; Ashworth and Ormond 2005; Francini-Filho and Moura 2008; Stobart *et al.* 2009; Cudney-Bueno *et al.* 2009). The spatial and temporal scales over which this may occur will depend on individual species' mobility and growth rates, connectivity between reef systems and the mechanisms of larval dispersal. Such benefits may outweigh the loss of fishing grounds in the long-term; however, short-term effects may not be readily evident to fishermen. Long-term monitoring and evaluation of changes in the ecological state and socio-economic pressures on the resources is required in order to assess management efforts and adapt protocols where required in order to achieve the desired conservation and management objectives.

The proposed no-take areas for the AENP MPA comprises considerably less area than is required to effectively achieve the conservation targets for Algoa Bay with only 58% of the conservation feature targets achieved. Although the no-take areas proposed require 25% less area than the footprint design which optimally represents features within Algoa Bay, they would result in an 11% impact to fisheries, only a 5% reduction from the best design of no-take areas within the footprint. The overall goal of the MPA is to conserve a representative mosaic of habitats and prevent overexploitation of marine resources (SANParks 2008). It is arguable whether the level of protection afforded by the proposed no-take zones will be effective in achieving this and expansion of the proposed no-take zones should be considered to improve the conservation value of the MPA. Designs to improve the conservation value of the AENP MPA were investigated using the proposed no-take areas as the basis for expansion. The resulting designs indicated that between 77 and 85% more PUs would be required than currently included in the no-take zones with the establishment of an additional offshore no-take zone to meet the deeper water conservation targets. The associated impacts to the fisheries as a result of such an expansion of the no-take zones to meet conservation targets would, however, increase to between 17 and 23%. This is higher than the impact associated with the optimal design arising from Scenario 5 where the overall displacement of effort would be 14%.

7.4.5 Contribution of the proposed AENP MPA to regional conservation

Over the past three decades the importance and value of MPAs for both the conservation of biodiversity and as a managerial tool for fisheries management has been increasingly recognised (Agardy 1994; Bohnsack 1998; Lauck *et al.* 1998; Gell and Roberts 2003b; Hilborn *et al.* 2004; Jones *et al.* 2007). South Africa is fortunate to have a long history of MPAs (first declaration in 1964) which now represent large sections of the coastline and nearshore environment. However, historically many of these were sited on an *ad hoc* basis with little systematic planning over a broader geographical scale (Hockey and Branch 1997). This has resulted in poor representivity of many habitat types and species within the existing MPAs in the Agulhas Bioregion and a need for expansion was identified (Clark and Lombard 2007). Expanding and building on the existing MPA network therefore provides a good opportunity for improvement, allowing for the identification of priority areas which would contribute most to enhancing the representation of poorly protected habitats and species.

There are currently 13 MPAs or closed areas which afford varying levels of protection to marine biodiversity and ecosystem processes within the Agulhas Bioregion (Table 7.9). The proposed AENP MPA would contribute considerably to the MPA network as it would be the largest MPA in the bioregion being approximately 4 times larger than any other MPA (take and no-take), while the proposed no-take portion would be approximately twice as large as any other single no-take zone/MPA in the bioregion. Furthermore the proposed AENP would increase the current overall protection of marine habitats within MPAs in the Agulhas Bioregion by approximately 80%, and no-take zones by approximately 70%. The proposed design would therefore account for 47% and 46% of MPA and no-take zones within the bioregion respectively (Table 7.9). Due to its large size, the proclamation of the AENP MPA would make a significant contribution to regional conservation. Furthermore large areas of soft-sediment bay habitats, which are important habitats for several species and which are currently poorly represented within the region, would be included in the MPA network. In addition the proposed AENP footprint is wider, extending further offshore than any existing coastal MPA thereby increasing representation of shelf habitats further offshore (up to 20km in the widest section) than any other no-take MPA where the maximum distance from the shoreline is three nautical miles (approximately 5km). The appraisal in this chapter therefore does not aim to highlight shortcomings of the proposed design as any addition to the existing network will contribute to regional conservation, but rather to highlight the benefits of using quantitative systematic planning approaches to facilitate decision-making through the identification of ways in which the proposed design can be improved to optimise the conservation benefits locally as well as within the bioregion. Two key aspects were taken into account when assessing options for improving the proposed design, improving regional representivity of poorly protected marine coastal and nearshore habitats, and reducing potential impact to consumptive resource uses thereby minimising conflicts as far as is practically possible.

Table 7.9. Contribution of MPAs to conservation in the Agulhas Bioregion.

MPA	Total area (ha)	% of total MPA area	% of total MPA no-take area	% of total MPA take area
Table Mountain National Park (including 4 no-take MPAs) *	27 992	11	1	23
Helderberg MPA	242	<1	<1	0
Betty's Bay MPA	2 029	1	0	2
De Hoop MPA	29 003	12	23	0
Goukamma MPA	3 400	1	0	3
Robberg MPA	2 620	1	0	2
Tsitsikamma National Park MPA	26 446	11	21	0
Sardinia Bay MPA	1 291	1	1	0
AENP MPA (including Bird Island MPA)	117 157	47	46	49
East London reserves (3 closed areas)	26 124	11	0	22
Dwesa-Cwebe MPA *	11 676	5	9	0
TOTAL	247 979			

* only taking into account the area of the MPAs considered to be within the Agulhas Bioregion as these MPAs overlap with adjacent bioregions

Despite approximately 19% of the coastline within the Agulhas Bioregion being designated as MPAs, only four out of ten shoreline features are currently represented sufficiently to meet the national target (20%) for MPAs (take and no-take). Furthermore, only one shoreline feature is represented sufficiently to meet the no-take target of 15%. Algoa Bay is well situated to contribute to improving regional coastal conservation as four additional shoreline features are represented in sufficient quantities to attain regional targets for MPAs. This will contribute to ensuring suitable representation of eight of the ten shoreline habitat types defined in the Agulhas Bioregion. The proposed AENP footprint incorporates sufficient estuarine shoreline to attain on additional MPA target but is unable to meet additional targets. Revision of the footprint boundaries to incorporate poorly represented shoreline habitats could therefore improve the overall conservation value of the AENP considerably.

In terms of the inshore habitats, only two of the four features are present within Algoa Bay, one of which is already adequately represented within the bioregion. The proposed AENP footprint will, however, contribute to meeting the conservation target for the second feature (sandy grounds) thereby ensuring three of the four regional inshore habitat targets are achieved. Furthermore, both Algoa Bay and the footprint contribute significantly to the representation of islands in the bioregion increasing the feature target from 48% in MPAs to 312% and 233% for Algoa Bay and the footprint respectively. This is due to the limited number of islands along the south-east coast of South Africa and reinforces the regional conservation importance of Algoa Bay and the proposed AENP MPA.

None of the MPA targets are achieved for the 20 inner shelf, outer shelf, shelf edge and pelagic features in the existing MPA network within the bioregion, with 14 of the features not represented at all. Algoa Bay would make a considerable contribution (>10%) towards achieving the desired target levels for eight of the 20 features, again indicating its regional importance. As many of these features are situated further offshore outside of the proposed AENP footprint, only four of the 20 feature targets are present and limited additional protection would therefore be achieved through the proclamation of the AENP MPA. An Offshore Marine Protected Area Project has been initiated in order to address the requirements for deepwater conservation areas in South Africa (Sink and Attwood 2008).

With respect to the proposed AENP no-take zones, representation of four features was improved considerably (>10%) and one additional regional feature target (islands) was attained for no-take MPAs. In comparison the design outputs for no-take zones from Scenario 5 and 6 made considerable (>10%) contributions to nine feature targets; however, no additional regional targets were met. Nonetheless the main objective of the AENP MPA is to enhance protection of coastal and inshore marine habitats and create a continuum of protected habitats between mountainous inland regions and the inshore marine ecosystem. The proposed location in Algoa Bay is therefore suitable as numerous poorly represented features are present and increased protection would make significant contributions to regional conservation efforts. Although the current design of the proposed AENP makes a considerable contribution to regional conservation efforts, refinements in the footprint and no-take boundary design would lead to considerable improvement in the overall contribution to regional and national conservation objectives.

The level of protection afforded to estuary surrounds regionally and nationally is poor and there is an urgent need to improve the protection level for heavily targeted linefish species which utilise these areas. Estuaries are not only important ecological areas for several species, but are also subject to the high levels of habitat degradation as a result of land use changes, poor catchment management and changes in hydrological characteristics through excessive freshwater abstraction (Whitfield and Cowley 2010). Increasing the representation of estuaries and their surrounding nearshore waters in no-take MPAs has been slow due to conflicting societal and ecological requirements. Two estuaries within Algoa Bay have already been changed irreversibly through industrial development and are no longer ecologically functional. The remaining estuaries in Algoa Bay are threatened through continued development pressure and high levels of extractive resource utilisation (Cowley *et al.* 2009). Despite the recognition of the threats to these systems, the proposed AENP MPA footprint only includes one of the three functional estuaries within Algoa Bay. Although they will benefit from improved coastal management and enforcement within the MPA footprint, continued high levels of extractive resource use at the estuary mouths will continue to place high levels of pressure on several estuarine dependent species which are considered overexploited. Although a seasonal closure at the Sundays

Estuary mouth has been suggested this was not included in this assessment as the conservation benefits for targeted species will be dependent on the duration and timing of closure which have not yet been agreed upon. Full protection of the Sundays Estuarine mouth will contribute to an 11% increase in attaining regional no-take targets and would contribute significantly to the management of inshore estuarine dependent linefish stocks in Algoa Bay.

An important aspect of MPA networks which cover large geographical areas is to take into account the influence of environmental gradients on biological communities. Although the Agulhas Bioregion has been defined as a warm temperate region which has similar biological communities on a national scale, changes in biological communities are evident within the bioregion. Comparison of linefish species composition across MPAs on the east coast of South Africa from controlled angling surveys indicates a change in species dominance from a roman dominated community along the south coast (Goukamma MPA and Tsitsikamma National Park), to a santer dominated community along the south east-coast (Bird Island MPA and Algoa Bay) and a slinger dominated community along the Pondoland coastline (Table 7.10). Although superseded by numerically dominant smaller species, a similar pattern in the important linefish species is observed using the results of UVC (Table 7.11), with hottentot and roman being the most abundant linefish species in the Cape (Table Mountain National Park), changing to roman along the south coast (Goukamma MPA and Tsitsikamma National Park), santer along the south-east coast (Bird Island MPA) and slinger along the Pondoland coastline. The longitudinal change in community structure along the south-east coast of South Africa highlights the importance of protecting representative areas of similar habitat over large geographical areas due to the changing species composition along environmental gradients. In addition the incorporation of habitats, such as coastal embayments, which are important nursery and aggregation areas for several marine species is of great importance, particularly as these areas are usually subject to greatest levels of anthropogenic influence. This study has illustrated the importance of sheltered embayments and reefs to species, as well as juveniles of numerous other species, which are not numerically abundant over reef complexes within the bioregion, and are therefore not well represented within the existing MPA network. Future expansion to include embayments and species and life-history stages utilising these habitats would enhance the overall conservation value of the MPA network. This would make significant contributions to the protection of species such as santer and silver kob, which are a commercially and recreationally important species and heavily targeted, yet poorly represented in the existing reserve network.

Connectivity between MPAs and the design of MPA networks which are capable of preserving ecosystem functionality over larger spatial scales has become increasingly important (Roberts *et al.* 2003; Jones *et al.* 2007; Botsford *et al.* 2009; Kaplan *et al.* 2009; Planes *et al.* 2009). The establishment of networks with MPAs which complement each other is seen as a critical step forward for the implementation of ecosystem based management in the marine environment. Networks allow for increased protection of biodiversity over large spatial scales, and if well designed, for the connectivity between breeding populations of resident species through larval dispersal. Larval dispersal, however, is complex, with interactions between oceanic currents and the swimming ability and behaviour of different species is not well understood (Pattrick 2007). Limited knowledge of spawning and larval behaviour for most of our linefish species is therefore a major limitation in assessing the degree of connectivity between our existing MPAs, and therefore the long-term persistence of populations exposed to high levels of exploitation. The adequacy and level of protection afforded to heavily targeted species through the existing MPAs network may not be sufficient. Santer, for example, dominate reef fish communities in Algoa Bay and are heavily targeted by commercial and recreational linefisheries. However, they comprise an insignificant portions of the reef fish communities in the Goukamma (3%)(Götz *et al.* 2009b), Tsitsikamma (<1%) (Bernard unpublished data) and Pondoland (1%) (Mann 2010) protected areas. Continued high levels of exploitation in Algoa Bay may therefore lead to local depletion of the stocks. Although the Bird Island MPA plays a vital role in the protection of adult spawning stock for this species, the small size of the MPA may be insufficient to reseed adjacent exploited areas through the export of larvae and juveniles. Increasing the extent of area protected within the Algoa Bay area would enhance the protection of this and other important linefish species and allows for the incorporation of potential nursery areas indentified in this study.

7.4.6 Conclusions

This study has demonstrated the conservation importance of Algoa Bay within the Agulhas Bioregion. The proposed AENP MPA footprint and no-take zones are well sited due to the representation of numerous biophysical features which are inadequately represented within the regional network of MPAs. The proclamation of the proposed AENP would therefore enhance the conservation value of the existing MPA network and long-term resilience of marine communities in the bioregion.

The systematic planning approach used in this study was successful in integrating spatially heterogeneous biophysical data and identifying key areas of conservation importance worthy of protection. Furthermore, through the incorporation of spatially explicit fisheries data, the planning approach was able to integrate both biophysical and socio-economic aspects into the selection of priority areas for conservation. This reduced the overall impact to fisheries activities considerably, yet still achieved all conservation targets. The application of differing fisheries cost layers resulted in the selection of similar areas for conservation investment but differed slightly based on the type of weighting used to integrate the different fishery sector cost layers. In doing so, different design alternatives were identified which resulted in differing levels of impact to each fishery sector. These alternatives provide a useful means to support decision-making by regulatory authorities and provide the necessary data to re-initiate the stakeholder engagement process.

By adjusting the BLM parameter in Marxan, feasible design options were developed, which resulted in the selection of sufficiently large areas to warrant practical implementation of no-take zones. However, these design outputs only provide a guideline to assist authorities and stakeholders in reaching agreement and making informed decisions regarding the final design of no-take zones within Algoa Bay. This study has shown that although there was strong overlap with the results of the SCP analyses and the MPA boundary, which was designed based on expert opinion, the conservation value of the AENP MPA can be improved through adjustments to the boundaries currently proposed. These adjustments will improve the conservation value of the no-take zones considerably but only result in slightly higher impacts to fisheries than the design currently proposed.

This study has shown that SCP can be effectively conducted in situations where spatial data have previously been limited and that it is a useful decision support tool which can provide management authorities and stakeholders with quantitative information to facilitate the engagement process and decision-making. The final design of the proposed AENP MPA footprint and no-take zones will need to take practical, logistical and financial aspects into consideration, as well as the views of stakeholder groups. This will contribute to the design of no-take zones which have greatest buy-in by stakeholders and which can be implemented successfully on the ground by the designated management authorities. The development of a monitoring framework is an integral component of adaptive management and is required to evaluate the success of implementation against pre-defined management objectives.

Table 7.10. Dominant offshore species cumulatively contributing up to 90% of the catch by number in MPAs and Algoa Bay within the Agulhas Bioregion.

Goukamma MPA Götz 2005; 2009		Tsitsikamma National Park Bernard 2010 unpublished		Bird Island MPA this study		Algoa Bay this study		Pondoland MPA Mann 2010 unpublished	
Number of species = 37		Number of species = 37		Number of species = 24		Number of species = 38		Number of species = 50	
Species	% composition	Species	% composition	Species	% composition	Species	% composition	Species	% composition
Roman	44	Roman	61	Santer	43	Santer	46	Slinger	38
Fransmadam	28	Dageraad	11	Roman	24	Fransmadam	14	Scotsman	17
Steentjie	8	Fransmadam	5	Fransmadam	22	Roman	9	Natal seacatfish	17
Blue hottentot	4	Smooth-hound	4	Steentjie	3	White seacatfish	7	Yellowbelly	8
Santer	3	Steentjie	4	Other	9	Silver Kob	6	Black musselcracker	5
Red tjor-tjor	2	Piggy	2			Steentjie	3	Catface	3
Dageraad	2	Red steenbras	1			Red tjor-tjor	3	Halfmoon	3
White seacatfish	2	Geelbek	1			Bluntnose spiny dogfish	2	Santer	1
Other	9	Other	10			Other	10	Other	8

Table 7.11. Dominant offshore species cumulatively contributing up to 90% of the observer fish during UVC in MPAs within the Agulhas Bioregion.

Table Mountain National Park Bernard 2010 unpublished		Goukamma MPA Götz 2005; 2009		Tsitsikamma National Park Bernard 2010 unpublished		Bird Island MPA this study		Pondoland MPA Mann *et al.* 2006	
Number of species = 22		Number of species = 30		Number of species = 31		Number of species = 44		Number of species = 121	
Species	% composition	Species	% composition	Species	% composition	Species	% composition	Species	% composition
Hottentot	22	Fransmadam	34	Twotone fingerfin	13	Fransmadam	70	Piggy	32
Roman	19	Steentjie	28	Roman	12	Steentjie	9	Slinger	9
Twotone fingerfin	14	Blue hottentot	12	Fransmadam	10	Strepie	7	Fransmadam	6
Redfingers	13	Roman	7	Blacktail	10	Blue hottentot	4	Blacktail	6
Fransmadam	8	Strepie	6	Blue hottentot	8	Santer	1	Bronze bream	5
Jutjaw	5	Twotone fingerfin	3	Barred fingerfin	8	Other	9	Striped grunter	4
Puffadder shyshark	3	Cape knifejaw	3	Janbruin	5			Zebra	3
Janbruin	3	Other	9	Red Fingers	4			Natal fingerfin	2
Steentjie	2			Steentjie	4			Blue emperor	2
Barred fingerfin	2			Cape knifejaw	4			Steentjie	2
Other	9			Red steenbras	3			Sea goldie	2
				Cape stumpnose	3			Blacksaddle goatfish	2
				Zebra	2			Other	25
				Dageraad	2				
				Other	10				

CHAPTER 8

DESIGN OF A FRAMEWORK FOR MONITORING AND EVALUATING THE IMPLEMENTATION OF AN ECOSYSTEM APPROACH TO FISHERIES MANAGEMENT IN ALGOA BAY

8.1 Introduction

The primary goal of this study was to obtain and analyse baseline data to understand spatial and temporal trends in the distribution and abundance of fish populations and fisheries activities in Algoa Bay. This data were required to inform spatial planning in Algoa Bay and develop a monitoring framework to evaluate the success of implementation through changes in biological and socio-economic parameters. In order to do so the key environmental drivers influencing productivity within Algoa Bay and gaps in knowledge were identified in Chapter 2; detailed spatial baseline information on the composition, relative abundance and size structure of targeted fish species was presented in Chapters 3 and 4; and fine scale spatial and temporal trends in fisheries activities were identified in Chapters 5 and 6. Marine spatial planning is one of the key tools which can be used aid the implementation of ecosystem based management (Crowder and Norse 2008; Douvere 2008; Gilliland and Laffoley 2008; Dalton *et al.* 2010; Ogden 2010) and a systematic conservation planning exercise was conducted in Chapter 7 to identify priority areas for conservation investment in Algoa Bay, and evaluate the socio-economic impacts thereof. In addition, the proposed footprint and no-take designs of AENP MPA were evaluated in terms of their contribution to local and regional conservation objectives and impacts to fisheries. This chapter aims to integrate the research outputs from previous chapters into an adaptive management framework through the development of a framework for monitoring implementation of management initiatives and evaluating the resultant long-term trends in environmental and socio-economic conditions which will allow for ongoing evaluation of the effectiveness of management strategies in achieving the overall conservation goals.

Monitoring changes in socio-economic parameters is a key aspect for ecosystem-based management yet identifying suitable indicators and developing monitoring protocols is a specialised task. The socio-economic monitoring component is therefore limited to assessing changes in the local resource use patterns with regards the key fisheries activities as well as changes in the harvest levels of each sector. Development of a detailed socio-economic monitoring protocol which determines the economic impacts of reserve establishment, the perceptions of stakeholders and their wellbeing was beyond the scope of this study and needs to be developed in collaboration with social scientists in the future.

Monitoring involves the repeated long-term measurement of a pre-determined set of parameters in order to quantitatively evaluate temporal changes (Vos *et al.* 2000). Programmes designed to monitor ecosystem based management approaches need to incorporate biophysical, social-economic and institutional parameters in order to evaluate the response of the environment and society to management interventions (Pajak 2000). The Pressure-State-Response (PSR) framework has been widely used for environmental monitoring and reporting. It is based on the evaluation of changes in key parameters representative of the pressures exerted by society (i.e. fisheries), the state of the resources (e.g. fish stocks), and the regulatory responses (e.g. no-take MPAs) which are implemented to minimise impacts with the aim of improving the health of the ecosystem (Pajak 2000). The cyclical nature of the PSR framework (Figure 8.1) allows for management to be informed through past experiences as new information becomes available allowing for continual improvement (Pomeroy *et al.* 2004; Nichols and Williams 2006; Day 2008). The adaptive management process is pivotal to the holistic management of complex ecosystems in which the interactions between environmental and societal pressures and the biophysical environment are not clearly understood. Indicators, reference points and performance measures are commonly used in the adaptive management process to assess and evaluate performance (Sainsbury and Sumaila 2003). Although such systems have been widely implemented in the management of terrestrial ecosystems, application to the marine environment has been far slower due to a general lack of guidance for marine monitoring. However, the PSR framework and derivatives are increasingly being used to evaluate the implementation and success of EBM approaches in the marine environment (Garcia and Staples 2000; Caddy 2004; Jennings 2005; Mangi *et al.* 2007; Ou and Liu 2010).

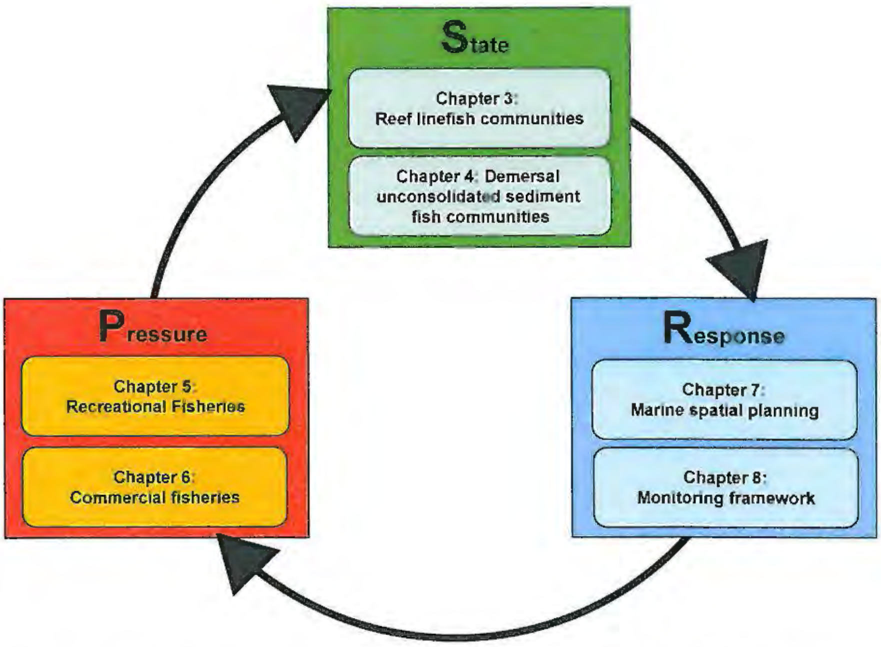


Figure 8.1. Thesis structure integrating baseline information on pressures and state with management responses using the PSR monitoring framework.

Monitoring frameworks require that high level management goals which outline what needs to be attained are translated into clear operational management objectives which have direct and practical significance (FAO 2003; Sainsbury and Sumaila 2003; Link 2005). Indicators, or a suite of indicators, are selected to represent each operational objective. They are quantifiable measures which serve as surrogates for parameters relevant to the management objectives which may be difficult to monitor (Noss 1990; Vos *et al.* 2000). Ecosystem indicators need to be scientifically defensible, practical and pragmatic, repeatable and cost effective to monitor, transparent and directly relevant to the management objectives (Jamieson *et al.* 2001). Where possible they should be based on readily available data sources; however, where this is not possible a limited number of effective indicators relative to management priorities should be identified and standardised methods for obtaining and evaluating the data developed (Garcia *et al.* 2000). Indicators are evaluated in terms of target reference points which define the "desired" conditions based on the management objectives. Limit reference points serve as triggers to initiate management actions when indicator values change from a desirable to an undesirable state and thereby aid in preventing irreversible damage (Sainsbury and Sumaila 2003; Hall and Mainprize 2004). The relationship between indicator and reference points indicates how well management objectives are being achieved and serves as the performance measure for evaluation (Sainsbury and Sumaila 2003). In the absence of clear management objectives or in instances where reference points cannot be easily defined, trajectories of change can be used to evaluate temporal trends in indicator condition (Jennings 2005; Jennings and Dulvy 2005).

The selection of a suite of PSR indicators which suitably represent the management objectives, and the definition of reference points against which they can be evaluated, is crucial for adaptive management in ecosystems where data are limited. Indicators of State have been used extensively for environmental monitoring, and Pressure and Response indicators are increasingly used for tracking progress and implementation of EBM (Pajak 2000). In order to be effective and allow interpretation of Responses, State indicators need to have a clear and causal relationship with Pressure indicators which will support and inform decision-making (Jennings 2005).

In order for EBM to be successful dedicated monitoring of selected PSR indicators is required to evaluate the outcomes of management and allow improvement on an ongoing basis. This chapter integrates baseline information from earlier chapters into a PSR monitoring framework (Figure 8.1) for evaluating the future implementation of spatial management initiatives in Algoa Bay, and outlines a protocol for data collection. Although all ecosystem components need to be included in the monitoring framework in the future, the focus of this research chapter was to develop a framework for the direct extractive fishery activities and targeted resources assessed in previous chapters in this thesis. The presented monitoring framework therefore takes into consideration the pressures exerted by the recreational and commercial fishery sectors identified in Algoa Bay and the state of the linefish resources, forming the basis for the development of a more holistic and integrated framework in the future. The key objectives of this chapter were:

1. to define key objectives for the monitoring framework;
2. to identify and discuss indicators to meet the monitoring objectives and define provisional reference points; and
3. to design and discuss a sampling and monitoring strategy with appropriate spatial and temporal replication

8.2 Development of the monitoring framework

A monitoring framework for evaluating implementation success of EBM requires the following four main steps which form the basis for the structure of this chapter:

- Step 1: Define long-term ecosystem related objectives;
- Step 2: Identify meaningful indicators and decide on reference points;
- Step 3: Develop a sampling programme and identify the analytical tools required for evaluation;
- Step 4: Delegate responsibilities for the collection of data and management of the programme (Vos *et al.* 2000; Curtin and Prellezo 2010).
- Step 5: Evaluate implementation of actions/recommendations from ongoing monitoring for adaptive management

8.2.1 Step 1: Define long-term ecosystem related objectives

In order to design an efficient monitoring programme the key management objectives for the marine and coastal environment in Algoa Bay need to be defined. Several authorities are involved in management of the coastal and marine resources within Algoa Bay, each with different roles and responsibilities. The Oceans and Coasts branch of the Department of Environmental Affairs (DEA) is the national environmental agency responsible for the management and conservation of coastal waters, while the Department of Agriculture, Forestry and Fisheries (DAFF) is responsible for regulation and management of fisheries activities. The Marine Living Resources Act (MLRA) (Act 18 of 1998) provides the legal background for management and regulation with the overall goal of conserving the marine ecosystem and achieving long-term sustainable utilisation of marine living resources in a fair and equitable manner (Republic of South Africa 1998). Specific management objectives outlined in the MLRA for MPAs include the protection of fauna and flora and the physical features on which they depend, to facilitate fishery management by protecting spawning stock, to facilitate stock recovery and contribute to stock enhancement, and finally to reduce user conflict (Republic of South Africa 1998).

Management of the Bird Island MPA and St Croix Reserves has been delegated to SANParks. The management objectives of the Bird Island MPA specify the protection of marine ecosystems and populations of threatened species, and the protection of reproductive capacity of commercially important species to allow for stock recovery (DEAT 2004), while the key objectives of the AENP MPA are to maintain a representative mosaic of habitats and to prevent overexploitation of targeted species (SANParks 2008).

Although covering different spatial and jurisdictional areas, the overarching high level management objectives for marine and coastal areas of Algoa Bay are broadly similar to, and echo those of EBM and can be summarised as follows:

1. To conserve and protect representative marine habitats and maintain ecosystem functionality
2. To allow equitable and sustainable use of marine resources
3. To enhance protection of threatened and overexploited species
4. To contribute to the recovery of depleted stocks

These management objectives can be translated into three broad categories for monitoring purposes, (i) to monitor trends in the state of the ecosystem, (ii) to monitor trends in the pressures which ultimately affect the state, and (iii) to monitor the implementation of the management responses. In order to understand and interpret the effects of management on changes in the state of the resources and differentiate between natural directional change or decadal cycles and fishery induced changes, the management interventions need to be monitored, thereby allowing for future improvement through adaptive management. These three categories (Pressure, State and Response) have been widely used for ecological monitoring in the past and are increasingly being used for ecosystem based management approaches for fisheries (Pajak 2000; Jennings 2005; Mangi *et al.* 2007).

The first monitoring objective would therefore be to monitor trends in the state, or the health of the ecosystem. In this case this would entail monitoring changes in the state of the living resources which are targeted by fisheries activities in Algoa Bay and would include the linefish (inclusive of species targeted by the demersal trawl fishery) (fishery independent assessments conducted in Chapters 3 and 4), pelagic fish and chokka-squid (no fisheries independent assessments conducted in this study) resources, as well as bycatch species from the different sectors. The second objective is to monitor trends in the pressures on the biophysical environment which may alter the health of the ecosystem. In this study this involves establishing protocols for evaluating changes in the pressures arising from extractive fisheries activities inclusive of commercial (Chapter 6), recreational and subsistence (Chapter 5) sectors. The final objective is to monitor the management actions which have been implemented in order to reduce the pressures on the ecosystem and contribute to improving ecosystem health. This is an essential component which allows for the evaluation of the effectiveness of the management actions in bringing about the desired changes in both pressure and state, and allows for management actions to be adapted and improved as required. Although Chapter 7 investigated alternative design options for the development of no-take MPAs for optimal conservation of marine biota in Algoa Bay, a public engagement process still needs to be undertaken and agreement reached and decisions made regarding the final design of no-take zones. As a result a flexible framework is proposed which can be easily adapted based on the final design decided upon for the AENP MPA and no-take zones.

No-take MPAs play a vital role in ecological monitoring as long-term natural temporal trends and directional changes in ecological parameters can be determined and distinguished from those caused by direct human pressures on the ecosystem (Roberts 1997; Sainsbury and Sumaila 2003). This contributes to understanding the effects of environmental drivers such as global climate change on natural communities in the absence of direct anthropogenic pressures. Furthermore as the targeted fisheries resources are exempt from direct extractive use within MPAs they are crucial for the development of reference points against which adjacent exploited areas can be evaluated and therefore serve as benchmarks for long-term comparison. The existing MPAs and proposed future no-take zones therefore play a pivotal role in design and implementation future ecological monitoring studies in Algoa Bay, contributing to evaluating stock recovery as a result of closures and understanding the long-term benefits of MPAs.

The identified objectives for the monitoring programme are broad and can be summarised as:

1. Monitor changes in the state of the linefish, pelagic fish and chokka-squid resources as well as bycatch species from the different sectors
2. Monitor changes in the pressures arising from extractive fisheries activities inclusive of commercial, recreational and subsistence sectors.
3. Monitor the management actions which have been implemented to improve and protect ecosystem health.

These objectives need to be evaluated in terms of reference points to ensure that the performance of management strategies is successful in achieving the desired state of the resources and limiting undesirable growth in the pressures (fisheries).

8.2.2 Step 2: Identify meaningful indicators and decide on reference points

Indicators are often used as surrogates to monitor changes in the condition or state of a parameter of interest which is not easily measurable (Noss 1990). In order for them to be effective they need to be easily quantifiable and have a clear causal relationship with the parameter which they represent (Vos *et al.* 2000). Furthermore, they must have clear links to management goals and objectives they were selected to signify (Pomeroy *et al.* 2004). Selecting appropriate indicators for assessing the performance of management actions is fundamental to evaluating whether management objectives are achieved and allows for future improvement (Pomeroy *et al.* 2005). The process of indicator selection and reference point development is an iterative process requiring initial selection followed by testing and evaluation which includes the participation of stakeholders (FAO 2003). In this section preliminary indicators are identified and provisional reference points recommended; these require further evaluation and testing in collaboration with stakeholders in the future.

In order to evaluate performance of each indicator, reference points need to be established which reflect the position of the indicator relative to the desired state as interpreted from the management objectives which it reflects (Caddy and Mahon 1995; Bennetts *et al.* 2007). Quantitative biological information obtained from stock assessment models may be used to set reference points, or alternatively they can be based on qualitative or semi-quantitative criteria which reflect societal expectations for desired future state (Caddy and Mahon 1995). Two types of reference points are commonly used in monitoring, namely, target and limit reference points. Target reference points (TRP) relate the high level management objectives to each indicator and reflect the desired state and expected and acceptable range of the indicator value which should be maintained through ongoing management (Garcia and Staples 2000; Sainsbury and Sumaila 2003). Limit reference points (LRP) are selected to signify when indicators attain unacceptable levels which threaten long-term sustainability of the resources and indicate when further management action is required (Caddy and Mahon 1995; Sainsbury and Sumaila 2003). They are therefore used to trigger additional management intervention in order to prevent unacceptable levels of ecological degradation from occurring (Sainsbury and Sumaila 2003; Rice and Rivard 2007). The performance of each indicator is assessed against TRPs and LRPs and allows for the overall evaluation of management initiatives in achieving the stated goals and objectives. Performance evaluation supports decision-making through identifying when additional management measures are required and can assist in identifying what types of intervention will be most effective. Although reference points such as maximum sustainable yield, spawner stock biomass estimates, and fishing mortality levels have been widely used in stock assessment models in the past, the models are complex, require considerable data and may have considerable uncertainty in parameter estimates (Caddy and Mahon 1995). In order to avoid the requirement for complex models, simple indicators have been selected and reference points recommended based on the best available data taking into account the local management objectives.

Candidate indicators were selected for the "Pressure" and "State" categories based on the baseline assessments of fishery activities (Chapters 5 and 6) and fish communities (Chapters 3 and 4) respectively. During the preparation of these chapters intensive investigations were conducted to determine what information was available for reporting on the state and pressures within Algoa Bay at a suitable spatial resolution. Where no suitable data were available cost-effective yet scientifically robust baseline studies were designed to fill gaps in knowledge. The data used in these previous chapters therefore represents the best available or most easily obtainable data which can be used for indicators in future long-term monitoring, and provides the basis for setting reference points. Where no detailed assessments were conducted for some targeted fisheries resources (pelagic stocks, chokka-squid etc.) in this thesis, potential indicators and data sources or programmes are suggested and identified, and further development is required in the future. Potential indicators and reference points are suggested and discussed separately below.

(a) Pressures

(i) Commercial fisheries

Candidate pressure indicators (Table 8.1) for commercial fisheries in Algoa Bay need to fulfil three main roles: i) they need to report on fine-scale spatial and temporal trends in fishing pressure within Algoa Bay as management actions implemented on a local scale (such as no-take zones etc.) may influence fishery dynamics over a small geographic area (within Algoa Bay); ii) they need to report on the annual trends in local fishing pressure in Algoa Bay for each sector to evaluate long-term temporal changes; and iii) they need to evaluate local trends relative to the national fishery to aid interpretation and differentiation between trends driven by national regulatory amendments (e.g. reduction or increase in the number of rights or TAC on a national level) and biophysical related changes (e.g. changes the distribution of stocks; local depletion) which may act over differing spatial scales.

Primary indicators of commercial pressure in Algoa Bay need to reflect the local resource use patterns (Pomeroy *et al.* 2004). Important indicators of fishing pressure for each sector are therefore the fishing capacity and effort (Garcia and Staples 2000; Degnbol and Jarre 2004; Piet *et al.* 2008). These are quantified as the number of vessels fishing per month (vessels.month^{-1}) as well as the total annual effort in boat-days (boat-days.year^{-1}) (Table 8.1). In addition the measure of the extent of extractive resource use in Algoa Bay in the form of total annual landings (tons.annum^{-1}) and the composition of the landings (where relevant) is an important indicator of the extractive harvesting pressure on the ecosystem (Degnbol and Jarre 2004; Pomeroy *et al.* 2004).

The data source for these indicators differs between the spatial scales on which the indicator is reported, with each data source having its own inherent biases. On a fine scale (per fishing ground within Algoa Bay) VMS or observer data, or a combination thereof (depending on the sector), needs to be used as reported logbook data does not reflect spatial trends at sufficient resolution to aid or assess local level management initiatives. However, the VMS and observer datasets do not necessarily represent the entire fleet or complete period due to possible equipment malfunction (VMS), and observer records are limited to infrequent onboard monitoring trips. Nonetheless, provided there is sufficient replication across vessels and seasons, both the observer and VMS data reflect fine scale spatial trends accurately. However, due to the incomplete nature of the data the units of measure are not comparable to other data sources (i.e. total annual effort cannot be determined). Fine scale spatial indicators therefore need to be reported as the relative proportion of effort or harvest in Algoa Bay per fishing ground determined using each data source (VMS or Observer) as this will allow long-term comparison which is independent of sample size or sample units. VMS and observer data are available from the national regulatory authority. The main fishing grounds used by each sector and the baseline levels of effort and harvest were indentified and reported in Chapter 6. This data serves as the benchmark for future monitoring representing conditions prior to any new local management initiatives being implemented and therefore serves as the basis for defining TRPs and LRPs.

On a local level (bay level) effort per commercial sector should be monitored as the mean number of vessels fishing in Algoa Bay per month (vessels.month^{-1}) and the total annual number of boat-days fished locally (boat-days.year^{-1}) (Table 8.1). Total harvest (tons.year^{-1}) captured locally and the composition of the harvest also serves as an important indicator of fishing pressure for each sector. Data for these indicators are based on fishery-dependent logbook records which are submitted by the rights holders and maintained by the national regulatory authority.

The overall management objectives for Algoa Bay in terms of commercial fisheries are to allow equitable use of the resources in a sustainable manner. This requires accommodating similar levels of fishing effort for all sectors, provided that the fishery is regarded as sustainable, but preventing further increases in effort which may place additional pressure on the resources locally and threaten future sustainability. The TRPs for indicators of commercial fishing pressure should therefore be based on maintaining similar levels of fishing effort to that occurring since the recent allocation of long-term rights (2006-2010). It is therefore desirable to maintain fishing effort and harvest within an acceptable range of the baseline levels and a TRP of within 10% of these baseline levels is considered acceptable for each sector. Any increase in commercial fishing effort on individual fishing grounds, or in Algoa Bay as a whole should trigger further investigation as to the reasons leading to the escalating fishing pressure, and possibly lead to additional management actions being implemented to curb or reduce further growth. A 25% increase in effort or harvest between consecutive years is deemed considerable and appropriate as a LRP which should trigger urgent additional management intervention. Reference trajectories should also be monitored and an increase in effort or harvest of 10% or more per year over three consecutive years is deemed sufficient to raise concern as to the pressures being placed on the target resources and the sustainability thereof. This should also serve as a LRP and trigger further investigation and management action. Baseline data provides a valuable means for evaluation of future trends, however, any significant changes in indicator values need to be carefully assessed and understood in light of changes in the ecosystem, This requires that adaptive management is based on a science-based and intelligent approach which takes the natural variability of the marine ecosystem into account, and reference points based on baseline data may need to be re-evaluated to account for this.

Fluctuating market demands and economic value of marine products may influence species specific targeting in multi-species fisheries. Monitoring changes in the composition of the landings of multi-species fisheries will therefore serve as an indicator of species specific changes in targeting pressure. The relative contribution of dominant species to the harvest is therefore an important indicator for monitoring. A stable catch composition should be maintained and a TRP within 10% of the mean baseline levels of dominant species should be used for evaluation. Furthermore there should be no significant differences in community structure of the catch composition between years using multivariate tests. A 25% change in the relative contribution of a species to the catch is deemed a significant shift in the composition and should serve as the LRP.

Unselective fishery practices may also exert pressures on the ecosystem through high mortality rates of unwanted species or undersized individuals through high grading. Monitoring the proportion of catch discarded relative to the total catch is therefore an important pressure indicator (Degnbol and Jarre 2004; Piet *et al.* 2008). This is particularly relevant to unselective and multi-species fisheries and will contribute to the understanding of ecosystem effects of such activities and to providing recommendations for improved management in the future. Observer data should therefore be used to quantify the proportion of catch discarded relative to the total catch. The discarded portion of trawl catches on the south coast of South Africa have been shown to range from 4 to 19% of the total landed catch (Walmsley *et al.* 2007a). A TRP of 10% of the total catch weight is considered sufficient to prevent selective targeting of bycatch species and high grading, taking into consideration the unselective nature of the fishing gears. A LRP of 20% of the total catch weight should be used to trigger further investigation and initiate additional management responses to regulate high discard rates locally in Algoa Bay.

Table 8.1. Proposed indicators for monitoring fisheries pressure in Algoa Bay.

	Description	Indicator	Sector	Data source	Target reference point	Limit reference point
Commercial fishery sectors						
Fine scale (fishing area)	Fishing effort	% boat-days / fishing ground	LF; CS; SPPSF; DT; SLL	DAFF VMS and Observer data	Proportion of effort within 10% of baseline levels	Proportion of effort within 25% of baseline levels Increasing trajectory in effort with annual increases of 10% or more per year over three consecutive years
	Harvest	Catch composition / fishing ground	LF; SPPSF; DT; SLL	DAFF Observer data	Relative contribution of dominant species to harvest within 10% of baseline levels	Relative contribution of dominant species to harvest within 25% of baseline levels
Local scale (Bay wide)	Fishing effort	Vessels.month^{-1}	LF; CS; SPPSF; DT; SLL	DAFF Logbook data	Effort within 10% of baseline levels	Effort within 25% of baseline levels Increasing trajectory in effort with annual increases of 10% or more per year over three consecutive years
		Boat-days.year^{-1}				
	Harvest	Tons.year^{-1}	LF; CS; SPPSF; DT; SLL	DAFF Logbook data	Annual harvest within 10% of baseline levels Relative contribution of dominant species to harvest within 10% of baseline levels No significant differences in multivariate analysis	Annual harvest within 25% of baseline levels Increasing trajectory in harvest with annual increases of 10% or more per year over three consecutive years Relative contribution of dominant species within 25% of baseline
		Catch composition				
	Discards	% of catch weight discarded	SPPSF; DT; SLL	DAFF Observer data	Within 10% of total catch	Within 20% of total catch
Relative to national fishery	Fishing effort	% of National fleet fishing Algoa Bay per year	LF; CS; SPPSF; DT: SLL	DAFF logbook system	Within 10% of baseline levels	Within 25% of baseline levels
		% of annual national boat-days.year^{-1} in Algoa Bay				
	Harvest	% of annual national harvest (tons) in Algoa Bay				
Recreational fisheries						
Fine scale	Fishing effort	Anglers.km^{-1}	Shore (including subsistence)	Aerial and RC surveys	Within 25% of baseline levels per survey zone	Within 50% of baseline levels per survey zone or access point Increasing trajectory in effort with annual increases of 10% or more per year over three consecutive years
		Launches.access point^{-1}	Skiboat	AP surveys and logbooks		
Local scale	Fishing effort	Est. Anglers.year^{-1}	Shore (including subsistence)	Aerial and RC surveys	Within 25% of estimated baseline value	Within 50% of estimated baseline value
		Est. Launches.year^{-1}	Skiboat	AP surveys and logbooks		
	Harvest	Catch composition (shore and skiboat)	Shore (including subsistence); skiboat	RC and AP surveys	Relative contribution of dominant species within 20% of baseline	Relative contribution of dominant species within 50% of baseline
		Mean length of dominant species (shore and skiboat)		RC and AP surveys	Mean length of dominant species within 10% of baseline	Mean length of dominant species within 20% of baseline
Illegal, unreported and unregulated fisheries						
	IUU incidents	Incidents.patrol^{-1} in Algoa Bay Incidents.month^{-1} Incidents.year^{-1} (Includes infringements on minimum legal size and exceeding total allowable harvest or bag restrictions)	Abalone poaching; All commercial and recreational sectors	DAFF monitoring and enforcement database; SANParks enforcement logs; DAFF Hotline; Independent monitoring surveys	Zero	Within current levels; increasing trajectory

Notes: LF = linefish; CS = Chokka-squid; SPPSF = Small pelagic purse seine fishery; DT = Demersal trawl; SLL = Shark long-line;
Commercial sector reference points based on averages from post long-term rights allocation 2006-2010
RC = roving creel; AP = Access point

Changes in fishery regulations, rights allocations of quotas in national fisheries may influence the pressures exerted locally in Algoa Bay through shifts in the spatial distribution of effort. The relative proportion of fishing effort and harvest in Algoa Bay to that of the sector nationally therefore needs to be monitored. Indicators to reflect local changes in fishery dynamics relative to the national fisheries include the annual effort (boat-days.year^{-1}) and harvest (tons.year^{-1}) in Algoa Bay expressed as a percentage of national values for each commercial sector (Table 8.1). The importance of Algoa Bay to each commercial sector differs based on the operational range of vessels, location of home ports and processing facilities, as well as the distribution of target stocks. Changes in the number of rights holders (vessels) utilising fishing grounds in Algoa Bay may reflect broad geographical changes in stock status or the allocation of fishery rights. Significant increases may require management intervention in order to alleviate fishing pressure locally. The number of rights holders utilising fishing grounds in Algoa Bay as a percentage of the total number of rights issued nationally is a key indicator of the importance of local fishing grounds and the distribution of fishing capacity on a national scale. Similarly the percentage of national effort exerted within, and the percentage of harvest originating from Algoa Bay provides a measure of the relative importance of Algoa Bay on a national level. These indicators are based on logbook data and annual TAE and TAC allocations which are available from the national regulatory authority. Management in Algoa Bay should aim to maintain a similar relative proportion national fishing effort in Algoa Bay and the TRP should therefore be within 10% of the baseline levels. Increases of more than 25% in effort from baseline conditions should serve as a LRP and trigger further investigation and management action.

Although long-term data are available for all commercial sectors, several changes have occurred in the management of these fisheries since the initiation of the logbook system. Long-term commercial rights were allocated in 2005/2006 and led to changes in rights ownership and the number of rights and quotas issued in many sectors, and therefore resulted in changes in the spatial distribution of fishing effort over differing scales. Reference levels for future evaluation of commercial fisheries in Algoa Bay should therefore be based on data since the allocation of long-term rights in 2005/2006 and prior to the implementation of new local management initiatives in Algoa Bay.

(ii) Recreational fisheries

Recreational fisheries in South Africa are open access with no limit on the number of participants. Participation in the South African recreational fishery has been on the increase (Clarke and Buxton 1989; Coetzee *et al.* 1989; van der Elst 1989; van der Elst 1990; Brouwer *et al.* 1997; McGrath *et al.* 1997; Brouwer 1997) placing increasing pressure on targeted resources. Monitoring trends in angler density and total estimated annual effort in the recreational sector is therefore a critical aspect for evaluating pressures on the local marine resources. Average density in anglers per kilometer (anglers.km^{-1}) within demarcated survey zones of predefined length is therefore a key indicator for tracking fine scale spatial trends in shore angler effort and should be combined with estimates of total angler numbers per year in Algoa Bay to assess long-term temporal trends locally (Table 8.1). Similarly the average number of boat launches per access point serves as an indicator of fine scale recreational skiboat fishing pressure and annual estimated effort within Algoa Bay provides a means to assess longer term temporal trends in the local skiboat fishery (Table 8.1).

Chapter 5 provides a comprehensive assessment of the recent levels of participation in the recreational fishery in Algoa Bay and TRPs and LRPs should be based on this baseline information. Due to the high variability in both angler density and annual estimates of effort, maintaining participation levels in the recreational fisheries within 25% of the baseline levels is considered appropriate and therefore serves as the TRP. Due to short-term and spatial fluctuations in recreational fisheries a high LRP of 50% is deemed acceptable between years; however, progressive increases of 10% per annum over a period of three consecutive years (reference trajectory) should also serve to trigger further investigation and management responses.

Travel distances and accessibility, holiday periods, weather and habitat related factors, as well as the angler's motivation for fishing (e.g. relaxation vs. competitive) all contributed to the high variability in recreational CPUE and effort data in the current study (Chapter 5) and is common in many recreational fisheries (Attwood and Farquhar 1999; Webley *et al.* 2010). As total annual landing is based on the product of CPUE and total effort (Pollock *et al.* 1997) the variability in this estimate is also highly variable and unlikely to provide an accurate indicator for monitoring changes in recreational pressure. However, monitoring changes in the composition of the retained catch, and the mean length of species will provide better indications of the pressures exerted by recreational anglers (Smale and Buxton 1985). Such changes in catch composition have been reported in the South African recreational shore fishery (Bennett 1991; Bennett *et al.* 1994; Brouwer *et al.* 1997; Brouwer and Buxton 2002) as well as the offshore fishery (Hecht and Tilney 1989; Brouwer 1997; Brouwer and Buxton 2002; Donovan 2010) indicating serial overfishing with a shift in pressure to new species. Monitoring the relative contribution of dominant species in the total catch, and the mean length is therefore a key indicator for recreational fishing pressure in Algoa Bay. Reference points of 25% for targets and 50% for limit should be used to monitor and evaluate changes in the relative proportion of dominant species in the catches. Length data provides a more accurate means to monitoring changes and TRP and LRP of 10% and 20% change in mean length of dominant species should be employed.

(iii) Illegal, unreported and unregulated fisheries

Illegal, unreported and unregulated (IUU) fishing activities place considerable additional pressure on fishery resources (le Gallic and Cox 2006) and in Algoa Bay the illegal abalone fishery is of particular concern (Raemaekers and Britz 2009). Monitoring trends in the level of illegal fishing in response to enforcement efforts is therefore critical for evaluating management effectiveness. Key indicators include the number of illegal incidents observed in Algoa Bay per vessel patrol conducted by the National DAFF compliance section. In addition continuous local enforcement is undertaken by SANParks and the number of incidents per month observed is an indicator of poaching pressure in Algoa Bay. A national reporting hotline also exists and the number of illegal incidents reported in Algoa Bay also serves as an indicator. Although not analysed in this thesis, IUU fisheries data does exist for Algoa Bay (Raemaekers and Britz 2009; Raemaekers 2009). In addition transgressions of the regulations by the commercial and recreational sectors also form a component of IUU. These include exceeding TACs or daily bag limits, and retaining catches below the species specific minimum legal size limits. Some of the required data are available through the existing national monitoring and reporting programmes but additional dedicated surveys will also be required, particularly for the recreational fisheries. Management objectives aim to achieve a zero incident record for the TRP and the LRP should not exceed the current levels of IUU activities reported in Algoa Bay. Should this occur additional management actions need to be considered.

(b) State

The overall health of an ecosystem and the condition of biological communities is largely dependent on sustaining the quality of the habitats which they inhabit. In this study reef areas were mapped coarsely to identify suitable study sites and diving was conducted to verify habitat quality where possible. Large scale sand movement occurs within Algoa Bay (Illenberger 1993) and temporal changes in habitat availability or quality may occur influencing the distribution of mobile species. Monitoring changes in the distribution and quality of habitats is therefore an important aspect to assessing the state of biological communities and the state of the ecosystem (Pomeroy *et al.* 2004). Furthermore water quality is an important indicator of the state of the local environment (Pomeroy *et al.* 2004). Changes in selected water quality parameters resulting from spillages or increased point source discharges may affect the structure of marine communities locally and need to be known to aid interpretation of causal relationships. Monitoring discharge points to ensure compliance with discharge standards and regulations is therefore a critical aspect for assessing changes in the state of the local Algoa Bay environment. In addition periodic monitoring of water quality across Algoa Bay will identify unusual conditions and aid in interpreting causal relationships between water quality and biological indicators.

Marine benthic communities play an important role in structuring fish communities in both reef and non-reef areas. Due to the sessile nature of most benthic invertebrates they provide a good means for evaluating long-term changes in ecosystem structure as they do not emigrate in response to short-term changes in environmental conditions. Data on marine benthic communities can therefore assist in the interpretation of the responses of fish communities to long-term environmental or anthropogenic changes. Although baseline data on invertebrate community structure have been collected, it did not form part of this thesis as the focus was primarily on assessing fisheries and fishery related species to facilitate marine spatial planning. Monitoring macro-benthic fauna and flora does, however, form an integral component for future evaluation and the existing data requires further investigation for the identification of indicators and development of monitoring protocols. Additionally interpreting changes in higher trophic levels (community or indicator species) can be facilitated through monitoring changes in the dietary composition of indicator species as changes in the local abundance of prey species may alter the community structure, abundance and size composition of higher trophic level indicators. This will require the establishment of dedicated research programmes.

Changes in the state of fishery resources can be monitored at varying scales from community to species level (Jennings 2005). State indicators include tends in community composition using multivariate analysis, univariate diversity indices, or indicator species which are used as surrogates (in respect of EAF). Community level indices are often criticised as they have no clear direct causal link to pressures and are therefore difficult to interpret (Keough and Quinn 1991). Nonetheless changes in fish community assemblages have been used successfully as an indicator of environmental change in response to the discharge of thermal waters in coastal ecosystems (Teixeira *et al.* 2009) and to assess distributional changes in response to climate change (Dulvy *et al.* 2008). Significant changes in community structure as a result of fishing pressure have also been reported over long temporal periods (McHugh *et al.* 2011). Furthermore, the influence of natural environmental drivers on ecological condition can be investigated through evaluation of shifts in fish community structure within MPAs where anthropogenic pressures have been eliminated. Community stability (Labropoulou and Papaconstantinou 2000) or directional changes in assemblages (Barrett *et al.* 2007) can then be directly linked to natural environmental drivers. These examples indicate the value of using community indicators for monitoring changes in response to natural or anthropogenic pressures. Furthermore, previous studies in South Africa have recommended the use of multivariate techniques on fish community data for monitoring responses to environmental change within MPAs (Bennett 2007). In addition the use of simple species lists (presence/absence data) have been demonstrated as being suitable for assessing community changes using the AvTD and VarTD diversity indices (Clarke and Warwick 2001a). Multivariate and univariate community analyses were therefore considered effective indicators for monitoring temporal changes in ecosystem state using fish communities and should be employed in future monitoring within Algoa Bay (Table 8.2).

Baseline data analysed during this study (Chapter 3 and 4) serves as the benchmark for future monitoring of fish community structure. Due to the multitude of factors influencing community structure and the difficulty in interpreting the driving forces of change, precautionary reference points have been suggested. The overall aim of local management in Algoa Bay is to protect and sustain fish stocks and biodiversity, and aid in the recovery of depleted stocks through the development of new MPAs which will contribute to supporting adjacent fisheries. TRPs for univariate diversity indices should therefore remain stable with no significant inter-annual differences. However, enhanced management measures may lead to a recovery of communities, particularly in new no-take zones, and diversity may therefore increase. TRPs should therefore be based on maintaining similar levels of diversity, or increasing the diversity of ichthyofaunal communities in Algoa Bay. Declining trends in diversity are seen as a sign of decreasing community health and serves as a trigger warranting further investigation and explanation as to the reasons for the change. The LRP is therefore based on a significantly lower diversity value between consecutive years, or a progressive declining trajectory in diversity indices. Changes in each diversity index used must, however, be interpreted with care as previously exploited species may become dominant as the stocks recover in areas where they are protected.

Similarly for multivariate analyses of fish assemblages stability should be maintained and no significant difference in community structure should be observed between consecutive years, and should serve as the TRP. A significant directional temporal change in community structure to a community which shows signs of overexploitation is the basis for the LRP using multivariate analyses. In addition where comparable exploited and no-take sites are available, the protected site can serve as a reference point. The TRP should aim to maintain similar diversity and community structure between exploited and protected sites. A significant decrease in diversity in exploited sites relative to protected sites, and a declining trajectory in the exploited site over multiple years should serve as the

LRP. Similarly significantly different community structure between exploited and protected sites should serve as a trigger for further investigation if the community in the exploited site exhibits signs of declining health.

Indicator species can be used effectively to represent the state of biological communities or ecosystems, particularly when the influence of direct pressures, such as fishing, have a clear causal relationship with the selected indicator species. Although there has been much criticism over the use of indicator species in the past their use is well-established in ecological studies (Noss 1990). However, care needs to be taken when selecting indicator species to ensure that they have a clear relationship with the parameter of interest. Suitable indicator species for monitoring the state of the resources when fisheries activities are the subject of investigation must be abundant within the study area, of socio-economic value, targeted and sensitive to fishing pressure, respond rapidly to management interventions, easy and cost effective to measure, representative of the community or ecosystem of interest and have clear links to the state of the ecosystem (Noss 1990; Keough and Quinn 1991; Carignan and Villard 2002; Nicholson and Jennings 2004; Piet and Jennings 2005; Rice and Rochet 2005; Shin *et al.* 2010a). Indicator species also serve an important role in evaluating the effects of long-term changes in environmental conditions when monitored within MPAs in which they are exempt for direct extractive pressures.

More than one indicator species may be required to ensure adequate representation of different ecosystem components and to track the influence of pressures which may impact on species differently. The selection and evaluation of specific indicator species is discussed in section 8.2.3 below. Relative abundance and mean length or mass have been used as indicators to monitor and evaluate responses of fish stocks to exploitation and management interventions (Degnbol and Jarre 2004; Pomeroy *et al.* 2004; Jennings and Dulvy 2005; Bennett 2007). Relative abundance and mean length of indicator species were selected for evaluating long-term temporal changes in the state of linefish populations in Algoa Bay as the data are readily obtainable using fishery independent survey methods (Table 8.2). An additional indicator was selected to represent the target stock available to commercial and recreational fisheries and serve as an early warning indicator for overexploitation. The relative abundance of the indicator species above the MLS was considered an important measure of the status of the population with regards to the fishery management regulations (Table 8.2).

Reference points for the relative abundance and mean size of indicator species need to be developed for both protected and exploited sites. In the absence of fishing pressure (MPA sites) the abundance and size of targeted species should remain stable, or increase progressively as stocks recover from past exploitation in recently established no-take zones. TRPs for the abundance and length in no-take zones should therefore be to maintain similar levels (no-significant difference between consecutive years) or alternatively indicate an increasing trajectory due to recently improved protection. The LRP would be a significant decrease between consecutive years or a declining trajectory over numerous years, as this would indicate declining stock status.

Table 8.2. Proposed indicators for monitoring the state of fishery resources in Algoa Bay.

Category	Indicator	Data source	Target reference point	Limit reference point
Fish stocks				
Community level	Diversity indices	Fishery independent surveys - Controlled angling; UVC; BRUVs; ROV; research trawls	**MPAs** – Diversity indices remain stable or increasing trajectory **ES** – No significant difference from similar MPA site; no significant difference between consecutive years in ES	**MPAs** – Significantly lower diversity in successive year; Declining trajectory in diversity indices **ES** – Significant difference from similar MPA sites; Declining trajectory in diversity indices in ES
	Multivariate analysis		**MPAs** – No temporal trajectory evident **ES** – No significant difference from similar MPA site; no significant difference between consecutive years in ES	**MPAs** – Temporal trajectory evident **ES** – Significantly different community from similar MPA site; Temporal trajectory in ES with move towards signs of overexploitation
Species level (indicator species)	Mean length		**MPAs** – No significant decline between consecutive years, or increasing trajectory in mean length **ES** – Mean length within 20% of similar MPA site; no significant difference between consecutive years in ES	**MPAs** – Significant decline between consecutive years; progressive declining trend **ES** – Mean length not less than 50% of similar MPA site; Significant difference between consecutive years or declining trajectory
	Relative abundance fish above MLS		**MPAs** – No significant decline between consecutive years, or increasing trajectory in relative abundance **ES** – Relative abundance of indicator species not less than 65% of similar MPA site; no significant difference between consecutive years in ES	**MPAs** – Declining trend in ratio **ES** – Less than 25% of similar MPA site; Declining trajectory
	Relative abundance Count per area; CPUE;		**MPAs** – No significant decline between consecutive years, or increasing trajectory in relative abundance **ES** – Relative abundance of indicator species less than 60% of similar MPA site; no significant difference between consecutive years in ES	**MPAs** – Significant decline between consecutive years; progressive declining trend **ES** – Relative abundance of indicator species not less than 40% of similar MPA site; Significant difference between consecutive years or declining trajectory
Pelagic stocks				
Biomass	Estimated biomass	Fishery independent acoustic surveys undertaken by DAFF	Distribution and abundance of stocks highly variable, future research to focus on identifying reference points	
Chokka-squid stocks				
Relative abundance / biomass	CPUE; estimated biomass	Fishery independent surveys on spawning aggregations - CPUE; acoustic and ROV surveys	New programme to be established	
Habitats and benthos				
Habitats	Distribution Map of reefs, sediment types etc	Side scan sonar; Acoustic Ground Discrimination Systems (AGDS)	New programme to be established	
Macro-benthos	Community structure	ROV, UVC, Jump camera surveys	Chalmers unpublished data, programme to be established	
Diet composition	Predator gut content analysis representative species	New independent research surveys	To be established	
Water quality	Selected parameters including: *E.coli*; hydrocarbons; heavy metals; oxygen; phosphates and nitrogenous compounds; turbidity	Municipal discharge monitoring programmes; SAEON plankton and water quality monitoring; New bay wide programme to be established to meet gaps.	Based on Department of Water Affairs water quality guidelines for marine water and industry specific effluent discharge guidelines	

Notes: MPA reference points based on temporal comparisons of the same protected area
ES = Exploited sites, reference points based on comparison to similar protected site

MPAs serve as a basis for setting reference points for long-term comparison with exploited sites (Sainsbury and Sumaila 2003). However, the recovery of populations within newly established no-take zones in the absence of fishing pressure must be taken into consideration if these sites are to be used as reference points for exploited populations. Reference points derived from indicators in no-take zones for exploited sites should therefore be based on a relative measure between the two sites. This will account for ongoing recovery in no-take zones, or potential future decline due to environmental drivers, thereby aiming to maintain a steady ratio in fish abundance between MPA and no-take zones in the absence of true baseline data. Management objectives for Algoa Bay aim to maintain sustainable fisheries, ensure protection of key species and contribute to the recovery of depleted stocks. The TRPs for the relative abundance of indicator species in exploited sites should therefore be set conservatively to contribute to overall stock recovery in Algoa Bay. Götz (2005) and Götz *et al.* (2009b) determined a 20% difference in roman CPUE between exploited and protected sites across the Goukamma MPA border resulted in a 40% reduction in roman biomass. A similar comparison of the roman CPUE between the Tsitsikamma National Park MPA and an adjacent fishing area at Plettenberg Bay indicated an approximate 70% lower abundance in the exploited site highlighting the potential magnitude of impact exploitation can have on fish populations (Smith 2005b). A conservative TRP for the relative abundance of indicator species in exploited sites in Algoa Bay is considered to be between these two studies and in the region of 60% of that in similar protected sites, while an acceptable LRP is considered to be 40% of protected levels. Where no comparable protected sites are available to serve as reference points, the TRP for relative abundance and mean length of indicator species should be based on the current values for Algoa Bay. The LRP should be based on significant decrease between successive years or a progressive declining trajectory in the relative abundance or mean size of indicator species.

(c) Response

Responses represent the actions which are implemented by regulatory authorities in order to bring about a change in pressure and state within an ecosystem (Pajak 2000). Within South African fisheries responses have traditionally been implemented and evaluated on a national rather than a local level. Examples include the number of rights issued per sector and annual TAC allocations, species specific size and bag regulations and closed seasons and areas. In moving towards ecosystem based management on a local level, specific management interventions need to be monitored and quantified in order to evaluate responses.

A local level management body or steering committee needs to be established to facilitate coordination of management interventions and monitoring tasks within Algoa Bay (Ehler 2003; Pomeroy *et al.* 2004) (Table 8.3). This steering committee needs to drive the process for development of a formal management plan including the development of a common vision, definition of management goals and objectives and the clarification of the roles and responsibilities of participating agencies.

The frequency of meetings and representation of agencies on the steering committee is a key indicator of commitment to integrated and holistic local level management. One of the objectives of this study was to identify priority areas for conservation which would contribute to enhancing the level of protection afforded to marine biodiversity. One of the key responses is therefore the management actions which are implemented to improve conservation, including spatial expansion of the MPA network within Algoa Bay. The percentage of coastal and offshore area proclaimed as MPA and no-take zones is therefore an important indicator of institutional response (Pajak 2000; Pomeroy *et al.* 2004) (Table 8.3). Additional responses include financial and manpower commitments to management, enforcement, monitoring and education/awareness (Ehler 2003; Pomeroy *et al.* 2004). Indicators of responses would therefore include operational budgets, capital investment and a number of positions allocated to management annually (Table 8.3). Furthermore, the number of patrols, percentage of vessels/anglers inspected per patrol, and compliance rates reflect the management response to enforcement responsibilities. Education and raising public awareness is a key aspect to improving compliance (Pomeroy *et al.* 2004). Partnership agreements for undertaking awareness and educational events as well as the numbers of meetings and amount of signage and pamphlets produced and distributed reflect the response of management agencies. Adaptive management requires ongoing research to improve management actions. The number of research projects and amount of budget allocated to research are therefore also important response indicators.

Table 8.3. Proposed response indicators for monitoring the implementation of management action.

<table>
<tr><th>Category</th><th>Indicator</th><th>Data source</th></tr>
<tr><td colspan="3">Institutional arrangements</td></tr>
<tr><td rowspan="7">Management and coordination</td><td>Development of a steering committee</td><td rowspan="7">SANParks, Steering committee</td></tr>
<tr><td>Development of a vision, goals and objectives</td></tr>
<tr><td>Clarification of roles and responsibilities</td></tr>
<tr><td>Number of institutions represented on the steering committee</td></tr>
<tr><td>Frequency of steering committee meetings</td></tr>
<tr><td>Number of institutional agreements/management plans</td></tr>
<tr><td>Number of stakeholder partnerships developed for monitoring</td></tr>
<tr><td colspan="3">Area protection</td></tr>
<tr><td rowspan="2">MPAs</td><td>% of Algoa Bay in no-take zones</td><td rowspan="2">Management authority</td></tr>
<tr><td>% of Algoa Bay in MPA</td></tr>
<tr><td colspan="3">Monitoring and enforcement</td></tr>
<tr><td rowspan="4">Financial</td><td>Operational budget/year</td><td rowspan="4">Management agencies – SANParks; DAFF; NMBM; Ndlambe</td></tr>
<tr><td>% operational budget spent/year</td></tr>
<tr><td>Capital investment/year</td></tr>
<tr><td>Number of permanent positions/year</td></tr>
<tr><td rowspan="5">Monitoring and enforcement</td><td>Number of patrols/year</td><td rowspan="3">SANParks; DAFF; NMBM; Ndlambe municipality</td></tr>
<tr><td>% of compliance checks/patrol</td></tr>
<tr><td>% non-compliance</td></tr>
<tr><td>% onboard observer trip/sector/year</td><td rowspan="2">DAFF</td></tr>
<tr><td>% observer inspection/access point</td></tr>
<tr><td colspan="3">Education, research and stakeholder engagement</td></tr>
<tr><td rowspan="6">Education, research and stakeholder engagement</td><td>Financial allocation/year for education and communication strategies</td><td rowspan="6">SANParks/Steering committee</td></tr>
<tr><td>Number of information boards erected</td></tr>
<tr><td>Number of workshops/information sharing meetings</td></tr>
<tr><td>Awareness of regulations (%)</td></tr>
<tr><td>Funding allocated to research projects</td></tr>
<tr><td>Number of research projects</td></tr>
</table>

8.2.3 Step 3: Develop a sampling programme and identify the analytical tools required for evaluation

Following the identification of PSR indicators, protocols for the collection of the required data need to be established. This section outlines the analytical methods used to evaluate baseline data presented in earlier chapters in order to inform the development of a scientifically robust monitoring protocol. In instances where indicators are based on available data sources and no sampling protocol is required, no methods are presented, yet the data sources are discussed with regards to the availability, quality and suitability for future monitoring. A summary of the proposed survey design is illustrated at the end of the chapter in Figure 8.11.

(a) Pressures

(i) Commercial fisheries

Data required for pressure indicators selected for monitoring commercial fisheries in Algoa Bay is based on three sources of information available from the national regulatory authority, DAFF. This includes fishery-dependent logbook data, onboard and access point observer programme data and VMS data which are routinely collected by DAFF to monitor fishery trends on a national level and for enforcement purposes. This data are therefore available for monitoring of commercial fishery activities on a local level at no additional cost and without the need for a dedicated programme. Although a sampling protocol is not required, lines of communication need to be established with DAFF to ensure data are readily obtainable when required. The following three sources of data can be used to develop a system to verify the accuracy of fisheries-dependent data and improve the accuracy of monitoring on a local level. Auditing logbook data through comparisons with observer and VMS data and communication of the results to fishery stakeholders will contribute to improving compliance with logbook reporting requirements.

Logbook data

All commercial rights holders are required to submit monthly logbook data which includes daily landings, effort and fishing locations on a coarse spatial grid. This data are captured by DAFF into central databases. Fisheries-dependent logbook programmes are useful as they provide low-cost information on fishery dynamics. However, the accuracy of the data is dependent on the honesty of the rights holder or vessel skipper and the efficacy of using such data for management purposes has been questioned (Cotter and Pilling 2007). Inaccuracies may include non-reporting of fishing trips, misreporting of landings to overcome regulatory restrictions or levies, incorrect spatial data and poor attention to species identification or accidental misidentification. Furthermore, no information on the mortality of bycatch species or discards due to high grading is reported in logbook systems. Despite these limitations logbook data have been widely used in the past to assess and monitor fishery activities locally (Crawford and Crous 1982; Griffiths 2000; Donovan 2010) and globally (Murawski *et al.* 2005; Cotter and Pilling 2007; Pedersen *et al.* 2009).

Within South Africa there has been no formal system in place to validate logbook catch returns against actual effort, and despite monthly submissions being a permit requirement for commercial rights holders, the reporting accuracy is often poor (Sauer *et al.* 1997; Attwood and Farquhar 1999; Griffiths

2000). The importance of catch return submissions was recently highlighted during the allocation of medium and long-term fishing rights in all commercial sectors which was primarily based on socio-economic dependence and historical performance in the fishery using logbook data submitted by rights holders themselves. Many historical rights holders failed to demonstrate active participation, and hence socio-economic dependence on the fishery as they had not submitted catch returns, or had only made infrequent submissions in the past. This process is likely to have increased the awareness of the remaining rights holders as to the importance of monthly submissions thereby contributing to more frequent and submission. However, the accuracy of each submission is still uncertain. VMS and observer data sources provide an independent means to validate the accuracy of logbook data (Palmer and Wigley 2009; Bastardie *et al.* 2010; Gerritsen and Lordan 2011) and through these comparisons compliance with reporting requirements can be improved.

Observer data

Although total effort or catch cannot be determined from observer programmes, they provide accurate information on the location of fishing effort, total catch and catch composition of individual fishing trips and therefore allow for the assessment of bycatch and discards. Comparison with logbook data for individual vessels where corresponding data exists allows for an assessment of the accuracy of spatial information provided by skippers, as well as the landings and species composition thereof. Regular comparison of observer and logbook data for commercial sectors where observer programmes exist would allow for quantification of the accuracy of logbook submissions as well as the estimation of annual bycatch and discards (Walmsley *et al.* 2007a; Walmsley *et al.* 2007b).

The South African observer programme for commercial fisheries consists of two components. Onboard observers which are present on larger vessels which usually spend extended durations at sea (e.g. demersal trawl), and access point observers who monitor catches made by smaller vessels which are usually at sea for a day, or possibly overnight. Although access point observers cannot obtain accurate spatial information, they can provide information on the species composition of landings for comparative purposes. Observer programmes are, however, costly to run, and are often implemented on a national scale by the delegated authority and therefore involve multiple fisheries over wide geographical scales. This often leads to low monitoring frequencies of vessels on local scales, and a disproportionate spatial allocation of monitoring effort across each fishery. For effort within Algoa Bay only 2% and 5% of the fishing days were observed by onboard observers for the pelagic and demersal trawls respectively, while 21% of fishing boat-days were observed at access points in the linefishery. Local management initiatives may be able to contribute by improving local monitoring capacity, particularly where vessels are based locally and operate over smaller geographic scales from one port. Although this may be easily achieved on a local scale for the linefishery through the deployment of additional monitors at access points, it is more difficult for other sectors in which onboard observers are used and which operate on a national scale. However, certain vessels operate predominately in the Algoa Bay region and increased observation of these vessels through additional monitoring effort deployed locally would be beneficial. There is no observer programme for the chokka-squid fishery as it is a single species and effort regulated fishery. Improved monitoring of landings at access points would allow for improved verification of data obtained from logbooks while VMS data can be used to validate effort (days at sea) and spatial information.

VMS data

Currently VMS is used as a regulatory tool to monitor compliance within the existing MPAs in South Africa (Hutchings *et al.* 2009) and despite the data being recorded and stored by DAFF, it is largely inaccessible to research scientists due to confidentiality issues and has thus far not been used extensively for research purposes. All commercial fishery sectors operating in Algoa Bay are legally required to have VMS units fitted and operational prior to vessels exiting the port. VMS data provides highly accurate spatial data from which fisheries activities can be inferred and it therefore provides a valuable means for monitoring fine scale temporal patterns in fishing effort and activity (Campanis and Thompson 2007; Witt and Godley 2007; Gerritsen and Lordan 2011). Not only can VMS data be used to monitor future effort displacement following the proclamation of new no-take zones in Algoa Bay, but also to quantify the amount of sea-days and/or fishing-days occurring within Algoa Bay. Comparison of logbook and VMS data can be used to validate the spatial accuracy of catch return information and through engagement with rights holders may lead to improved reporting.

The proportion of fishing effort calculated from VMS/observer and logbook data in each broad fishing area identified per sector in Algoa Bay displayed a high degree of similarity for each commercial sector. However, the displacement of effort calculated using spatially accurate VMS/observer data or less accurate logbook data as a result of the proposed reserve designs identified in Chapter 7 differed considerably. This was due to the improved and finer spatial resolution of the VMS/observer data in comparison to the course spatial grid data from logbooks (Figure 7.9). The overall effort displacement for the commercial fishery calculated using the VMS/observer data was 12%, while that calculated using the logbook data was 21%. This represents a difference in the evaluation of the extent of the impact of a reserve design on the commercial fisheries of 9% between the two data sources. This highlights the importance of using VMS data to verify the accuracy of logbook information and encourage rights holders to improve their reporting accuracy.

Logbook, observer and VMS data therefore provide an acceptable standard for monitoring fisheries activities on a local scale and through integration of the three sources, data accuracy can be verified and improved in the future. Data acquisition agreements need to be established with the national management authority (DAFF) to obtain the data on an annual basis. In order to overcome confidentiality problems, agreements with the fishing industry may need to be made.

(ii) Recreational fisheries

The absence of spatially explicit recreational fisheries data in Algoa Bay required that a comprehensive baseline survey be undertaken (Chapter 5). This survey employed stratified aerial, roving and access point survey techniques to obtain the required spatial data as well as information on the catch, effort and socio-economics of the recreational sector. Similar approaches would be required in order to investigate future trends and obtain the required data for the indicators identified above. Power analyses were conducted in order to determine the level of sampling effort required in future monitoring surveys in order to obtain statistically robust results.

Methods

Analyses were conducted on angler and vessel count data in order to determine the number of surveys required for future monitoring of recreational shore and skiboat fisheries using stratified aerial, roving creel and access point surveys. This was based on the methods described in Willis *et al.* (2003) for Poisson count data with overdispersion using a GLM approach. The standardised number of anglers and an estimate of the dispersion parameter for each zone or access point were predicted using a Poisson GLM. The power and sample size to detect an effect size based on a percentage increase and decrease of the predicted mean was estimated using the following equation for a two-tailed test:

$$z_{\beta} = \frac{log(k)}{\sqrt{\frac{\phi}{n\mu_1}\frac{k+1}{k}}} - z_{\alpha/2} \qquad \text{Equation 8.1}$$

Where Z_{β} is the standard normal quantile, k is the effect size calculated as the ratio of the predicted and hypothesized means, ϕ is the measure of overdispersion in the Poisson model calculated as the deviance divided by the degrees of freedom, n is the observed sample size from the annual survey, μ_1 is the lower hypothesised mean calculated incrementally as a percentage of the predicted mean, and $Z_{\alpha/2}$ is the z-value corresponding to a significance level of 0.05 for a two-tailed test.

The power was calculated for a range of effect sizes based on incremental changes to the predicted mean for each study area. In order to calculate the sample size required to detect future changes at a range of effect sizes the formula was rearranged to make n the subject of the formula.

Results and discussion

Unlike commercial sectors no national monitoring strategy is currently in place for assessing recreational fishing activities in South Africa. Dedicated fishery-independent surveys are therefore required to obtain information on the catch and effort for both shore (recreational and subsistence) and recreational skiboat fisheries. Indicators selected for monitoring the recreational shore (inclusive of subsistence sector) and skiboat fisheries include estimates of angler density and annual participation (angler number and launching effort) as well as catch composition and the mean size of target species.

Designing a monitoring programme for the shore fishery is complicated by the diffuse nature of the fishery, the multiple points through which the public can access the shoreline, and the distances between access points in large study areas. Roving creel surveys are the most effective means for obtaining catch and effort information with good spatial accuracy in diffuse fisheries with multiple access points (Pollock *et al.* 1994). Nonetheless they are labour intensive as only relatively short distances can be traversed at one time, therefore requiring numerous surveys to be conducted to cover large study areas. In addition they can be costly to implement particularly if ORVs are required to access remote areas and traverse long stretches of shorelines between access points. Power analysis of this studies roving creel data indicated that an unrealistically large number of surveys were required per zone (ranging between 135 and 375 surveys/zone/year; Figure 8.2) to determine inter-annual differences in angler number as a result of the high spatial and temporal variability observed in the baseline survey.

Although perceived to be costly, aerial surveys provide a cost-effective means to cover large geographical areas which could take several days to traverse using roving creel surveys (Pollock *et al.* 1994). They have been shown to be highly effective for obtaining accurate spatial information on various shore and nearshore based activities and for quantifying effort in large study areas (Pollock *et al.* 1994; Smallwood 2009; Smallwood *et al.* 2011). Shore angler count data for the whole of the Algoa Bay coastline obtained from aerial surveys revealed that approximately 29, 63 and 124 aerial surveys would be required per annum to detect 50%, 30% and 20% changes in the number of shore anglers between successive years respectively (alpha 0.05, power 0.8). Aerial surveys are therefore far more effective for monitoring changes in total effort within Algoa Bay. A randomly stratified aerial survey design with a sufficient number of flights to be statistically robust would therefore allow for accurate monitoring of angler density per survey zone and improve estimates of annual shore angler number within Algoa Bay, both of which are selected indicators of pressure for the recreational shore fishery. They would also allow for accurate and easy long-term comparison of temporal trends in the spatial distribution of nearshore boat-based fishing effort for the commercial and recreational sectors.

Although a halving or doubling in the number of anglers is a considerable change in effort, it is unlikely that more than 29 aerial surveys can be implemented per year due to the associated financial costs. Furthermore as the quantity of data improves over multiple years, directional responses in angler number will become apparent and statistical power will increase. It is therefore recommended that two aerial surveys be conducted per month (24 in total) appropriately stratified between week and weekend days with additional survey days conducted during peak holiday season in March/April and December/January (additional 4 days). Although stratification by time of day would be optimal, strong winds often preclude flights later in the day and visibility may deteriorate, reducing the accuracy of counts. It is therefore recommended that aerial survey be conducted only on days when weather conditions are suitable and during the morning period when visibility is likely to be greatest. Although this may lead to an overestimation of fishing effort (Pollock *et al.* 1994), coupling aerial surveys with on-site roving creel surveys will allow daily temporal patterns in angler number to be determined. Furthermore roving creel surveys will allow for trends in abundance over varying weather conditions to be determined which can then be used to calibrate estimations of annual shore fishing effort from aerial surveys. This form of aerial-roving angler survey design is commonly used in recreational fishery assessments where information on both the angler effort and catch is required (Pollock *et al.* 1994). As the primary effort data will be obtained from aerial surveys, roving creel survey effort can be targeted at days when the interception of anglers is likely to be highest over weekends and holiday periods thereby improving the efficiency of the surveys to obtain catch information (Pollock *et al.* 1994). Roving creel surveys must still, however, be stratified across season and period of day to obtain catch information which reflects temporal trends. Regular on-site engagement with anglers and dissemination of information during on-site interviews will also contribute to improving awareness of and compliance with regulations in the long-term.

The baseline survey of the recreational skiboat effort (Chapter 5) made use of randomly stratified effort counts (counts of trailers) at launch sites where no launch records were maintained (Kenton and

Boknes), while catch interviews were conducted during high use periods (weekends and holidays) when data acquisition on species composition and spatial information could be maximised. Conducting effort counts at access points was time consuming and costly but was required to estimate annual launching effort. Since the imposition of the ban of ORVs in the coastal zone, local authorities of skiboat clubs have been required to apply for permits to operate vessel launch sites, and became responsible for management of launch site activities. Although little formal management currently occurs at the beach launch sites in Algoa Bay, the management bodies should be encouraged to initiate compulsory launch registers. A logbook system is currently in place at the PEDSAC launch site in the Port Elizabeth harbour. A similar system has been widely implemented along the KwaZulu-Natal coastline of South Africa (Pradervand 2006) and provides accurate information on launching effort. A wider use of this logbook system in Algoa Bay could provide more accurate data on the number of vessels launching per year at each site and reduce the need for frequent effort counts. On-site monitoring can then be concentrated on high use periods in order to obtain information on catch composition, as well as the spatial locations of fishing activities which could be supplemented by aerial surveys.

As the beach launch sites are unmanned, completion of logbooks is likely to be poor initially. Compliance with the launch site logbook system can be easily checked by random inspections at each access point because tow vehicles and trailers are parked in close proximity to the launch site and registration numbers can be compared to the logbook entries. This would require far less effort than undertaking regular effort counts, and issuing fines or penalties would rapidly improve compliance. A well implemented and managed logbook system will improve the accuracy of the data for launching effort. Provided that a logbook system is implemented and compliance with this system enforced, access point surveys can be stratified across seasons and focused during high use periods to obtain information on the catch composition and length frequency of targeted species.

(iii) Illegal, unregulated and unreported (IUU) fisheries

No data on IUU fisheries were assessed in this study yet data exists through DAFF and SANParks enforcement programmes. Data on the number of occurrences, types and locations of illegal fisheries activities in Algoa Bay must be compiled from public hotline responses, DAFF and SANParks enforcement patrol records. No regular monitoring activity can be conducted and this indicator is based on opportunistic information. Accurate records should, however, be maintained by regulatory authorities and data shared between management agencies to monitor and report on the number and types of illegal incidents occurring per month.

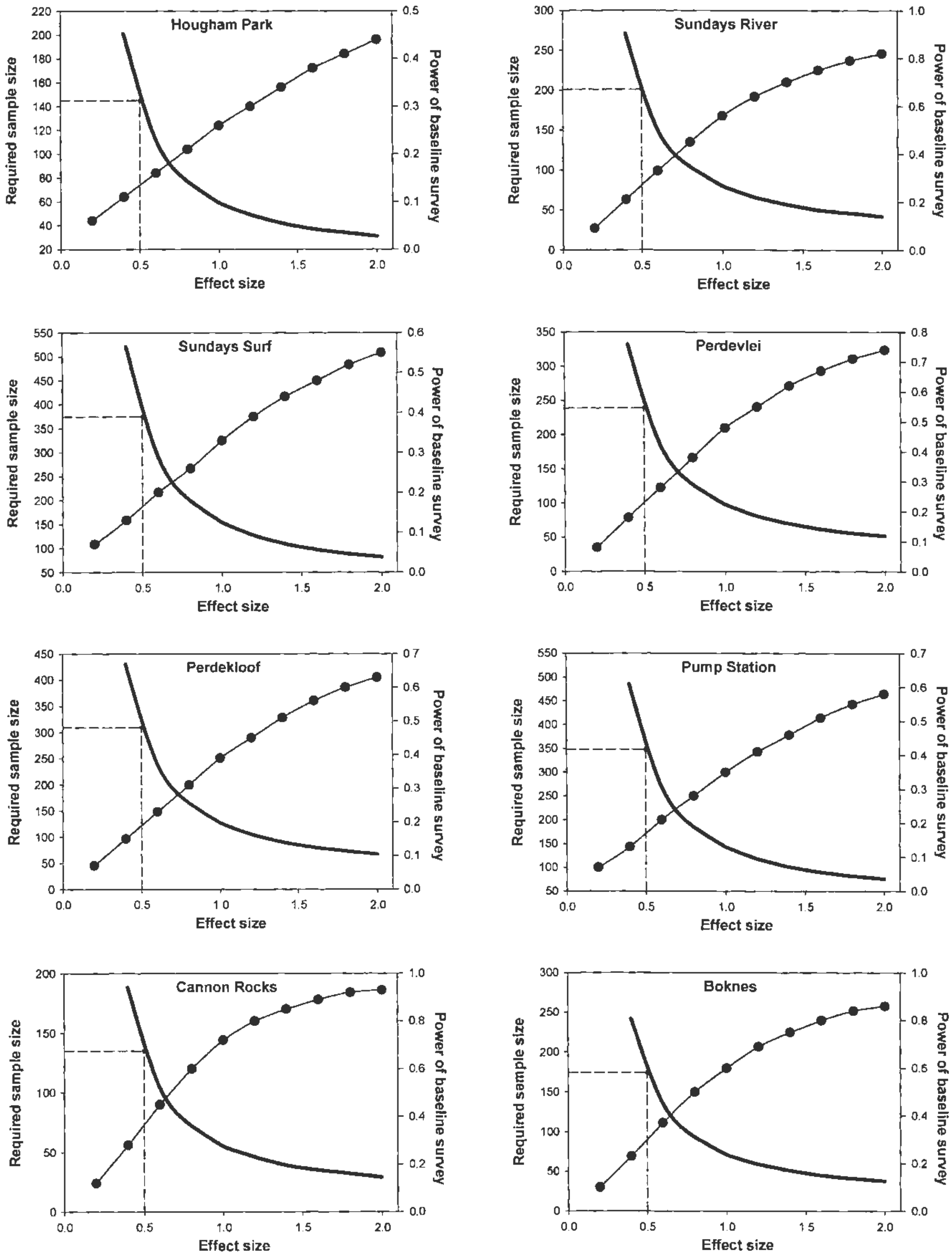


Figure 8.2. Power (closed circles) of the current survey design to detect changes in angler number per access point, and the required sample sizes (solid thick line) to detect a 50% change in number at a power of 0.8. Note differing scales on the y-axis.

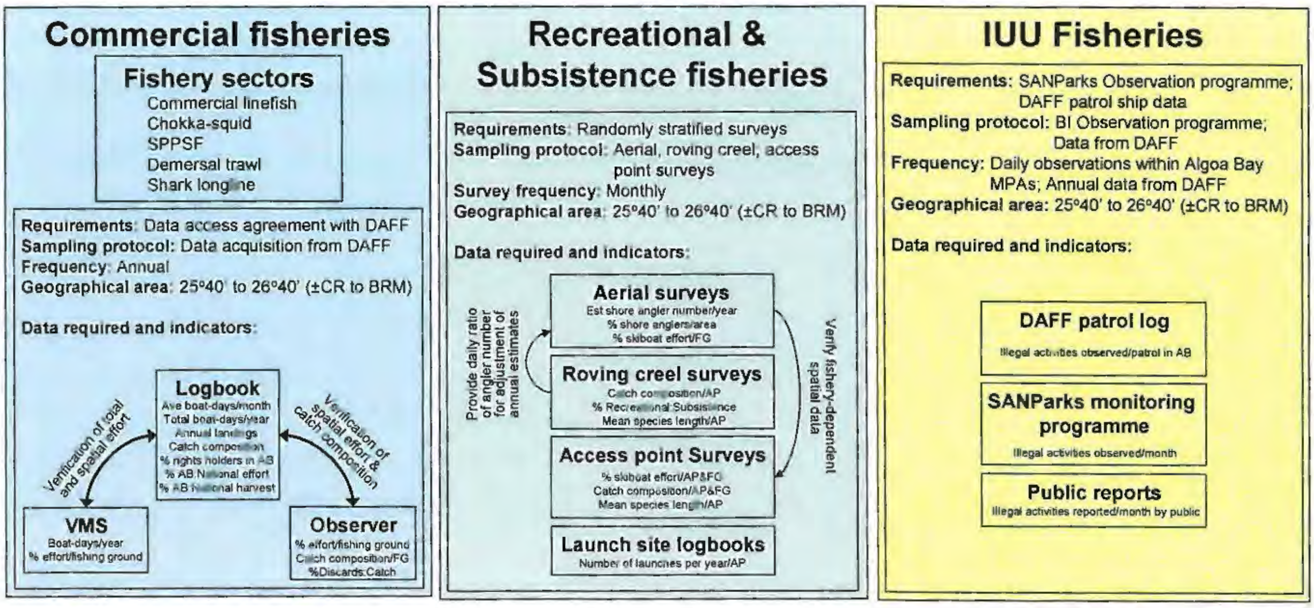


Figure 8.3. Proposed survey design for monitoring fishery pressures within Algoa Bay.

(b) State

Temporal trends in indicators of state often have high variability and identifying a directional signal as a result of specific pressures over and above the noise generated by a multitude of environmental factors requires high statistical power. It is therefore important that monitoring programmes are designed to take natural variability into account by ensuring that sample sizes are sufficiently large to achieve adequate statistical power for identifying responses which are biologically meaningful. Increasing sample size reduces variability; however, a trade-off exists between the amount of data which can be collected and the cost of acquiring the data. In order to design a cost-effective monitoring programme which is feasible to implement, a balance between sample size, statistical power and the size of the effect to be detected needs to be reached within certain cost, logistical and weather constraints.

Although fishery dependent abundance data are readily available from the national regulatory authorities, it is not suited for monitoring trends in the state of resources. This data lacks appropriate stratification, is influenced by differences in fishing gear and techniques between vessels, and is also influenced by fluctuating market demands which alter prices leading to preferential targeting of certain species. Trends in resource state observed through the analysis of logbook data may therefore reflect regulatory changes or technological advances in the fishery rather than the true status of the resources. Furthermore, only economically valuable species are retained and no information on trends in bycatch species, or other non-targeted species in the community are obtained. In addition, fishery dependent data cannot be used for comparing MPAs to exploited areas due to the exclusion of extractive resource use in no-take zones (Murphy and Jenkins 2010). Monitoring the state of resources within no-take MPAs is important for evaluating long-term trends in biological communities in response to environmental drivers in the absence of anthropogenic pressures, and standardised methods need to be employed within and outside of MPAs in order to be comparable. Fishery-independent monitoring in MPAs allows for the effects of exploitation on species abundance in unprotected sites to be quantified, as well as the degree of response to protection.

Randomly stratified fisheries independent surveys using standardised gear are therefore most appropriate for assessing trends in the state of fishery resources. Such dedicated surveys, however, require considerable investment of financial resources and skilled manpower and a trade-off between cost-efficiency and statistical power (sample size) needs to be achieved. This thesis focused on obtaining baseline information on the reef and demersal fish resources as they are most heavily targeted in Algoa Bay. However, monitoring the trends in the state of habitats utilised by fish communities, and the small pelagic and chokka-squid fishery stocks is also required. All aspects are discussed briefly below.

(i) Habitat quality

Habitat quality and complexity plays an important role for fish communities and therefore requires monitoring. Monitoring shoreline habitats can be conducted via remote sensing to evaluate temporal trends in large scale sediment movement (Mason *et al.* 2010) or via monitoring beach elevation to determine patterns of accretion or erosion (Quartel *et al.* 2008). In the offshore environment habitat maps can be prepared and periodically evaluated using sidescan sonar or acoustic ground discrimination systems (Flemming 1980; Foster-Smith and Sotheran 2003; Wilding *et al.* 2003; Brown *et al.* 2005). Water quality is an important aspect influencing habitat quality (Pomeroy *et al.* 2004) requiring dedicated monitoring. Dedicated programmes currently exist for monitoring point source discharges into the marine environment as well as selected parameters at recreational beaches. In addition plankton research and monitoring programmes currently undertaken in Algoa Bay include some aspects of water quality at selected sites. These programmes need to be assessed in order to identify current gaps, and ensure that a programme is designed which ensures water quality monitoring is undertaken in a holistic manner throughout the bay and includes a range of parameters which are indicative of potential issues of concern.

The community structure of macro-benthic invertebrate communities should also be monitored using appropriate methods such as point intercept methods on scuba (Watson and Barnes 2004), using photoquadrats (Preskitt *et al.* 2004), jump cameras (Roberts *et al.* 1994; Smale *et al.* 2010) or remotely operated vehicles (ROVs) (Parry *et al.* 2003). These aspects were not investigated in this thesis and future research in Algoa Bay needs to focus on identifying the most appropriate methods for monitoring habitat related changes and benthic communities in both reef and non-reef areas, and develop protocols which can be incorporated into the overall monitoring framework.

Similarly changes in the gut content of selected indicator species are important for assessing changes in the abundance of food items locally. No programmes currently exist and future research projects should aim to include aspects of feeding ecology and gut content analysis for key fishery species which can be incorporated as a component in the overall monitoring programme.

(ii) Reef fish communities

Methods

In order to evaluate the performance of potential indicator species the variability in abundance and size estimates was evaluated using the coefficient of variation, which is calculated as follows:

$$CV = [SD(x_i)]/\bar{x} \qquad \text{Equation 8.2}$$

Where CV is the coefficient of variation, *SD* is the standard deviation of the sample and $\bar{x}$ is the estimated mean of the population.

The power to accurately detect trends in indicators of state is often poor (Jennings 2005) due to high levels of natural variability within populations. Baseline survey data on the relative abundances of indicator species were used to determine the power and sample size requirements for future surveys based on the methods described in (Willis *et al.* 2003) for Poisson count data. Standardised relative abundances of indicator species for each study area were predicted using a Poisson GLM and an estimate of dispersion. The power and sample size to detect an effect size based on a percentage increase and decrease of the predicted mean abundance was estimated using the following equation for a two-tailed test:

$$z_\beta = \frac{log(k)}{\sqrt{\frac{\phi}{n\mu_1}\frac{k+1}{k}}} - z_{\alpha/2} \qquad \text{Equation 8.3}$$

Where Z_β is the standard normal quantile, *k* is the effect size calculated as the ratio of the predicted and hypothesised means, ϕ is the measure of overdispersion in the Poisson model calculated as the deviance divided by the degrees of freedom, *n* is the observed sample size from the annual survey, μ_1 is the lower hypothesised mean calculated incrementally as a percentage of the predicted mean, and $Z_{\alpha/2}$ is the z-value corresponding to a significance level of 0.05 for a two-tailed test.

The power was calculated for a range of effect sizes based on incremental changes to the predicted mean for each study area and curves were plotted. In order to calculate the sample size required to detect future changes at a range of effect sizes the formula was rearranged to make *n* the subject of the formula.

Similarly the data from baseline surveys were used to determine the power and sample size requirements for detecting significant differences in the mean length of indicator species. The power of the baseline surveys was determined using the following equation for a one sample two-tailed test (Rosner 1995):

$$1-\beta = \Phi\left[z_{\alpha/2} + \frac{|\mu_1 - \mu_2| \times \sqrt{n}}{\sigma}\right] \qquad \text{Equation 8.4}$$

Where 1-β is the estimated power, $Z_{\alpha/2}$ is the Z value for a two-tailed test corresponding to an α of 0.05, Φ is the underlying normal distribution, μ_1 and μ_2 are the observed and hypothesised mean lengths respectively, *n* is the sample size of the current study and σ the standard deviation.

The required sample size to detect significant differences in mean length in future monitoring was estimated with the following equation for a two-tailed test (Rosner 1995):

$$n = \frac{\sigma^2 (z_{1-\beta} + z_{1-\alpha/2})^2}{(\mu_1 - \mu_2)^2} \qquad \text{Equation 8.5}$$

Where n is the required sample size, $Z_{1-\alpha/2}$ is the Z value for a two-tailed test corresponding to an α of 0.05, $1-\beta$ is the desired power, μ_1 and μ_2 are the observed and hypothesised mean lengths respectively, and σ the standard deviation.

Results and discussion

Community type and selection of study sites

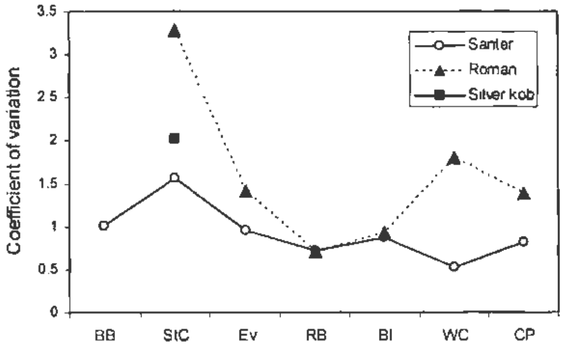
Two reef fish community types were distinguished in Algoa Bay by means of multivariate statistics (Chapter 3). Both communities are dominated by important commercial and recreational fishery species and are therefore subject to high levels of pressure. Monitoring trends in community structure and key parameters of selected indicator species from both community types is therefore required for evaluating future progress towards protection of reef fish communities in Algoa Bay. Evaluating changes in the abundance and size of key species is one of the most widely used indicators of management success (Pomeroy *et al.* 2004). The proposed no-take zones for the AENP MPA present opportunities for monitoring both protected and unprotected reef fish populations from both community types.

Selection and evaluation of indicator species

Indicator species representative of the linefish communities in Algoa Bay are required for long-term monitoring in Algoa Bay. Both reef linefish communities identified in Algoa Bay were dominated by santer, which would serve as a good indicator species as it is well represented in all reef areas in Algoa Bay. Furthermore it has also been well studied locally with research conducted on its general biology (Coetzee 1978), age, growth and diet (Coetzee and Baird 1981a), reproductive biology (Coetzee 1983) and exploitation (Coetzee and Baird 1981b; Smale and Buxton 1985; Hecht and Tilney 1989; Brouwer and Buxton 2002). Tagging conducted during this research project also indicated that santer are highly resident, with individuals having been recaptured within close proximity to the tagging site after extended periods at liberty (number of recaptures=11; Mean distance moved 243±297m; mean days at liberty 250±183; maximum time at liberty 674 days with a distance of 119m moved). Santer is heavily targeted by the commercial and recreational skiboat fisheries within Algoa Bay and changes in abundance and mean length will therefore serve as early warning indicators of a declining state of linefish stocks as a result of fishing pressure. The CV for santer CPUE from controlled angling surveys ranged from 0.5 to 1.6 in the Woody Cape (WC) and St Croix (StC) areas respectively. However, the CV in all Group 2 reef areas was below 1 (Figure 8.4a). The CV for santer length was much lower, ranging from 0.13 to 0.19 in the Riy Banks (RB) and Bell Buoy (BB) areas respectively (Figure 8.4b). In addition the CV from UVC surveys in the Bird Island (BI) MPA was 1.45. These lower CV values for relative abundance and size indicate that santer would be a good indicator species for reef fish (Group 1 and 2) communities in selected sites in Algoa Bay.

Roman was selected as an indicator species in the Tsitsikamma and Plettenberg Bay areas as it was the most dominant and abundant linefish in the area and illustrated low variability in abundance and length estimates (Bennett 2007). Furthermore, roman has several desirable characteristics having been the focus of much research (Buxton 1984; Buxton 1987; Buxton and Smale 1989; Götz *et al.* 2008); being heavily targeted and sensitive to fisheries pressure, as well as being highly resident (Kerwath *et al.* 2007a; Kerwath *et al.* 2007b). Despite not being as dominant in Algoa Bay as on the south coast, roman would also serve as a good indicator species in the Group 2 community reef areas due to its high abundance (0.9 and 0.7 respectively), and low CVs, particularly in the Bird Island (BI) and Riy Banks (RB) study areas. It would also be a suitable indicator species for comparison between protected (BI) and exploited (RB) areas within Algoa Bay, as well as allowing for broader regional comparisons between other MPAs within the bioregion where santer abundance is low. The inclusion of roman as an indicator species and the selection of monitoring sites in Algoa Bay where it is abundant would therefore contribute to regional monitoring initiatives allowing for comparisons between Table Mountain National Park, Goukamma MPA, Tsitsikamma National Park and AENP. The monitoring design should take into account the sample sizes required to monitor roman in the RB and BI areas in Algoa Bay where variability in the estimates of abundance was lowest. Although insufficient UVC surveys were conducted in the RB area, the CV of roman abundance was low in BI (0.84) confirming its suitability for both, controlled angling and UVC surveys.

Silver kob is an important linefishery species which is heavily targeted and stocks are considered collapsed (Griffiths 1996a; Griffiths 1997b). Important aggregation and nursery areas have been identified around the StC area in Algoa Bay in this and previous studies (Smale 1984; Wallace *et al.* 1984a). However, it is migratory and uncommon in most reef areas in Algoa Bay and therefore not a good candidate indicator species (Hilty and Merenlender 2000). Due to high spatial and temporal variability in abundance estimates in the StC area power analysis indicated that the required sample sizes to monitor silver kob abundance are extremely high and monitoring would therefore not be cost-effective. Nonetheless due to the importance of the StC area for juvenile silver kob, controlled angling should still be conducted in this area to collect long-term data which will improve the understanding of the use of this aggregation area. In addition, the use of alternative methods such as acoustic surveys, tagging and telemetry, ROVs or BRUVs should be investigated to complement controlled angling surveys.



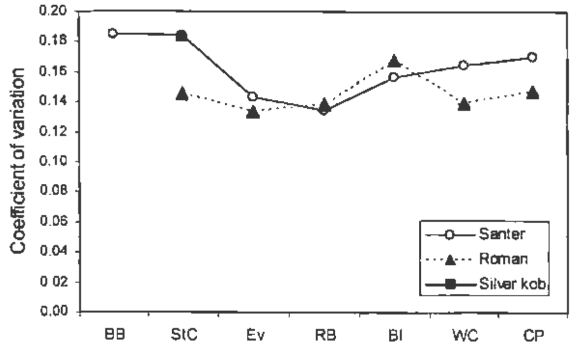


Figure 8.4. Variability in (a) CPUE, and (b) length of thee candidate indicator species within each study area from controlled angling surveys.

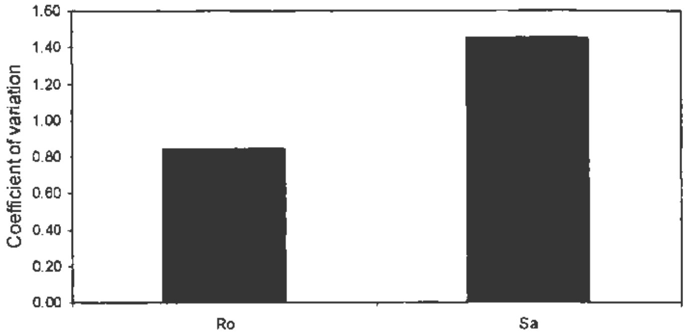


Figure 8.5. Variability in relative abundance of roman and santer observed within the BI area during UVC.

Selection of survey methods

Diving surveys within the BI area allowed for the assessment of a greater range of species than the controlled angling survey technique. Roman has the lowest variability around the mean abundance, followed by fransmadam and santer, suggesting that these species would all be good indicator species. Fransmadam is of limited socio-economic value and is not targeted directly by the linefishery making it less important as an indicator species. Comparison of the CV between the two survey techniques used in the BI area indicated that controlled angling provided less variable estimates of santer abundance but high variability in roman abundance in comparison to UVC (Table 8.4). This indicates that both methods are suitable for monitoring but the sensitivity is dependent on the species. Future monitoring of reef linefish communities in Algoa Bay should therefore employ both survey technique in a complementary manner.

Table 8.4. Comparison of the precision of UVC and controlled angling survey methods in the BI area using the coefficient of variation (the sample size of controlled angling was reduced to equal that of the UVC survey by randomly selecting 79 samples).

	Santer	Roman
UVC	1.45	0.84
Controlled angling	0.92	1.06

UVC has been widely and successfully used to monitor fish populations (Claudet *et al.* 2006; Mann *et al.* 2006; Bennett *et al.* 2009; De Raedemaecker *et al.* 2010), particularly in MPAs where non-destructive techniques are required. However, controlled angling has been shown to have a low impact with low mortality rates in previous studies in MPAs in South Africa (Götz 2005) provided that appropriate training and precautions are implemented (barbless hooks, gas bladder deflation etc.).

Controlled angling is also beneficial as it allows for accurate measurement of fish length whereas divers estimate fish size during UVC. Although UVC has the advantage of *in situ* assessment of habitat characteristics and the observation of the entire fish community, including non-predatory fish species not sampled by angling, practical implementation was difficult in the current study. This was due to both poor weather conditions (mainly poor water visibility in most study areas) and the legal requirements for conducting research diving in South African. Scientific diving is governed by regulations under the Occupational Health and Safety Act (No 85 of 1993) in South Africa. This legislation requires that all divers are appropriately trained to the level of Class IV Scientific Diver (1 month training) and there must be at least one Class IV Supervisor on site at all times (additional training and experience required). Furthermore it requires that a minimum of four qualified members are present in the dive team at any one time. UVC surveys therefore require a highly skilled team of scientists using specialised equipment in order to conduct the research. Historically few researches have received the appropriate dive training in South Africa, placing a heavy reliance on using professional commercial divers in order to meet the legal requirements. This places severe financial and logistical constraints on conducting research diving. In comparison, controlled angling requires basic field equipment and can be conducted by volunteers who can be easily trained and supervised on site, thereby only requiring one trained scientist to be present. Controlled angling is also less affected by environmental conditions, particularly water visibility, and is financially and practically more easily to implement. Each technique therefore has its benefits and limitations and a complementary approach needs to be employed based on the overall objectives of the monitoring programme.

Power and sample size

Power analysis has been advocated as a useful tool for the design and planning of research projects as well as for the interpretation of results (Fairweather 1991; Steidl and Hayes 1997). The use of power analysis to calculate sample size requirements for detection of predetermined effect sizes in ecological studies has not been common, resulting in studies with weak statistical power, potentially leading to incorrect interpretation of results and false conclusions being made (Peterman 1990). In order to determine an appropriate sample size for future studies the desired effect size (difference between populations), an estimate of the population variance as well as the alpha (usually 0.05) level and desired power (1-beta, usually 80%) are required. In ecological studies it is important that the effect size is based on meaningful biological differences; however these are not easily determined and are largely based on expert judgment and past research. Edgar and Barrett (1997) suggested that the detection of a doubling (100% change) in relative abundance was appropriate for monitoring fish using UVC. In areas where populations are heavily exploited this could result in extreme levels of overexploitation and a reduction in abundance and biomass of key fishery species prior to the impact being detected. Considering the poor status of reef linefish in South Africa, a detection of 50% change in relative abundance was considered the largest possible effect size that would be appropriate for future monitoring. Changes in the size of organisms are far easier to detect than changes in abundance due to the inherent natural variability of population abundance. A 10% change was therefore suggested as an appropriate measure of changes in length of indicator species (Edgar and Barrett 1997); however, a reduction of 5% was considered feasible in this study.

Measuring ecological responses as a result of human pressures or changes in management is difficult and requires carefully planned and designed studies. Several studies have documented the effects of fishing on fish populations, which include decreased mean length of fish (Buxton and Smale 1989; Cowley *et al.* 2001; Yemane *et al.* 2004), lower fish abundance (Buxton and Smale 1989; Bennett and Attwood 1991; Cowley *et al.* 2001; Götz 2005) and changes in fish communities (Ruttenberg 2001; Götz *et al.* 2009b), and a clear causal relationship between fishing pressure and the response of fish populations has been established (Mosquera *et al.* 2000; Lester *et al.* 2009). As a result, changes in abundance and size of indicator species, and changes in community structure, can be used to evaluate the effects of fishing on reef linefish communities where exploited and control (no-take) sites are present, or alternatively baseline data are available against which future changes can be evaluated in response to management intervention. Monitoring long-term temporal trends in indicator species in no-take areas also allows for the evaluation of indirect environmental or anthropogenic pressures such as climate change, diffuse pollution sources or increased development and shipping activity on reef linefish communities.

Controlled angling

The baseline survey design in the BI (n=145 angling sites) and WC (n=56 angling sites) areas had good power to detect future changes in santer abundance. An estimated 79 and 30 angling sites are required in each area respectively to detect a future 50% change in santer abundance (0.8 power; 5% alpha level) (Figure 8.6). The power of the baseline survey design in the CP (n=62 angling sites) and RB (n=40 angling sites) areas was lower and estimated sample sizes of 80 and 65 sites respectively are required for detecting future halving or doubling in santer abundance during monitoring surveys. The power of the baseline survey design in the Ev (n=15 angling sites), BB (n=13 angling sites) and StC (n=28 angling sites) areas was low due to the small sample sizes and lower catch rates and future monitoring effort of 48, 52 and 119 angling sites would be required, respectively.

The power of the baseline survey design to detect changes in roman abundance was generally lower than that for santer (Figure 8.5) due to the lower catch rates and greater dispersion in each survey area. Sample sizes of 115, 216, 195, 52 and 111 angling sites are required at BI, WC, CP, RB and Ev, respectively, in order to detect 50% changes in roman abundance during future monitoring surveys (0.8 power; 5% alpha level) (Figure 8.7).

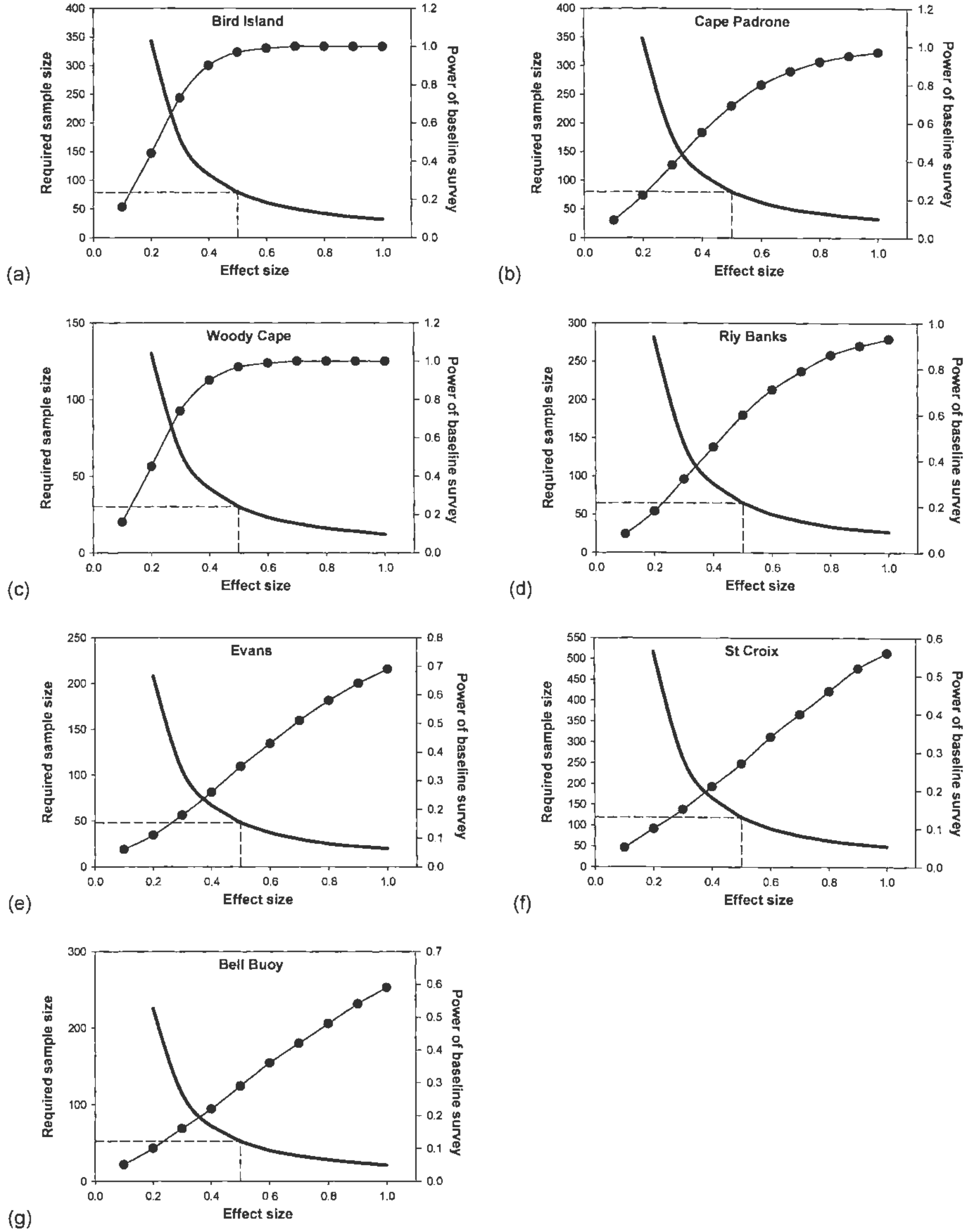


Figure 8.6. Sample size required to detect changes in abundance of santer using controlled angling at a power of 0.8 and significance level of 0.05 (thick line), and power of the baseline survey (solid circles) (a) BI, (b) CP, (c) WC, (d) RB, (e) Ev, (f) StC and (g) BB. Drop lines indicate the effect size detectable at a power of 0.8 and significance level of 0.05.

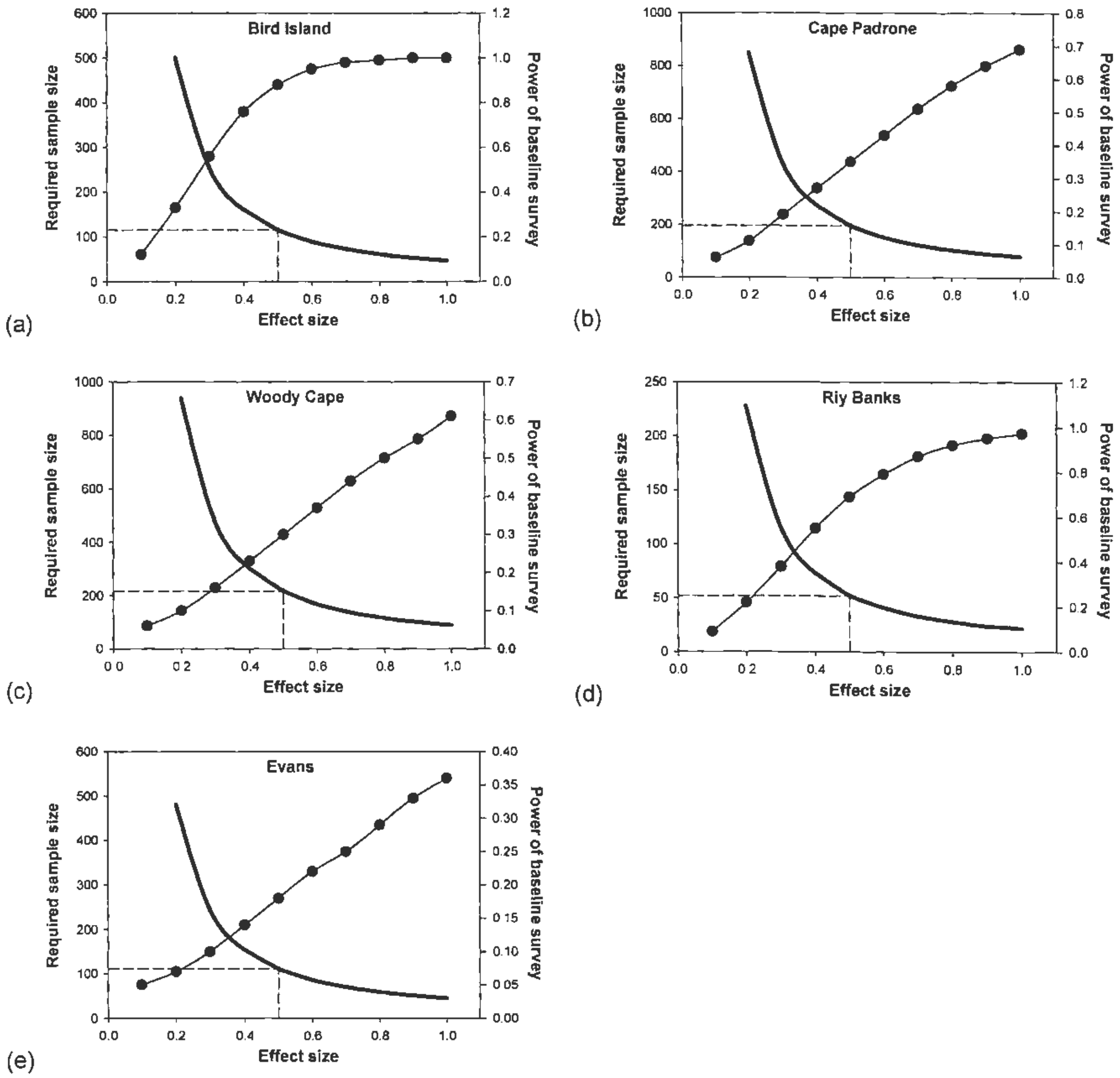


Figure 8.7. Sample size required to detect changes in abundance of roman using controlled angling at a power of 0.8 and significance level of 0.05 (thick line), and power of the baseline survey (solid circles) (a) BI, (b) CP, (c) WC, (d) RB and (e) Ev. Drop lines indicate the effect size detectable at a power of 0.8 and significance level of 0.05.

Sample sizes in the baseline survey were sufficient in all areas to detect 7% change in mean length of santer (0.8 power; 5% alpha level) (Figure 8.8). Similarly in Group 2 areas, with the exception of Ev, all sites had sufficient power to detect a 6% change in roman length (Figure 8.9). Future sample sizes (number of fish captured) required to detect a 5% change in length of santer ranged from 108 at BB to 57 at RB (Figure 8.8), and from 77 at BI and 60 at RB for roman (Figure 8.9). Based on the average CPUE for each species this would require between 15 and 28, and 17 and 20 stations for santer and roman respectively.

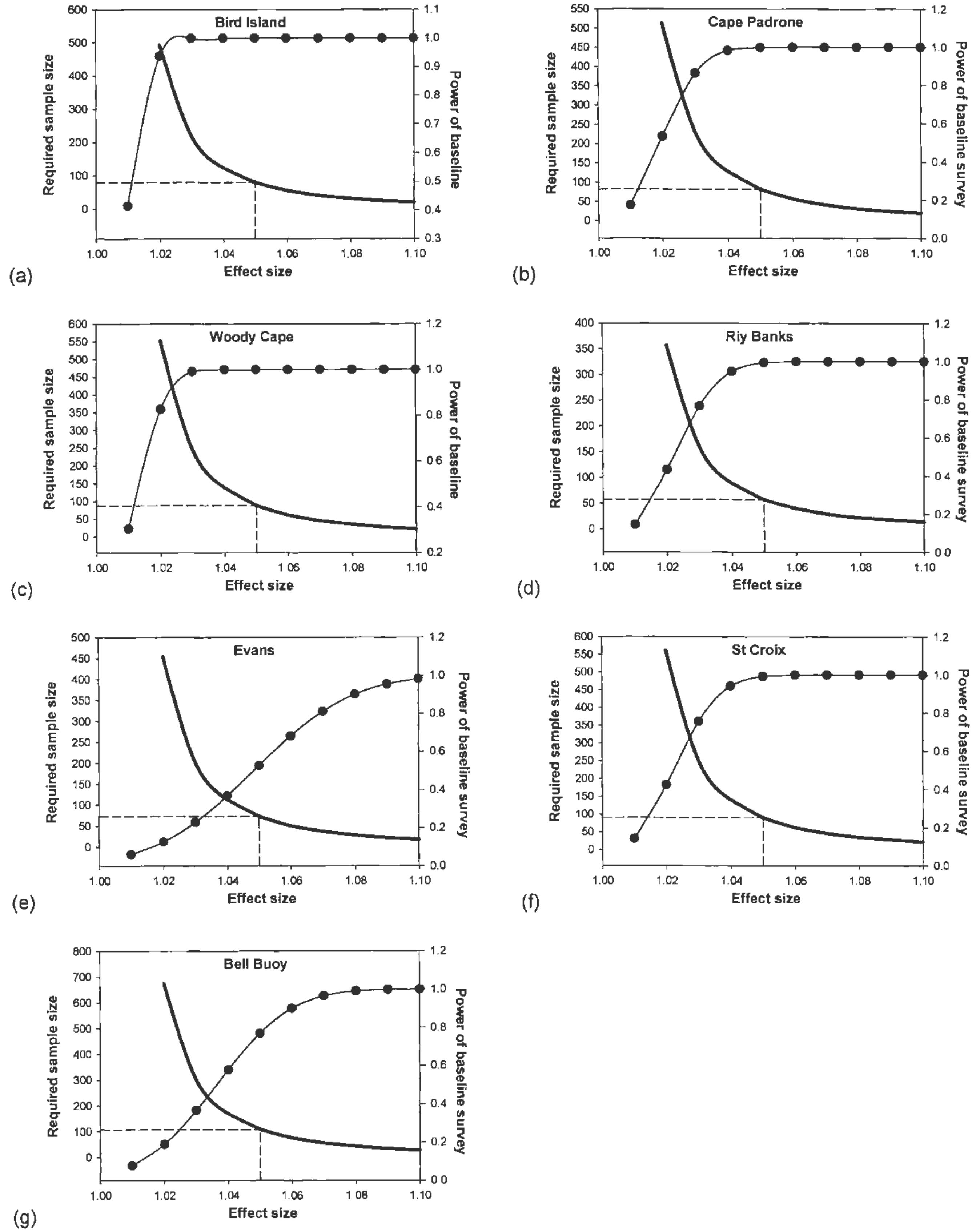


Figure 8.8. Sample size required to detect changes in mean length of santer using controlled angling at a power of 0.8 and significance level of 0.05 (thick line), power of the baseline survey (solid circles) (a) BI, (b) CP, (c) WC, (d) RB, (e) Ev, (f) StC and (g) BB. Drop lines indicate the effect size detectable at a power of 0.8 and significance level of 0.05.

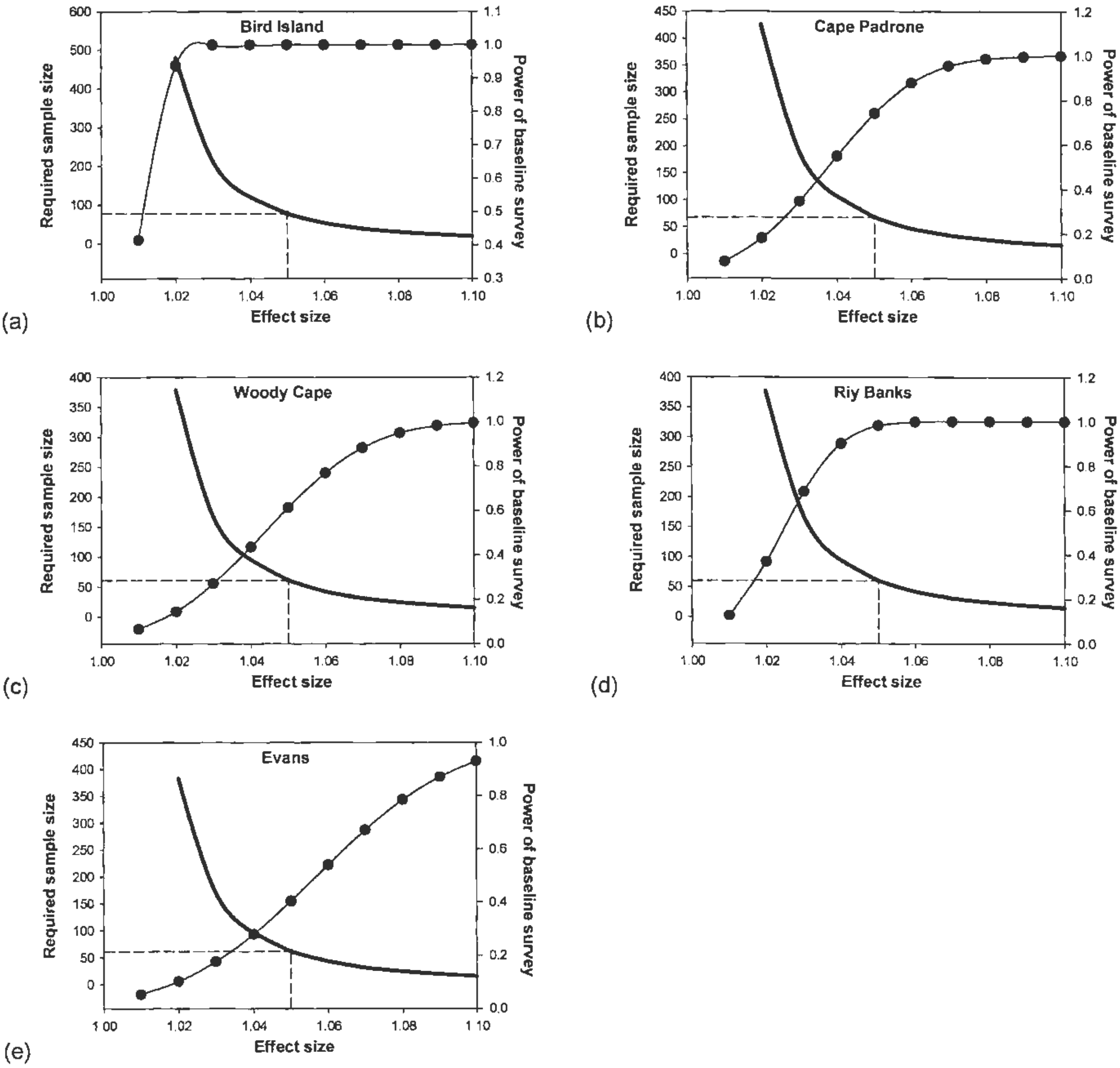


Figure 8.9. Sample size required to detect changes in mean length of roman using controlled angling at a power of 0.8 and significance level of 0.05 (thick line), power of the baseline survey (solid circles) (a) BI, (b) CP, (c) WC, (d) RB and (e) Ev. Drop lines indicate the effect size detectable at a power of 0.8 and significance level of 0.05.

UVC surveys

The baseline UVC survey of 80 dives in the BI area had sufficient power to detect a 50% change in abundance of roman (0.84) but not santer (0.4) (Figure 8.10). An estimated 72 and 218 dives are required in order to detect a 50% change in relative abundance for each species respectively. Both 10% and 5% changes in roman lengths estimated during the UVC baseline survey in the BI area could be detected with high power. An estimated sample size of 350 and 88 fish would need to be observed to detect a 5% and 10% change in roman length, respectively. This would entail between 32 and 126 dives to be conducted based on the average abundance values of roman.

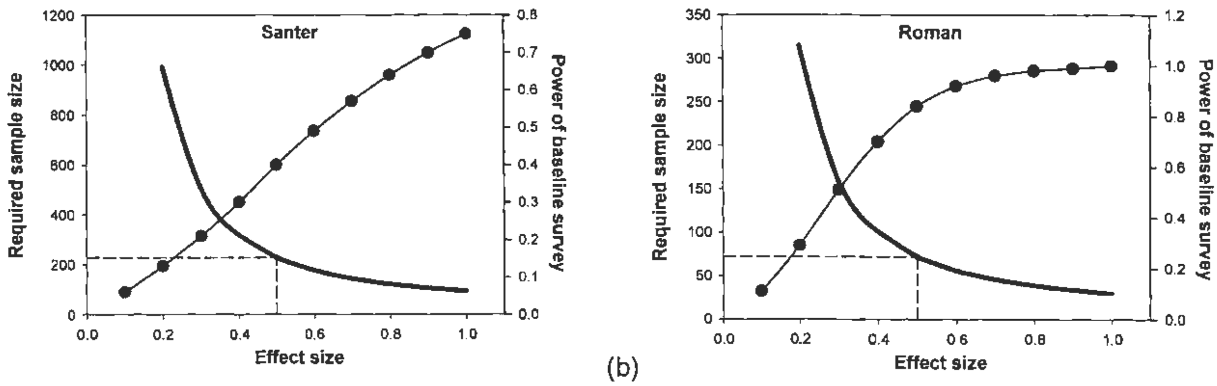


Figure 8.10. Power of the baseline survey (closed circles) and required sample size (solid line) to detect changes in abundance at a power of 0.8 and significance level of 0.05 with UVC, (a) santer and (b) roman. Drop line indicate the effect size detectable at a power of 0.8 and significance level of 0.05.

Survey design

Controlled angling

Sample size determination taking into account the variability in catch rates within each area using power analysis suggests that the number of angling sites required for detecting a 50% change in relative abundance of santer ranged between 30 and 119 per area per annum. Considering travel distances to access the different study areas (up to 45km one way for some areas) an average of ten angling sites can be completed per working day. This means that between three and 12 sampling days are required per annum per area to monitor changes in santer abundance, which translates into two to six days per season (Table 8.5).

Monitoring roman abundance in the Group 2 communities (see Chapter 3; insufficient roman in Group 1 communities) requires between 52 and 216 sites, which translates to between six and 22 sampling days per area per annum (Table 8.3). This is a greater sampling effort than is required to monitor santer abundance and is due to the greater variability in catches and the lower catch rates. Monitoring roman abundance is, however, important as it will allow spatial comparisons with other monitoring programmes in the Agulhas Bioregion (e.g. Tsitsikamma National Park conducted by SAEON). Future monitoring programmes in Algoa Bay should therefore include sufficient sampling to ensure roman can be used as an indicator species with statistical robustness in at least one protected and one exploited site. The Riy Banks (RB) and Bird Island (BI) areas would be most appropriate due to the higher catch rates for roman and similar habitat characteristics between these two areas. Furthermore, diving conditions in these areas are better than those on inshore reef areas and would allow the use of complementary survey techniques.

Detecting changes in mean size is more sensitive than detecting changes in mean abundance of a species. Edgar and Barrett (1997) suggest that a 10% change in the mean size of an indicator species is sufficient to detect responses to pressures on an ecosystem. In order to detect this level of change, between 14 and 27 santer and 15 and 19 roman are required from each area. Taking the average CPUE for each species into account in each area this translates to between 3 and 11 sites for santer and 7 and 24 sites for roman depending on the sample area. This can be completed within a maximum of three sampling days (Table 8.5). This is less than the effort required to detect changes in abundance of these species and a monitoring programme should be based on the minimum requirements for detecting change in abundance of the selected indicator species.

UVC surveys

In accordance with the SAEON Diving Operations Manual[23] a maximum of two dives may be conducted per person per day when diving is conducted to depths greater than 24m and multiple day diving is anticipated. As diving conditions in Algoa Bay are usually favourable for only short durations, repetitive diving over several days is often conducted. These regulations would limit a diving operation to four dives per day with a dive team of four qualified researchers. In addition to the practical difficulties and higher financial costs required to conduct a UVC survey, the results from this study indicate that under these regulations greater sampling effort (number of days – santer 55; roman 18) is required to detect similar changes in the abundance of the two selected indicator species (Table 8.3). Bennett (2009) reached similar conclusions. Even though UVC provided less variable estimates of relative abundance in his study, the sampling efficiency for controlled angling surveys for monitoring temperate reef fish communities was superior due to the restrictions imposed by SCUBA on UVC. Using complementary survey methods has been recommended for monitoring programmes (Lincoln Smith 1989; Bennett 2007; Bennett *et al.* 2009) and conducting UVC as a component of the Algoa Bay monitoring framework allows for comparisons between methods as well as *in situ* observation and potential validation of trends identified through controlled angling. Although not dealt with in this thesis, UVC surveys allow for the assessment and monitoring of the macro-benthic communities simultaneously and therefore offer other advantages over controlled angling surveys. It is therefore recommended that a combination of UVC and controlled angling surveys be conducted within Algoa Bay at selected sites.

Table 8.5. Number of sampling days required in each area for effective monitoring of CPUE and length for two indicator species.

Survey method	Controlled angling				UVC		
Parameter	CPUE		Length		Relative abundance		Length
Study area	Santer	Roman	Santer	Roman	Santer	Roman	Roman
BI	8	12	1	1	55	18	30
WC	3	22	1	3			
CP	8	20	1	2			
RB	7	6	1	1			
Ev	5	12	1	1			
StC	12	-	2	-			
BB	6	-	1	-			
Total	49	90	8	11			

Note: - indicates not effective to monitor roman due to low abundances

[23] Coastal research diving is conducted in collaboration with SAEON and all diving operations must therefore comply with their Operations Manual

Programme design

In order to ensure optimal monitoring of reef fish communities in Algoa Bay, reef sites from both community Group 1 and Group 2 need to be selected (see Chapter 3). Furthermore, in order to evaluate future responses to changes in fishery management regulations, sites which are likely to be protected in the future need to be selected and compared to sites which will remain open to exploitation (Status). Another dimension in the potential monitoring framework design is the differentiation of nearshore and offshore reef communities which showed a slight distinction from each other in the hierarchical classification of Group 2 communities (Chapter 3) (Inshore/Offshore). Due to the low abundance of roman in the nearshore Group 2 communities it is also possible to design a programme that only evaluates changes in santer abundance in these areas. The Ev reef area was excluded from further considerations as it is comprised of a very small reef area and its long-term value may be compromised by natural perturbations (e.g. sand movement). Based on these considerations, three options for monitoring reef fish communities using controlled angling with varying levels of intensity are presented in Table 8.6.

The first option is the optimal design, which includes Community, Reserve Status and Inshore/Offshore reefs designed to monitor both indicator species. This design requires a total of 78 sea-days per year. Option 2 presents a compromise in which the framework is designed to only take into account the requirements of santer as an indicator species but retains the Community, Reserve Status and Inshore/Offshore factors in the programme and would require 47 sea-days per annum. The monitoring framework which is considered to contain the minimum requirements for monitoring in Algoa Bay includes only the Community and Status factors in the design and requires only 36 sea-days per annum.

Including a UVC component into the design with sampling in the RB and BI areas only would involve an additional 18 days at BI to effectively monitor roman abundance. As insufficient diving was conducted in RB to evaluate sample size requirements, it is assumed that a similar level of intensity would be needed leading to a total of 38 dive days with a team of four researchers. The time requirements for UVC could be reduced through increasing the team size.

Season, depth and reef profile were shown to have significant influences on either community structure or individual species abundances and sizes or a combination of both. Stratification of effort in each study area during the design of the sampling protocol is therefore important. Furthermore, important parameters such as bottom temperature, visibility and reef rugosity should be recorded where possible to facilitate the analysis and interpretation of results.

These proposed sampling designs involve a considerable commitment of financial resources and skilled manpower yet represent what is required for effective ecological monitoring of the state of reef fish communities in Algoa Bay. To improve the value and interpretation of future results, sidescan sonar and multi-beam surveys of selected reef monitoring sites should be conducted. It is also recognised that this study was limited to depths of approximately 35m in order to avoid unnecessary barotrauma injuries to fish, and safe diving limits are constrained to 30m. Several areas important for linefish species (carpenter, panga, silver kob) in Algoa Bay occur below this depth, but there is currently little knowledge of the composition of reef fish communities in these habitats. A baseline survey of deeper habitats is therefore warranted to identify potentially important linefish areas for consideration in future marine spatial planning and management in Algoa Bay. A combination of acoustic studies and remote camera systems such as ROVs and BRUVs could be used in order to conduct such research.

Table 8.6. Design options of the controlled angling monitoring programme in Algoa Bay. Numbers indicate days required for field sampling.

	Protected		Exploited		Indicator species
[illegible]					
Group 1 reef community	StC (f)	12	BB	6	Santer
Group 2 reef community	BI	12	RB	6	Santer and roman
	WC (f)	22	CP	20	Santer and roman
Total		46		32	78 days total
[illegible]					
Group 1 reef community	StC (f)	12	BB	6	Santer
Group 2 reef community	BI	12	RB	6	Santer and roman
	WC (f)	3	CP	8	Santer only
Total		27		20	47 days total
[illegible]					
Group 1 reef community	StC (f)	12	BB	6	Santer
Group 2 reef community	BI	12	RB	6	Santer and roman
Total		24		12	36 days total

(iii) Non-reef fish communities

Due to the nature of the research, demersal trawl surveys and the variability in catches, it was not feasible to determine the number of trawls required using a power analysis. However, assessing changes in community composition and the use of univariate measures of diversity have been shown to be effective in assessing temporal changes in community structure (Clarke and Warwick 2001a). Sampling effort was therefore determined as the number of research trawls required to adequately obtain a representative sample of the demersal fish communities in Algoa Bay during future surveys.

Methods

The cumulative number of species captured per successive trawl for each survey season and year was determined taking into account only those species which were present in more than 5% of trawl stations. A logistic curve was fitted to the data using a non-linear least squares procedure and was used to estimate the number of trawls required in order to successfully sample 50% and 95% of the species present. This was conducted to determine the sampling effort required for monitoring changes in community assemblages using species lists. The logistic curve took the form (Brown and Walker 2004):

$$n = \frac{1}{1 + exp\left(-ln19\frac{(E - E_{50})}{(E_{95} - E_{50})}\right)} \qquad \text{Equation 8.6}$$

where n is the maximum number of species, E is the number of trawl station, E_{50} is the number of trawl stations required to sample 50% of the species present and E_{95} is the number of trawl stations required to sample 95% of the species present.

Results and discussion

Biannual demersal biomass research trawl surveys have been conducted along the south-east coast of South Africa since 1986. This data were used to assess trends in community composition, abundance and size of selected key species over trawlable grounds in Algoa Bay (Chapter 4). The research cruises are based on a pseudo-random stratified survey design based on the available area in each depth stratum along the south-east coast between Cape Agulhas and Port Alfred (Badenhorst and Smale 1991). The resulting station locations within Algoa Bay may therefore not be optimal on a local scale, nor be representative of habitats within Algoa Bay. A least squares procedure was used to estimate the number of trawls required to sample 95% of the species previously encountered within Algoa Bay. This would allow for accurate preparation of species list which are suitable for assessing community changes using the AvTD and VarTD diversity indices (Clarke and Warwick 2001a). The data used in this assessment suggests that a minimum of five to six seasonal trawls are required to ensure that 95% of the species contributing to the community assemblages are sampled. An average of eight trawls have been conducted in Algoa Bay annually (1986-2008) suggesting that an increase in effort would be required to successfully sample and monitor the community assemblages in Algoa Bay. The trawls also provide important biological information on key species which can be used as indicators for the state of demersal non-reef fish communities. The transient nature of many species contributes to high variability in relative abundances; however, mean length will provide more accurate information on long-term trends in these communities.

Demersal trawl surveys are one of the only current techniques used to assess non-reef fish communities yet they are highly destructive and the efficacy of using these techniques for long-term ecological monitoring programmes, and within MPAs is questionable. In light of the proposed future proclamation of additional no-take zones incorporating vast areas of unconsolidated sediments in Algoa Bay, the use of other non-destructive survey techniques such as acoustic surveys, ROVs and BRUVs should be investigated.

(iv) Other targeted marine living resources

Due to the poor status of linefish stocks and the declaration of a state of emergency in the linefishery in 2000 (DEAT 2000) this thesis focused on providing baseline information on the state of linefish communities in Algoa Bay. An assessment of pelagic fish and chokka-squid stocks was beyond the scope of this study. Future monitoring in Algoa Bay needs to incorporate an assessment of the state of both these targeted stocks to determine the success of management interventions. Current programmes for assessing these stocks are in place on a national level. Their applicability for local level assessments needs to be determined.

Biomass surveys of commercially important pelagic species (sardine and anchovy) are conducted biannually by DAFF along the west and south-east coasts of South Africa (Hendricks and Bali 2011). These surveys make use of hydroacoustics to estimate fish biomass and midwater trawls are conducted to obtain samples for biological analysis. Furthermore a dedicated study using hydroacoustics was initiated in 2009 to investigate the effects of the pelagic fishery closure around Bird and St Croix Island (Merkle and Rademan 2011). An evaluation of these surveys needs to be conducted in order to investigate feasible options and design a protocol for future long-term monitoring of pelagic stocks in Algoa Bay.

Chokka-squid CPUE data are used to establish trends in stock status which is used to determine the duration and time of seasonal closures for the following season. A fishery independent means of data collection from key spawning areas in Algoa Bay needs to be prioritised in order to allow for analysis of long-term spatial and temporal trends in chokka-squid spawning stock density in these areas.

(c) Response

Indicators of response are based on the implementation of management actions by regulatory authorities. In order to obtain the required information, accurate logs of all actions undertaken including monitoring, enforcement, education, research and stakeholder engagement need to be maintained by each agency (Pomeroy *et al.* 2004). Due to multi-jurisdictional management in Algoa Bay records will need to be collated from various government agencies and clear information sharing agreements and procedures need to be established at the onset of the monitoring programme. This will ensure that appropriate information is recorded by each agency and will contribute to the overall evaluation of management responses in Algoa Bay. An inventory of the key management and research agencies and personnel in Algoa Bay needs to be collated and a steering committee with representatives from all above agencies should be established to facilitate the development and implementation of a standardised protocol for data collection. A lead agency needs to be identified to which the tasks of management, coordination and implementation of the monitoring programme are delegated. The key institutions involved in management in Algoa Bay include the Oceans and Coasts branch of DEA, DAFF, SANParks, the Nelson Mandela Metropolitan municipality and the Ndlambe local municipality. Furthermore, research institutions such as SAIAB, SAEON, Rhodes University, Bayworld and the Nelson Mandela Bay Metropolitan Municipality are actively involved in research projects in Algoa Bay, providing information to resource managers.

Records of financial and manpower commitments (operational budgets) to management need to be maintained by all institutions in Algoa Bay. Protocols for accurate recording of coastal and offshore enforcement and monitoring patrols need to be established to record the number, dates and times of patrols, their spatial coverage, the types and extent of activities occurring as well as compliance levels with regulations. The imposition of new no-take zones and regulations is often met with opposition from stakeholders, and improving support for and compliance with new regulations requires an active education and awareness programme (Agardy *et al.* 2011). This can take the form of stakeholder meetings, schools educational programmes, improved advertisement of regulations through appropriate signage and pamphlets, and information dissemination during monitoring and enforcement patrols. Partnerships need to be established with institutions to facilitate education and awareness campaigns.

In the long-term a steering committee should be responsible for the expansion of the proposed monitoring framework to include additional PSR indicators over and above the fishery related indicators identified in this study. Examples include indicators of water quality, point discharges, coastal development, shipping etc.

8.2.4 Step 4: Delegate responsibilities for the collection of data and management of the programme

Although many organisations will be involved in implementing the tasks outlined above, the overall responsibility for coordination and implementation needs to be delegated to one authority. SANParks are mandated to manage protected areas in South Africa, and the responsibilities for management of some MPAs, including Bird Island in Algoa Bay, has been delegated from the Oceans and Coasts branch of DEA. SANParks will therefore be directly responsible for large areas of the marine and coastal environment in Algoa Bay following proclamation of the AENP MPA and it is therefore recommended that they be tasked with the overall management and implementation of the monitoring framework.

The first task should be to establish a steering committee inclusive of all agencies. Furthermore, setting up appropriate lines of communication with regulatory authorities and research institutions and the identification of responsible personnel needs to be undertaken to ensure coordination of tasks and to address any potential data sharing issues. An important component would be to establish roles, responsibilities and timeframes for implementation of tasks as well as the submission standards and protocols for data to the management body.

One of the current aims in natural resource management is to improve the participatory management approach including stakeholders in all aspects of management (Pinkerton 2009). This includes the task of monitoring, and developing partnerships with public stakeholders could improve the amount and range of monitoring data collected within the restrictions of the limited capacity in many regulatory departments. Furthermore, allocating tasks to stakeholders could greatly reduce the costs of monitoring, and some NGOs may be able to assist through the provision of staff time or funding (Pinkerton 2009). Incorporating stakeholders will require the development of a training programme to ensure suitable methods are utilised for data collection, and periodic quality control checks will need to be put in place to ensure the data meets the required standards.

A central database will need to be designed to incorporate all aspects of the monitoring protocol. Protocols for data management, entry, screening and analysis will need to be established to ensure timeous evaluation of monitoring progress and suitably qualified personnel will need to be appointed. Quality assurance procedures will also need to be developed and implemented. Any gaps in data collection of submissions will need to be identified timeously and actions implemented to rectify the problems.

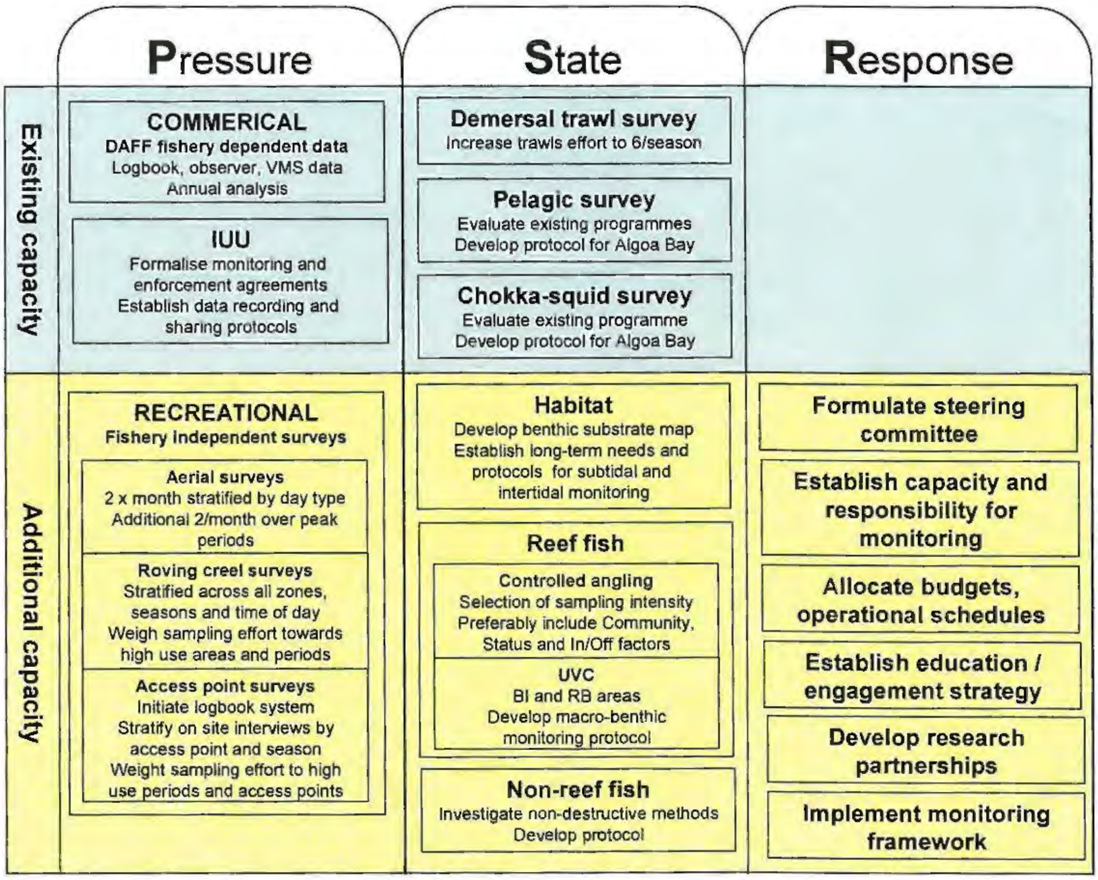


Figure 8.11. Monitoring components and tasks required in Algoa Bay.

8.2.5 Step 5: Evaluate implementation of actions/recommendations arising from ongoing monitoring for adaptive management

Ongoing monitoring in Algoa Bay will identify where additional management action is required and necessitate changes in the existing protocols. Several agencies are responsible for management and monitoring in Algoa Bay and they will therefore be required to update routine protocols or implement new actions in order to capture new requirements arising from periodic evaluation of monitoring results. This is a central aspect to adaptive management in the context of ecosystem based management, however, new recommendations arising from such programmes are seldom incorporated by the responsible agencies defeating the purpose and cyclical nature of adaptive management. The steering committee, established as one of the key responses of this programme must, therefore take responsibility for communicating the required changes and additions to monitoring and management in Algoa Bay to all agencies periodically. A key role of the steering committee will therefore be to ensure that all agency specific protocols are updated to capture any new requirements or changes to existing activities. This may also require sourcing additional funding for expanding existing capacity, or identifying potential sources of funding to which proposals should be submitted by the responsible agency. A central component to this process will be a periodic feedback mechanism which ensures that all agencies report back to the steering committee as to their progress on implementing required new actions. Through such a mechanism the steering committee will be able to evaluate the effectiveness of adaptive management and identify problem areas which require addition attention in the future.

8.2.6 Conclusions

This chapter has presented a monitoring framework for Algoa Bay with the overall objectives being to evaluate long-term trends in the state of the biophysical resources, pressures exerted by fisheries and to evaluate the management initiatives implemented locally. This framework will allow the quantification of the effects of spatial conservation initiatives on the biophysical and socio-economic environments in Algoa Bay. Although it focuses primarily on monitoring resources targeted by fisheries and extractive resource uses, it provides a basis for initiation and implementation of a protocol which can be expanded to include additional features and activities as funding and capacity becomes available.

Monitoring programmes need to be carefully designed in order to allow actual responses to be distinguished from the natural variability in the parameter of interest. This requires careful selection of parameters to be monitored, which reflect the management objectives, and a comprehensive understanding of the factors influencing the variability in the parameter of interest. There is currently no formal fisheries independent monitoring programme to evaluate trends in the state of targeted resources or the pressures exerted by recreational fisheries, and dedicated programmes are required to capture this information. Key indicators therefore need to be identified and protocols for data collection established. Chapters 3 and 4 provided baseline analyses of the reef linefish and demersal ichthyofaunal communities, and aided in identifying the key factors influencing the distribution and variability in communities and dominant species in each community. This data provided the information required to aid the design of a monitoring programme for reef fish communities through the identification of representative sample sizes and the determination of sample size requirements at each site. A flexible approach with options of varying sampling intensity was also presented. This allows decisions regarding the monitoring design to be based on the final design of no-take areas once decided upon, the financial resources available for this component of the monitoring programme, and the level of data required for evaluating responses to implementation.

The demersal soft substratum communities are currently monitored through trawl surveys designed and stratified on a national level. This study revealed that the number of trawls conducted locally needs to be increased in order to improve the representivity of the demersal communities in the catch. However, these surveys require large specialised vessels which are costly, and are therefore beyond the logistical and financial resources of local management authorities. Furthermore, they are highly destructive to benthic habitats and biota and are not suitable for sampling within MPAs, thus alternative non-destructive techniques for assessing the demersal communities need to be investigated. Although other species are targeted in the study area (chokka-squid; sardine; sharks) they were not the focus of this study and future research needs to be conducted to aid in the design of monitoring protocols for these species.

Chapter 5 highlighted the high degree of spatial and temporal variability in recreational fishing effort. On-site survey techniques are labour intensive, time consuming and costly to implement. Due to the inaccessibility and length of the coastline, and the high level of variability in angler numbers, aerial surveys are recommended for obtaining shore fishing effort data as they provide a rapid and cost-effective means to obtain data over the large study area. The implementation of a formal logbook system for the recreational boat-based fishery at launch sites provides a cost-effective means to obtain total launching effort at each legally registered site but will require effective management and compliance evaluation to ensure the accuracy of data. On-site creel surveys can then target high use and peak periods in order to increase the opportunities for obtaining catch data. This proposed design for monitoring recreational fisheries ensures that trends in effort and changes in catch composition or size of species can be quantified through a cost effective sampling protocol. However, dedicated management of the programme is required to ensure the consistency and accuracy of data.

Commercial fisheries have traditionally been monitored using logbook data with no dedicated fisheries independent monitoring protocols in place. However, there are many biases and inaccuracies associated with logbook data, raising concerns about its use for evaluating trends in catch and effort. VMS units are legal requirements for commercial vessels and capture highly accurate spatial data which can be used to quantify fishing effort through the use of rules to assess vessel behaviour. Integrating these two data sources provides an excellent means to evaluate the accuracy of the reported logbook effort and through active engagement with fishery stakeholders, can be used to improve the accuracy of data recorded and submitted by them. Fishery stakeholders are likely to be willing to improve the accuracy of their data in light of future spatial closures and potential loss of access to former fishing grounds. Programmes for the collection of this data are already established, and the costs associated with monitoring commercial activities locally will be limited to obtaining and analysing the required data and engaging with local fishery stakeholders. An essential requirement is that data access agreements be reached with the national regulatory authorities and fishery stakeholders. As a result a local monitoring programme for the commercial fisheries can be established with minimal financial investment.

Numerous parameters were identified to monitor trends in the pressure, state and responses in Algoa Bay and where possible these were based on readily available data sources to ensure a cost-effective monitoring programme. Certain components, however, require dedicated programmes to be established and frameworks have been recommended based on the baseline information. Periodic revision and updating of the sampling protocol and reference points will be required to ensure that the monitoring programme achieves its objectives. The steering committee will play a central role in establishing a reporting mechanism to ensure that additional requirements identified through ongoing monitoring are successfully implemented by the responsible agencies.

CHAPTER 9
CONCLUSION

Globally, there is sufficient evidence with which to illustrate that single species management approaches have largely failed to maintain sustainable harvest levels of target stocks and ecosystem health. The development of holistic, participatory and transparent ecosystem based approaches for fisheries management have therefore received considerable attention, with significant conceptual advances occurring since the Reykjavik Declaration on Responsible Fisheries in 2001. Following this, at the 2002 World Summit of Sustainable Development, countries committed to developing ecosystem approaches to fisheries with a target date for implementation of 2010. Subsequently several countries have made considerable progress towards developing EAF principles and approaches, yet implementation has been a major challenge, with only few successful examples (Link 2010; Katsanevakis *et al.* 2011). The major focus of EBM is to manage diverse human activities in a geographically defined area, particularly those such as fisheries which have direct impacts, in order to maintain ecosystem health. Assessing spatial trends in resource use and developing spatial management plans is therefore a central component for EBM. Marine spatial planning and ocean zoning have increasingly been used to integrate the diverse and competing management objectives often encountered in ecosystem based management. The development of systematic planning tools, such as Marxan, has facilitated the identification of key areas for conservation while taking cognisance of socio-economic aspects in design and planning. This has facilitated the development of numerous spatial plans in recent years and contributed to a major step towards more holistic, transparent and participatory ecosystem based management in many coastal regions of the world (Lombard *et al.* 2007; Alonso *et al.* 2008; Gutiérrez-Moreno *et al.* 2008; Klein *et al.* 2008a; Klein *et al.* 2010; Agostini *et al.* 2010). Furthermore in South Africa systematic spatial planning has been used as a tool to evaluate the existing reserve networks nationally and regionally (Lombard *et al.* 2004; Clark and Lombard 2007) to aid in the identification of future requirements of conservation. However, this study is one of the first to be conducted on a local level in South Africa that has incorporated detailed, spatially explicit, socio-economic data from a range of fisheries from the onset of the process, and identified and obtained the required data for key biophysical features deemed important for planning. This study has therefore contributed to South Africa moving towards developing ecosystem based management approaches on a local level in line with other international best practice standards.

This study applied a precautionary, scientifically robust, holistic ecosystem based planning approach to provide reliable biological and fisheries data to support future management decisions for marine conservation in Algoa Bay. Alternative spatial designs for enhancing the conservation and protection of marine habitats and communities through the development of no-take areas were identified and evaluated against the impacts to fisheries using recognised systematic planning methods. This provided a quantitative means to evaluate costs and benefits of establishing new no-take zones locally within Algoa Bay. Stakeholder engagement is a key component of ecosystem-based management, and previous attempts to expand the MPA network in Algoa Bay failed due to poor communication with stakeholders and a lack of data with which to support decisions made regarding the proposed design of the MPA footprint and no-take zones. These limitations resulted in wide scale public opposition to the development of new no-take zones in Algoa Bay and raised doubt as to the level of consideration the management authorities had given to accommodating the socio-economic activities in the area.

Previous designs for the proposed MPA footprint were based on expert opinion with little consideration of quantitative spatial data in the overall design process. There was no means to explicitly quantify what and how much would be protected within the proposed footprint, which raised concerns as to the conservation value of the design both locally and regionally. Furthermore, there was no way to quantify the impact of the closure on the commercial and recreational fishing sectors. This stressed the importance of improving the knowledge on what important biophysical features are present within Algoa Bay and how they are distributed spatially.

In order to overcome these limitations, this study identified all available sources of spatially explicit biophysical data important for marine conservation. The absence of spatial information on fish resources, which are heavily targeted by numerous fishery sectors and most stocks are considered overexploited, was seen as a major limitation in the design process as the protection of fish stocks was a key management objective for the proposed MPA in Algoa Bay. Data on the fish resources were therefore obtained (Chapters 3 and 4) in order to better understand the spatial and temporal dynamics and develop spatial layers for planning purposes. Where no quantitative data were available, qualitative behavioural data on fish movements in Algoa Bay were used to provide spatial information for inclusion into spatial planning. This ensured that the fish resources were adequately considered in the design process along with other key biophysical features.

These data provided the means to identify the optimal spatial design for the inclusion of biophysical features in no-take zones and to evaluate the design formerly proposed by experts. This process confirmed that the current St Croix reserve and Bird Island MPA are located in the areas of highest conservation importance in Algoa Bay, and that the overall MPA footprint and no-take zones proposed by experts accommodated the areas of greatest conservation importance relatively well. However, it was evident from the spatial analysis that improvements to the proposed MPA footprint design could be made which would enhance both the local and regional conservation value, without greatly increasing the associated impacts on fisheries. This process was therefore successful in identifying and mapping biophysical features in Algoa Bay which were deemed important for spatial planning and management. In doing so, it was possible to use a quantitative approach to aid in the design of no-take zones which attained the desired levels of protection for these features, thereby providing adequate justification for the selection of areas for protection. Based on these findings, it is recommended that this type of process becomes a minimum requirement that should be conducted in future studies as a first step in identifying areas for inclusion in no-take MPAs.

Inadequate consideration of fisheries activities and consultation with stakeholders in the design and proclamation of no-take MPAs has led to conflict and poor compliance in several instances. This has resulted in MPAs being ineffective in protecting the resources they were established to conserve, and led to the need for intensive and costly enforcement programmes. Public opposition resulted in the cessation of the proclamation of the proposed MPA in Algoa Bay and led to the initiation of this project aimed at quantifying the impact of conservation on fishery activities. A key aspect of this project was therefore to determine what fisheries activities occur within Algoa Bay and to develop spatial indices which could be used for systematic conservation planning.

Systematic conservation planning incorporated spatially explicit fisheries data into the planning and design process *a priori*. In doing so, optimal no-take reserve design options which attained the desired conservation targets for the biophysical features but minimised the selection of areas important to the fisheries were identified. The inclusion of the fisheries data into the systematic conservation planning process resulted in a considerable reduction in the level of impact to all fisheries sectors while still attaining the desired conservation targets. The approach was effective in integrating the competing objectives of conservation and fisheries into the reserve design process. Furthermore, it allowed for the evaluation of the effects of different interpretations of the fishery costs (effort versus economic importance) on the overall design process and provides a solid platform from which to engage stakeholders. Further development of the cost layer can be conducted in collaboration with fishery stakeholders and the effects on the reserve design options investigated in order to foster this sector's support and gain agreement on the final MPA design. The quality and accuracy of the spatial fisheries data played an important role in the overall design process, and the detailed information obtained in this study contributed to the successful consideration and inclusion of fishery activities in the design options identified. The development of spatial indices of socio-economic "costs" is fundamental to the integration of conservation and fisheries objectives into EBM, and this study has demonstrated the need for fishery independent surveys to evaluate recreational effort and the value of VMS and observer data in developing spatial indices of fishing effort for commercial sectors.

The development of new MPAs is usually met with strong opposition from user groups due to the exclusion of commercial/economic and recreational activities. Furthermore, politicians in developing countries are faced with enhancing local economic opportunities for impoverished coastal communities and often look to fishery resources as a solution, failing to take into account the existing pressures on the resources and the depleted status of the stocks. Balancing the need for improved conservation of marine habitats and biota with management of fisheries stocks in developing countries in which there is increasing political pressure for further fisheries development is therefore critical in the future management of marine ecosystems. MPAs offer an effective means to ensure protection of critical habitat and spawner biomass of targeted species within a spatially defined area and contribute to re-seeding adjacent fisheries. This study has demonstrated the value of systematic conservation planning in providing alternative designs and supporting information to aid decision-making, and providing a platform with which to initiate stakeholder engagement. Although output designs from systematic planning present the optimal designs, several additional factors (practicality, financial costs, stakeholder buy-in etc.) need to be considered before agreement can be reached on the final design of the MPA. However, SCP was successful in ensuring that alternative options were presented, biophysical features were adequately represented within the no-take zones and their reasons for inclusion were justified with scientific data, fisheries activities were considered *a priori*, and that costs were quantified. These are critical aspects for ensuring that the design process is scientifically defensible and the decisions made are suitably justifiable. This is the first in-depth marine spatial planning project to be conducted on a local scale in South Africa and it has therefore made an important contribution to understanding the role that SCP can play in adopting an ecosystem based approach locally. It has also demonstrated that previous data limitations preventing the use of quantitative approaches can be overcome through carefully planned and designed studies. The baseline fish and fisheries data have contributed to understanding the complex spatial heterogeneity in marine ecosystems which occur on a small scale, and contributed to evaluating the conservation importance of the AENP on a regional and national level. This study has also contributed to understanding the effects that different fisheries cost layers can have on the reserve design process, and that acceptable compromises between fisheries and conservation objectives can be reached. It has demonstrated the important role SCP can play in developing defensible products with which to engage stakeholders where conflicting interests have previously hampered marine conservation efforts in South Africa. The approach used in this study is therefore a valuable means for future marine conservation planning in other areas where data are limited.

This study has demonstrated the successful integration of biophysical and fisheries data in order to improve the design of no-take zones in MPAs. However, there is often great uncertainty as to whether the desired benefits of spatial closures will be achieved. In order to ensure the long-term support of local stakeholders, the benefits and impacts of MPA establishment need to be quantified on an ongoing basis. The effectiveness of no-take zones in achieving the desired responses is dependent on successful implementation and enforcement of the spatial regulations decided upon by management authorities and stakeholders. Monitoring the implementation of management actions, changes in the pressures exerted on the ecosystem, and the responses of biota is therefore critical for evaluating and quantifying the effects of MPAs. Monitoring and evaluation is also a key aspect of the adaptive management cycle, allowing for continual improvement in the understanding and management of complex ecosystems in which data are limited, and contributes to a better understanding of decadal cycles in the biophysical environment.

No-take MPAs aim to protect critical habitats and representative populations of targeted species through the exclusion of fisheries activities. In doing so it is anticipated that the protected spawner biomass will re-seeding adjacent fisheries through larval dispersal and spill-over of adults. These benefits are best realised in long-lived resident species and can be quantified through differences in abundance and size between exploited and protected sites. Long-term monitoring programmes are required to evaluate whether the imposition of no-take zones effectively reduce pressure and contribute to improving the status of the resources. Designing a monitoring programme to evaluate future changes requires a comprehensive understanding of the spatial and temporal variability in the parameters to be monitored. This was achieved in this study through baseline assessments of the fish communities, representing the state of the resources, and the fishing activities, indicative of the direct pressures on the ecosystem. This provided the information necessary to design monitoring protocols which were statistically robust. Furthermore, these baseline studies provided reference points against which future comparisons can be made. Tools and guidelines for monitoring the implementation of ecosystem-based management and evaluating responses are generally lacking and poorly developed.

In this study the PSR framework was used to develop a simple yet comprehensive protocol for evaluating changes in key parameters locally. Reference points were established using baseline data as a guide to determine the accepted level of variability from baseline values. As there are not formal guidelines for the development of reference points, these represent provisional levels with which to evaluate management performance and will require periodic review to ensure target reference points are sufficient to ensure acceptable conditions are maintained, and that limit reference points do indeed trigger further research and management actions when required. The implementation of a monitoring programme from the onset of MPA establishment will play a valuable role in quantifying the benefits of no-take MPAs to the adjacent fisheries. Fishery stakeholders should be actively engaged during the monitoring process as this will improve their understanding of the importance and role of no-take MPAs in marine conservation and improve fisheries data used in evaluation.

Conservation and fisheries management efforts have focused on identifying suitable sites for the establishment of MPAs, with far less emphasis placed on monitoring their effectiveness in achieving the desired objectives. As a result, baseline data for long-term comparative assessments is often lacking and the effectiveness of many MPAs cannot be established. The baseline surveys conducted during this study have provided much of the spatial data required for planning purposes, detailed information on the relative abundance, size structure and composition of fish resources, the relative pressures exerted by fisheries, and the composition of the harvests, all of which are essential for evaluating future changes. This study has made a significant contribution to understanding the data and sampling requirements for monitoring and evaluating future long-term trends in fish communities and fisheries activities within the temperate region of the South African coastline. Ongoing monitoring is fundamental to improving our limited knowledge of complex marine ecosystems, the interactions between ecosystem components and the way in which they respond to management. As monitoring improves our understanding of ecosystem interactions and responses to changes, management measures can be continually improved through adaptive management.

9.1 Recommendations for future improvement

Determining the spatial distribution of habitat types is of critical importance to both spatial planning and monitoring yet is often a major hurdle which needs to be overcome in marine environments. While supra- and intertidal habitat types can be mapped relatively easily via remote sensing and field surveys, mapping subtidal habitats is far more difficult and costly, requiring the use of specialised equipment. In this study a cost-effective method was used to map potential reef areas and verification was conducted using jump cameras and diving surveys. However, this approach is only suitable for use in areas where reef is known to occur and reef complexes over a broader spatial scale need to be identified via alternative means. Improved mapping of the subtidal habitats will allow for higher accuracy and confidence in the location of reef complexes. This will contribute to improving spatial planning, and designing baseline surveys and monitoring strategies by allowing consideration of a range of unconsolidated sediment types. Future effort should therefore be placed on using sidescan, multi-beam and acoustic ground discrimination systems (AGDS) to improve the accuracy of subtidal habitat maps for marine planning and monitoring studies.

The biophysical layers used in the planning process were limited to key habitat types, important fishery species and key process areas identified through this and past studies. Macro-benthic communities may differ across physically similar habitats due to environmental conditions or other anthropogenic drivers. Characterisation of macro-benthic communities will advance the classification of habitat types within the study area thereby contributing to improving the overall planning process. Due to the sessile nature of these communities they also provide an effective means to monitor ecosystem changes. Future studies that characterise these communities will benefit both planning and monitoring in the long-term.

This study focused specifically on the management of fishery activities, as they have the greatest direct impact on marine resources. However, other non-consumptive activities (e.g. passive recreation and tourism) and anthropogenic drivers (e.g. development; pollution etc.) may also influence marine ecosystems and conflicts between competing needs may arise. These activities need to be identified and mapped for future consideration in spatial planning. In this study only take and no-take zones were considered in spatial planning due to the extractive use of resources by fisheries. Multiple zonation plans which accommodate different types of activities in a spatially explicitly manner may further reduce conflicts between different user groups, contributing to a more integrated approach to ecosystem management. Future research should focus on obtaining spatial data on the full range of activities occurring locally so that they can be included in spatial planning analyses and contribute to the development of a MPA that is zoned for multiple uses. The investigation of temporal closures to protect species which are seasonally abundant and vulnerable to fisheries activities should also be investigated further.

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APPENDIX 1: RULES AND REGULATIONS FOR MPAS IN SOUTH AFRICA

These are the general rules for marine protected areas in South Africa as outlined in the Marine Living Resources Act (Act 18 of 1998) and which will apply to any new no-take zones proclaimed in Algoa Bay.

CHAPTER 4

MARINE PROTECTED AREAS

Marine protected areas

43. (1) The Minister may, by notice published in the *Gazette*, declare an area to be a marine protected area—

(a) for the protection of fauna and flora or a particular species of fauna or flora and the physical features on which they depend;
(b) to facilitate fishery management by protecting spawning stock, allowing stock recovery, enhancing stock abundance in adjacent areas, and providing pristine communities for research; or
(c) to diminish any conflict that may arise from competing uses in that area.

(2) No person shall in any marine protected area, without permission in terms of subsection (3)—

(a) fish or attempt to fish;
(b) take or destroy any fauna and flora other than fish;
(c) dredge, extract sand or gravel, discharge or deposit waste or any other polluting matter, or in any way disturb, alter or destroy the natural environment;
(d) construct or erect any building or other structure on or over any land or water within such a marine protected area; or
(e) carry on any activity which may adversely impact on the ecosystems of that area.

(3) The Minister may, after consultation with the Forum, give permission in writing that any activity prohibited in terms of this section may be undertaken, where such activity is required for the proper management of the marine protected area.

APPENDIX 2: SUMMARY OF TELEOST AND ELASMOBRANCH SPECIES CAUGHT IN ALGOA BAY DURING RESEARCH TRAWL SURVEYS

Taxa	Family	Scientific name	Common name	Frequency of occurrence (FoC) (%)	% of catch (biomass)
Agnatha	Myxinidae	*Myxine capensis*	Cape hagfish	1.6	0.0
Chondrichthyes	Alopiidae	*Alopias vulpinus*	Thresher shark	0.8	0.0
	Callorhinchidae	*Callorhinchus capensis*	St Joseph	93.5	4.8
	Carcharhinidae	*Carcharhinus brachyurus*	Bronze whaler	7.3	0.4
		Carcharhinus obscurus	Dusky shark	0.8	0.0
	Dasyatidae	*Dasyatis brevicaudata*	Short-tail stingray	2.4	0.0
		Dasyatis chrysonota	Blue stingray	65	1.7
		Dasyatis thetidis	Thorntail stingray	1.6	0.0
		Dasyatis violacea	Pelagic stingray	4.9	0.1
	Gymnuridae	*Gymnura natalensis*	Backwater butterfly ray	26	0.9
	Hexanchidae	*Notorynchus cepedianus*	Broadnose sevengill shark	1.6	0.1
	Myliobatidae	*Myliobatis aquila*	Eagle ray	83.7	4.2
		Pteromylaeus bovinus	Bull ray	4.9	0.1
	Narkidae	*Narke capensis*	Cape onefin electric ray	18.7	0.0
	Odontaspididae	*Carcharias taurus*	Spotted ragged-tooth	0.8	0.0
	Pristiophoridae	*Pliotrema warreni*	Sixgill sawshark	7.3	0.0
	Rajidae	*Dipturus pullopunctata*	Slime skate	3.3	0.0
		Leucoraja wallacei	Yellow-spot skate	3.3	0.0
		Raja miraletus	Twin-eye skate	79.7	0.4
		Raja straeleni	Biscuit skate	85.4	1.1
		Rajella dissimilis	Ghost skate	0.8	0.0
		Rostroraja alba	Spearnose skate	74.8	2.1
	Rhinobatidae	*Rhinobatos annulatus*	Lesser guitarfish	61	1.0
	Scyliorhinidae	*Halaelurus natalensis*	Tiger catshark	83.7	0.4
		Haploblepharus edwardsii	Puffadder shyshark	22	0.1
		Poroderma africanum	Pyjama catshark	9.8	0.0
		Poroderma pantherinum	Leopard catshark	6.5	0.0
	Sphyrnidae	*Sphyrna lewini*	Scalloped hammerhead	0.8	0.0
		Sphyrna zygaena	Smooth hammerhead	35	0.7
	Squalidae	*Squalus megalops*	Bluntnose spiny dogfish	91.9	4.6
	Torpedinidae	*Torpedo fuscomaculata*	Blackspotted electric ray	30.9	0.1
		Torpedo nobiliana	Atlantic torpedo	5.7	0.0
	Triakidae	*Galeorhinus galeus*	Soupfin shark	35	1.1
		Mustelus mustelus	Smooth-hound shark	70.7	2.0
		Mustelus palumbes	White-spotted smooth-hound shark	43.1	0.5
		Triakis megalopterus	Spotted gullyshark	1.6	0.0

Taxa	Family	Scientific name	Common name	Frequency of occurrence (FoC) (%)	% of catch (biomass)
Osteichthyes	Acropomatidae	*Neoscombrops annectens*	Sombre splitfin	0.8	0.0
		Synagrops japonicus	Japanese splitfin	8.1	0.0
	Ariidae	*Galeichthys ater*	Black seacatfish	4.9	0.0
		Galeichthys feliceps	White seacatfish	85.4	3.7
	Ariommatidae	*Ariomma indica*	Indian driftfish	1.6	0.0
	Batrachoididae	*Chatrabus hendersoni*	Chocolate toadfish	0.8	0.0
		Chatrabus melanurus	Humpback toadfish	3.3	0.0
	Bothidae	*Arnoglossus capensis*	Cape flounder	26.8	0.0
	Callionymidae	*Paracallionymus costatus*	Dragonette	4.1	0.0
	Carangidae	*Carangoides equula*	Whitefin kingfish	0.8	0.0
		Pseudocaranx dentex	White kingfish	1.6	0.0
		Trachurus delagoa	African maasbanker	2.4	0.0
		Trachurus trachurus capensis	Horse mackerel	82.1	18.4
	Centracanthidae	*Spicara axillaris*	Windtoy	0.8	0.0
	Centriscidae	*Macroramphosus scolopax*	Longspine snipefish	2.4	0.0
		Notopogon macrosolen	Trumpet fish	1.6	0.0
	Centrolophidae	*Schedophilus huttoni*	Hutton's driftfish	0.8	0.0
	Chaetodontidae	*Chaetodon marleyi*	Doublesash butterflyfish	0.8	0.0
	Champsodontidae	*Champsodon capensis*	Lizardfish	9.8	0.0
	Cheilodactylidae	*Cheilodactylus fasciatus*	Red fingers	6.5	0.0
		Cheilodactylus pixi	Barred fingerfin	8.9	0.0
		Chirodactylus brachydactylus	Twotone fingerfin	2.4	0.0
	Clupeidae	*Etrumeus whiteheadi*	Red-eye round herring	69.1	1.7
		Sardinops sagax	Sardine/pilchard	35	6.6
	Congiopodidae	*Congiopodus spinifer*	Spinenose horsefish	34.1	0.1
		Congiopodus torvus	Smooth horsefish	3.3	0.0
	Cynoglossidae	*Cynoglossus capensis* [24]	Sand tonguefish	4.1	0.0
		Cynoglossus zanzibarensis	Sandrat	76.4	0.2
	Emmelichthyidae	*Plagiogeneion rubiginosus*	Rubyfish	0.8	0.0
	Engraulidae	*Engraulis encrasicolus*	Anchovy	36.6	1.1
	Fistulariidae	*Fistularia petimba*	Flutemouth	7.3	0.0
	Gonorynchidae	*Gonorhynchus gonorhynchus*	Beaked sandfish	18.7	0.0
	Haemulidae	*Pomadasys olivaceus*	Piggy	14.6	0.2
	Merlucciidae	*Merluccius capensis*	Shallow-water hake	87	8.5
	Moridae	*Physiculus capensis*	Cape codlet	1.6	0.0
	Nomeidae	*Cubiceps capensis*	Cape flathead	0.8	0.0
	Ogcocephalidae	*Halieutaea fitzsimonsi*	Circular seabat	0.8	0.0
	Ophidiidae	*Genypterus capensis*	Kingklip	34.1	0.2
	Parascorpididae	*Parascorpis typus*	Jutjaw	1.6	0.0
	Peristediidae	*Satyrichthys adeni*	Armoured gurnard	1.6	0.0
	Pomatomidae	*Pomatomus saltatrix*	Elf	38.2	0.4
	Priacanthidae	*Priacanthus hamrur*	Crescent-tail bigeye	2.4	0.0
	Sciaenidae	*Argyrosomus sp.*	Kob	64.2	1.0
		Atractoscion aequidens	Geelbek	26	0.1
		Umbrina canariensis	Baardman	44.7	0.6

[24] Possible under-recording due to misidentification with *C. zanzibarensis (R.W.Leslie pers. comm.)*

Taxa	Family	Scientific name	Common name	Frequency of occurrence (FoC) (%)	% of catch (biomass)
	Scombridae	*Katsuwonus pelamis*	Skipjack tuna	4.9	0.0
		Sarda orientalis	Striped bonito	0.8	0.0
		Scomber japonicus	Mackerel	38.2	0.6
	Scombropidae	*Scombrops boops*	Gnomefish	5.7	0.0
	Scorpaenidae	*Helicolenus dactylopterus*	Jacopever	2.4	0.0
		Scorpaena scrofa	Bigscale scorpionfish	0.8	0.0
		Sebastes capensis	Cape scorpionfish	1.6	0.0
	Serranidae	*Acanthistius sebastoides*	Koester	4.1	0.0
		Serranus cabrilla	Comber	1.6	0.0
	Soleidae	*Austroglossus pectoralis*	East coast sole	94.3	0.9
	Sparidae	*Argyrozona argyrozona*	Carpenter	28.5	0.3
		Boopsoidea inornata	Fransmadam	1.6	0.0
		Cheimerius nufar	Santer	9.8	0.0
		Chrysoblephus gibbiceps	Red stumpnose	0.8	0.0
		Chrysoblephus laticeps	Roman	3.3	0.0
		Diplodus sargus capensis	Blacktail	0.8	0.0
		Gymnocrotaphus curvidens	Janbruin	0.8	0.0
		Lithognathus lithognathus	White steenbras	1.6	0.0
		Lithognathus mormyrus	Sand steenbras	0.8	0.0
		Pachymetopon aeneum	Blue hottentot	14.6	0.1
		Pachymetopon blochii	Hottentot	0.8	0.0
		Pachymetopon grande	Bronze bream	0.8	0.0
		Pagellus natalensis	Red tjor-tjor	61	11.6
		Pterogymnus laniarius	Panga	56.9	7.1
		Rhabdosargus globiceps	White stumpnose	31.7	0.1
		Rhabdosargus holubi	Cape stumpnose	1.6	0.0
		Spondyliosoma emarginatum	Steentjie	35.8	0.1
	Sphyraenidae	*Sphyraena acutipinnis*	Sharp-fin barracuda	2.4	0.0
		Sphyraena chrysotaenia	Yellowstripe barracuda	0.8	0.0
		Sphyraena flavicauda	Yellowtail barracuda	1.6	0.0
	Stromateidae	*Stromateus fiatola*	Blue butterfish	4.9	0.0
	Syngnathidae	*Syngnathus acus*	Longsnout pipefish	2.4	0.0
	Synodontidae	*Saurida undosquamis*	Largescale lizardfish	0.8	0.0
		Synodus indicus	Indian lizardfish	0.8	0.0
		Synodus variegata	Variegated lizardfish	0.8	0.0
	Tetraodontidae	*Amblyrhynchotes honckenii*	Evileye blaasop	18.7	0.0
		Arothron hispidus	Whitespotted blaasop	0.8	0.0
		Arothron immaculatus	Black-edged blaasop	0.8	0.0
		Arothron meleagris	Guineafowl blaasop	0.8	0.0
		Sphoeroides pachygaster	Blunthead blaasop	6.5	0.0
		Takifugu oblongus	Black spotted blaasop	0.8	0.0
	Tetrarogidae	*Coccotropsis gymnoderma*	Smoothskin scorpionfish	0.8	0.0
	Trichiuridae	*Lepidopus caudatus*	Ribbon fish	8.9	0.0
		Trichiurus lepturus	Cutlass fish	14.6	0.0
	Triglidae	*Chelidonichthys capensis*	Cape gurnard	82.1	2.5
		Chelidonichthys kumu	Bluefin gurnard	12.2	0.0
		Chelidonichthys queketti	Lesser gurnard	94.3	4.9
		Trigloporus lastoviza africanus	African gurnard	12.2	0.0

Taxa	Family	Scientific name	Common name	Frequency of occurrence (FoC) (%)	% of catch (biomass)
	Zeidae	*Zeus capensis*	Cape dory	22.8	0.0
		Zeus faber [25]	John dory	3.3	0.0
Cephalopoda	Loliginidae	*Loligo reynaudi*	Chokka-squid	94.3	2.0
		Lolliguncula mercatoris	Thumbstall squid	8.1	0.0
	Octopodidae	*Octopus vulgaris*	Common octopus	2.4	0.0
	Sepiidae	*Sepia australis*	Southern cuttlefish	45.5	0.0
		Sepia hieronis	Unspined cuttlefish	4.1	0.0
		Sepia papillata	-	17.1	0.0
		Sepia simoniana	-	20.3	0.0
		Sepia tuberculata	-	0.8	0.0
		Sepia typica	-	4.9	0.0
		Sepia vermiculata	Common cuttlefish	7.3	0.0

[25] Possible misidentification with *Z. capensis* (R.W.Leslie *pers. comm.*)

APPENDIX 3: ROVING CREEL SHORT QUESTIONNAIRE

GREATER ADDO ROVING CREEL SURVEY – SHORT QUESTIONNAIRE

Group Number: ______ **Name of Interviewer:** ________________

Time: ____:____ **Date:** ___/___/___ **Day of week:** ____________ **Public holiday:** Yes ☐ No ☐

Location: S ______________________ E ______________________

ZONE

1. Boknes – Cannon Rocks	☐
2. Cannon Rocks – Cape Padrone	☐
3. Cape Padrone – Woody Cape	☐
4. Woody Cape – Sundays River	☐
5. Sundays River – Coega	☐

Nearest Access point: ________________ **Transport:** 4x4 ☐ Vehicle ☐ Walking ☐ other __________

Name: (Confidential): ______________ Surname: ________

Race group: Black ☐ White ☐ Coloured ☐ Asian ☐ Other ______________

Age: _____ **Sex:** Male ☐ Female ☐

Home language: English ☐ Afrikaans ☐ Xhosa ☐ Other ______________

Place of residence: ______________________ **Occupation:** __________________________

Do you have a valid fishing/bait collecting permit? Yes ☐ No ☐
Recreational ☐ Subsistence ☐

What type of permit? Angling ☐ Cast net ☐ Bait/Molluscs ☐ Rock lobster ☐ Spearfishing ☐ Mud crab ☐

Do you have the permit with you? : Yes ☐ No ☐ **Permit number:** __________________

CATCH INFORMATION

# fishing	# rods	Time start	Time of interview	Expected time finish
		:	:	:

FISHING INFORMATION

Species for angler/group above	Length (mm)	Gear	Bait	Released Y/N	Substrate (Rock; Sand; Rock/sand; Sand/rock)

BAIT INFORMATION

Bait organism	Method used	Number of collectors	Number / quantity	Place obtained	Cost

LONG QUESTIONNAIRE COMPLETED YES ☐ NO ☐

APPENDIX 4: ROVING CREEL LONG QUESTIONNAIRE

GREATER ADDO ROVING CREEL SURVEY QUESTIONNAIRE

Group Number: _______ **Name of interviewer:** ___________________

Date: _________ **Day of week:** _____________

SOCIO-ECONOMIC INFORMATION

First Name: (Confidential): _____________ **Surname:** _________________

Highest educational qualification: ___________________________________

Occupation: ____

Total number people in household (including you)? _______ **How many dependants?** _______

What other sources of income in your household?

None ☐ Fixed employment ☐ Pension ☐ Casual labour ☐ Other _________________

How many *other* people in your household fish? ____________

How important is your fish/other catch in your household diet?

Crucial ☐ Fairly important ☐ Not very important ☐

How many times per week does your household eat meat (red & chicken)? ________

Permanent place of residence: _____________ **Postal code:** ______________

Are you on an: day ☐ overnight ☐ weekend ☐ or longer/trip holiday ☐

IF NOT A DAY TRIP THEN:

Approximate distance traveled one way to destination: ____________________

Where are you staying? Town ________________ Type (B&B; family, friend etc) ______________

How much is it costing for you to stay there (whole group)? ______________

Description of cost ____

What method of transport did you use? _________________________

Total number of people in group? ____________

How many days is your trip? __________________

How many people will fish during the stay? _____________

How many days will you fish? _________________

RESOURCE USE INFORMATION

Why do you fish? For Food ☐ To earn living ☐ Recreation ☐ Competition ☐ Other ____________

What do you generally do with your fish catch?

	All	Some	Minimal	None
Eat it				
Sell it				
Give it away				
Return it alive				

Do you utilize other marine organisms? Yes ☐ No ☐

Besides fish what other marine organisms do you utilize from this area?

	Eat	Bait	Sell	Method/Implement used	Where obtained
Alicrekel					
Chitons					
Perlamoen					
Seaweeds					
Octopus					
Crabs					

When do you prefer to fish? Morning ☐ Midday ☐ Afternoon ☐ Evening ☐ Night ☐ High ☐ Low ☐ Spring ☐ Neap tides ☐ Doesn't matter ☐

Do you fish more over? Weekdays ☐ Weekends & public holidays ☐ During fish runs ☐ School holidays ☐ Everyday ☐ Anyday ☐

How many days have you been fishing/Bait collecting in the last: week _____ month _____ 12 months _____

How much of this time (%) is spent between Boknes & Coega? _________

Do you fish at night? Yes ☐ No ☐

If you fish at night how often do you fish at night: in this area (%)_________ in other areas (%)_______

How far have you traveled to fish today (one-way)? ________________

What method of transport did you use? _________________________

How many years have you been fishing for? ____________

Which areas do you fish most often now__________ in the past__________

Do you belong to a fishing club? Yes ☐ No ☐ **Name:**______________________

How much money did you spend for this trip? Bait_____ Tackle _______ Fuel _______

Food/refreshments _______ Gillies _______

How much are you prepared to pay to go on a fishing trip like this one? _______

What is the maximum you are prepared to spend on bait for a days fishing? ________

How much have you (*the person being interviewed*) spent on tackle in the last month?______________

How much have you spent on rods & reels in the last 12 months? ______________

What is the estimated value of all your angling equipment? Rods ________ Reels _________ Tackle ________

For which fish/bait species do you often and never catch your bag limit & for which species are the legal limits insufficient?

Often catch bag limit	Never catch bag limit	Insufficient legal limits	
		Species (eg Kob)	Reason (eg bag limit)

OWNERSHIP & ACCESS TO THE LIVING RESOURCES

Who owns the living resources along the coastline?
All SA citizens ☐ the Anglers ☐ God ☐
The government ☐ People living in the area ☐ Ancestors ☐

How did you obtain the right to fish in this area?
By inheritance ☐ Local traditional chief ☐ MCM permit ☐ South African citizen ☐
SANParks ☐ Ancestors ☐ Gate entrance fee/permit ☐
Other ______________________

Did you previously have a 4x4 vehicle which you used for shore fishing? Yes ☐ No ☐

Has the ban of ORV's affected your fishing & beach use? Yes ☐ No ☐

In what way: Less fishing effort ☐ Same effort same areas ☐ Same effort different areas ☐
More effort same areas ☐ More effort different areas ☐

Is your use of the coastline now: More pleasurable ☐ Less pleasurable ☐

Are you in favor ORV on the beaches? Yes ☐ No ☐

Would you be in favour of a limited day access/permit system for vehicular beach access? Yes ☐ No ☐

How much would you be prepared to pay for such a license ____________

MANAGEMENT OF THE COASTLINE

Who do you think should be responsible for managing the living resources on this coastline?
National gov MCM ☐ Provincial gov nature Cons. ☐ Local Municipality ☐
Anglers ☐ Local residents ☐ Everyone ☐ Other ____________

Are you currently involved in management of the resource in any way? Yes ☐ No ☐
Informed IAP ☐ Consulted ☐ Enforcement ☐ Quota allocation ☐
Research & monitoring ☐ Angling clubs ☐ Other ____________

If not would you like to be involved? Yes ☐ No ☐
Informed IAP ☐ Consulted ☐ Enforcement ☐ Quota allocation ☐
Research & monitoring ☐ Angling clubs ☐ Other ____________

Which of the following regulations, in your opinion, are effective ways of managing our resources?
Minimum size limits ☐ Bag limits ☐ Closed seasons ☐ MPAs ☐

Are you familiar with these regulations? Yes ☐ No ☐

Do you obey these regulations? Yes ☐ No ☐ Frequently ☐ Infrequently ☐

Have you ever sold your catch? Yes ☐ No ☐

Do you buy bait from subsistence collectors? Yes ☐ No ☐

Would you buy bait from collectors if it was legal? Yes ☐ No ☐

Are you familiar with the regulations of your five target species? Yes ☐ No ☐

What are you 5 major target fish/bait species?

Species	Minimum size limit	Bag Limit	Closed Season	Preferred bait
1.				
2.				
3.				
4.				
5.				

How often has your permit been checked by an inspector?
Never ☐ 1 in 10 outings ☐ 1in 50 outings ☐ 1 in 100 outings ☐ less than 1 in 100 ☐

How often has your catch ever been inspected by a fisheries officer?
Never ☐ 1 in 10 outings ☐ 1in 50 outings ☐ 1 in 100 outings ☐ less than 1 in 100 ☐

How often do you encounter MCM or SANParks officials in this area?
Never ☐ 1 in 10 outings ☐ 1in 50 outings ☐ 1 in 100 outings ☐ less than 1 in 100 ☐

Are you happy with the current MLR permit system? Yes ☐ No ☐

If not why not? ____________

MAINTAINING BIODIVERSITY

In which way does your current fish catch differ from the past?
More ☐ Less ☐ More species ☐ Less species ☐
Bigger ☐ Smaller ☐ No difference ☐ Don't know ☐

In which way does your bait differ from the past?
More ☐ Less ☐ More species ☐ Less species ☐
Bigger ☐ Smaller ☐ No difference ☐ Don't know ☐

Which fish/bait species are noticeably scarcer than before?

Organism	Scarcer	More common	Smaller	Larger

Do you think the living marine resources are threatened? Yes ☐ No ☐

What are the main threats to the living resources of the coastline?
Agricultural pollution ☐ Industrial pollution ☐ Recreational fishing ☐
Subsistence fishing ☐ Commercial fishing ☐ Poor management/enforcement ☐
Poaching/illegal activities ☐

Have there been noticeable changes? ____________

What do you think has caused these changes? ____________

What do you think can be done to improve the situation? ____________

APPENDIX 5: ACCESS POINT QUESTIONNAIRE

ACCESS POINT SURVEY – SHORT QUESTIONNAIRE

Boat Number: ____________ **Boat Name:** ______________ **Name of interviewer:** ____________

Time: ___:___ **Date:** __/__/__ **Day of week:** ________ **Public holiday:** Yes ☐ No ☐

Recreational ☐ Commercial ☐ Charter ☐ *If Charter Cost/person:* _____ Other ______________

Skiboat ☐ Rubber duck☐ Deckboat ☐ Other ______________

Name: (Confidential): ____________ **Surname:** ____________ **Boat owner** ☐ **Skipper** ☐ **Crew member** ☐

Race group: Black ☐ White ☐ Coloured ☐ Asian ☐ Other ________ **Age:** ____ **Sex:** Male ☐ Female ☐

Home language: English ☐ Afrikaans ☐ Xhosa ☐ Other ________

Highest educational qualification: ____________ **Occupation:** __________ **Town of residence:** __________

Are you on an: Day ☐ Overnight ☐ Weekend ☐ or longer/trip holiday ☐

IF NOT DAY TRIP THEN
Type accommodation? (B&B, renting etc) ______________
Length of stay _____ Accommodation cost / day _____
Number days fishing _____ Number of fishermen in group _____
On a scale of 1-5 how important is skiboat fishing in terms of your holiday activities? _____

How far did you travel by vehicle to come fishing today? (km one-way) ______________

What method of transport did you use? Make ______ Model ______ How many people in the vehicle? ______

Do you belong to any fishing clubs? Yes ☐ No ☐ Name: ________

Do you fish more over? Weekdays ☐ Weekends & public holidays ☐ During fish runs ☐ School holidays ☐ Everyday ☐ Anyday ☐

How often have you fished from a ski-boat in last (estimated): week ____ month ____ 12 months ____

On average how often would you say you fish from a ski-boat days/ month? Summer _____ Winter _____

Is your vessel night rated? Yes ☐ No ☐ if yes how often have you been fishing at night in the past: week ____ month ____ 12 months ___

Do you know the regulations for your main target species? YES NO (ask main target species and then the regulations for each to test them)

Species	Time targeted (summer winter etc)	Minimum legal size	Bag limit	Closed season

Do you often reach your total daily bag limit? Yes ☐ No ☐

For which species: ________ ________ ________ ________ ________

What do you usually do with these fish? Keep & eat ☐ Give away ☐ Sell ☐ Bait ☐ other ________

What areas do you fish most often (Number MAP)? 1________ 2________ 3________
4________ 5________ 6________ 7________ 8________

How much money did you / do you spend on average / trip? Bait ______ Tackle ______ Boat Fuel ______
Food/refreshments ______ Gillies ______

What is the maximum amount you are prepared to pay for a day fishing trip (includes bait, drinks, fuels etc) ________

What is the estimated value of your equipment? (What you would sell it for)

Boat ________ Outboards ________ Trailer ________ Tow vehicle ________

Rods ________ Reels ________ Tackle ________

What do you spend on insurance, licensing, storage & maintenance of your boat per year? ________

How many years have you been ski-boat fishing for? ______

Overall do you think your catch has changed over the past 5-10 years? Yes ☐ No ☐ Don't know ☐

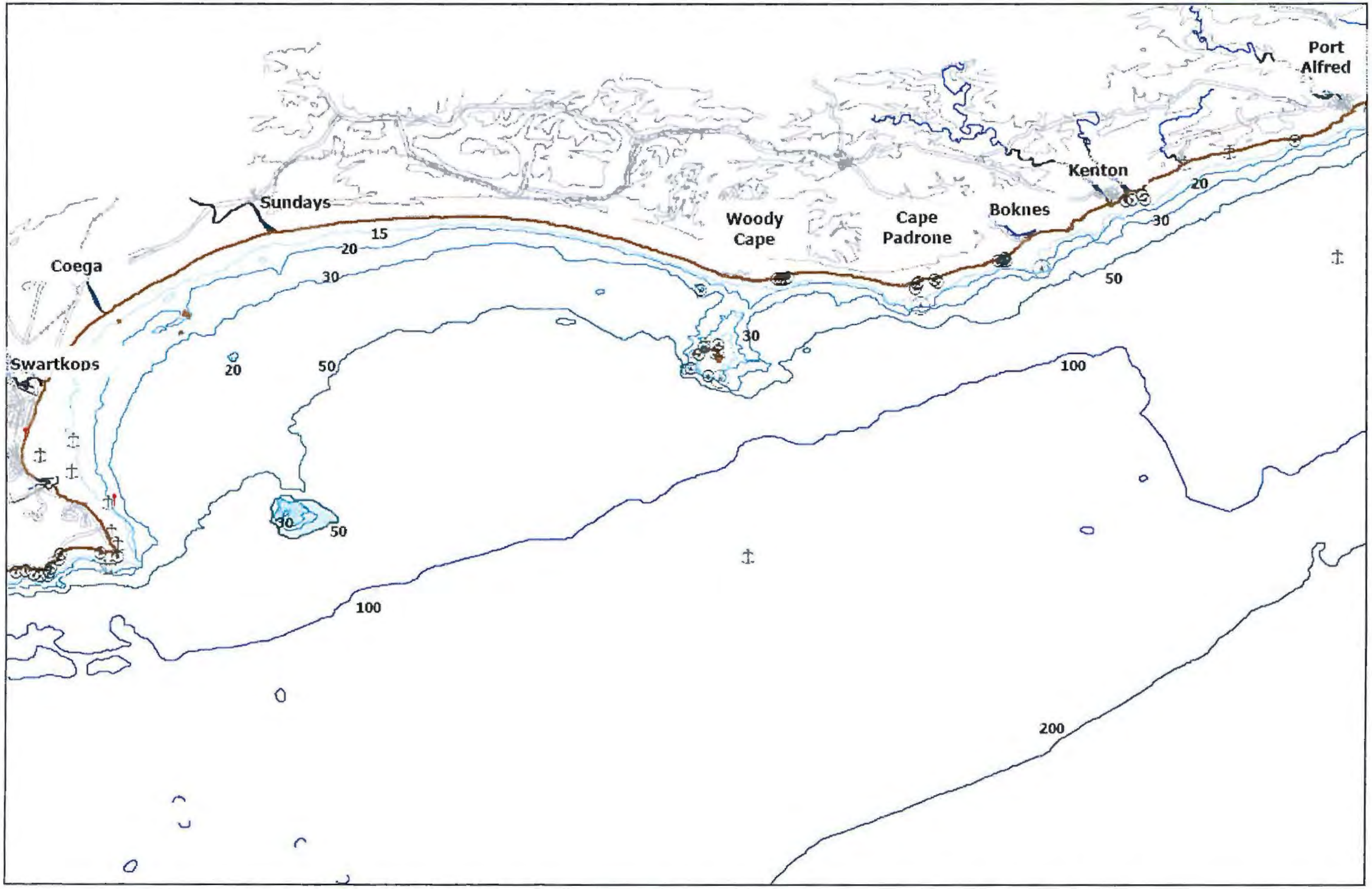
If yes, How? ______________________

How often has your permit and catch been checked by an inspector? Never ☐ 1 in 10 ☐ 1in 50 ☐ 1 in 100 ☐ less than 1 in 100 ☐

How often has your permit and catch been checked by an inspector over the last 12 months? ________

Which of the following regulations, in your opinion, are effective ways of managing our marine resources? Yes/No
Minimum size limits ☐ Bag limits ☐ Closed seasons ☐ MPAs ☐

Do you obey these regulations? Yes ☐ No ☐ Frequently ☐ Infrequently ☐


Port Alfred
Kenton
Boknes
Cape Padrone
Woody Cape
Sundays
Coega
Swartkops
15
20
30
50
100
200

APPENDIX 6: ACCESS POINT CATCH DATASHEET

Boat Interview

ACCESS POINT SURVEY – CATCH SURVEY

Date: ___/___/___ Time: ___:___ Boat Name: ________________ Registration #: _________ Length: ____

Commercial (Co) Recreational (R) Charter (Ch): _______ Rubberduck (R) Skiboat (S) Deckboat (D) _____

Start time: ____:_____ End time: ____: ____ Launch Site: _____________ Crew #: _____

 # Male: _______ # Female: _________ # White _____ # Black ____ # Coloured _____ # Other _____

OWNER NAME Occupation Age Town of residence

Areas (See on MAP) substrate (R;S;R/S) & depths fished:

1 Area _____________ Depth _____ Target species ____

2 Area _____________ Depth _____ Target species ____

3 Area _____________ Depth _____ Target species ____

4 Area _____________ Depth _____ Target species ____

5 Area _____________ Depth _____ Target species ____

6 Area _____________ Depth _____ Target species ____

Hook size: _________Hook type: C J

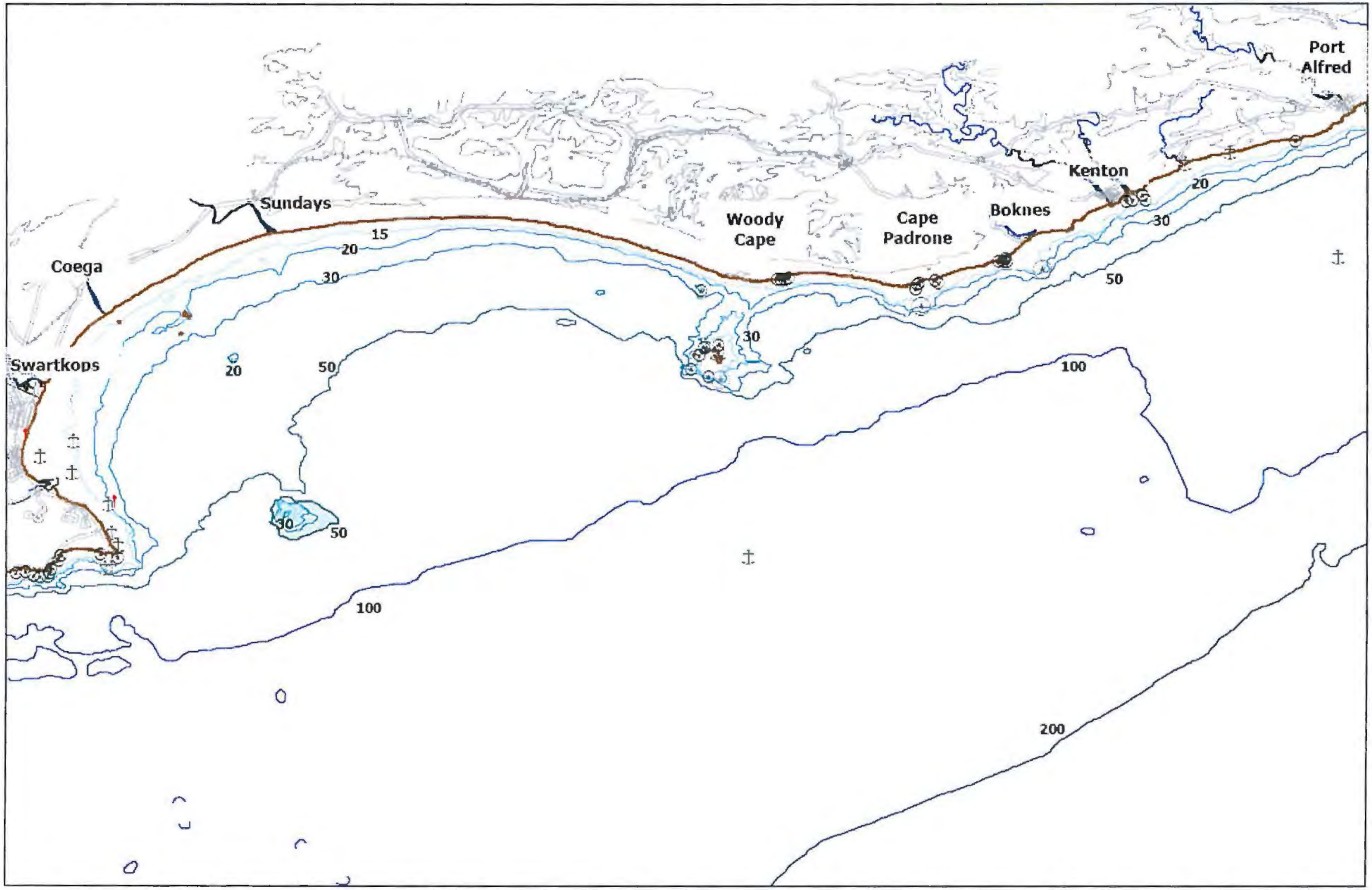
Hook size: _________Hook type: C J

Hook size: _________Hook type: C J

Hook size: _________Hook type: C J

Bait: Pilchard Chokka other ____

	Species	Total kept	Total released				Lengths	Weight
1			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
2			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
3			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
4			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
5			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
6			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
7			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
8			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
9			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
10			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
11			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
12			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
13			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
14			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
15			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
16			1	2	3-5	5-10		
			10-15	15-20	20-30	>30		
17			1	2	3-5	5-10		


Port
Alfred
Kenton
Boknes
Cape
Padrone
Woody
Cape
Sundays
Coega
Swartkops
15
20
30
50
100
200

APPENDIX 7: CONFERENCE PRESENTATIONS

Chalmers, R., Götz, A., Sauer, W.H.H. and Holness, S. 2009. *Coastal bays, MPAs and fisheries – trying to balance conservation and socio-economic objectives.* Presented at the Western Indian Ocean Marine Science Association (WIOMSA) Symposium, 24-29 August 2009, Saint Denis, Reunion.

Chalmers, R., Götz, A. and Sauer, W.H.H. 2008. *Development of a spatially based conservation and management plan for the Addo Elephant National Park Marine Protected Area.* Presented at the Southern African Wildlife Management Association Symposium. Biodiversity Conservation: The Science Management Interface, 16-19 September, Mpekweni South Africa 2008.

Chalmers, R., Götz, A. and Sauer, W.H.H. 2008. *Strategic planning for the Greater Addo MPA – Understanding the key issues.* Presented at the South African Marine Science Symposium (SAMSS) 2008: Our changing seas, 29 June – 3 July, Cape Town South Africa.

Chalmers, R. and Götz, A. 2007. *Development of a long-term monitoring protocol for marine biota in the proposed Greater Addo Marine Protected Area (MPA).* Presented at the 1st SAEON Student Symposium, 11-13 September 2007, Cape Town, South Africa.

Chalmers, R., Götz, A. and Sauer, W.H.H. 2007. *Assessment of the ichthyofaunal and macro-benthic community structure in the proposed Greater Addo Marine Protected Area (MPA): Experimental design and preliminary results.* Presented at the Western Indian Ocean Marine Science Association (WIOMSA) Symposium, 27-31 October 2007, Durban, South Africa.