OPTIMISATION OF A SAMPLING PROTOCOL FOR LONG-TERM MONITORING OF TEMPERATE REEF FISHES

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ABSTRACT

Marine Protected Areas (MPAs), the Ecosystem Approach to Fisheries management (EAF) and Integrated Coastal Management (ICM) have been identified as possible alternatives to traditional linefish management measures, which have largely failed. Monitoring and assessment of fish communities on a long-term basis is necessary, and will provide a means to evaluate the effectiveness of such management measures. Therefore, standardised protocols and optimal sampling methods for long-term monitoring (LTM) and assessment of coastal fish communities are essential.

This study aimed to identify suitable methods and develop a protocol for assessment of inshore reef fish communities.

A suitable location for evaluation of proposed methods was identified in the warm temperate biogeographical region of South Africa, encompassing the well-established Tsitsikamma Coastal National Park MPA and an adjacent exploited area. *Chrysoblephus laticeps* (roman) was identified as an indicator species for the study, as it has been well-studied and is well represented in the area.

Underwater visual census (UVC) and controlled fishing were identified as suitable methods. UVC transects were found to be superior to point counts, in terms of sampling efficiency, variability, bias and required sample size. An effort of two angler hours per fishing station was shown to provide low catch variability, while at the same time a representative catch and low overall cost and required time. The methods were incorporated in a proposed sampling protocol, and evaluated. The methods were able to detect known differences between protected and exploited communities. It is

recommended that LTM within protected areas, for detection of natural change, be focused on community-level indicators, while LTM in exploited areas, aimed at detection of anthropogenic change, be focused on species-level indicators.

The proposed protocol with standardised methods will allow for comparisons across a network of LTM sites and provide the opportunity for a broad-scale assessment of the effects of environmental variables on reef fish stocks.

The protocol developed in this study has application in other biogeographical regions in South Africa, and other parts of the world. Shift in the focus of much marine research, in South Africa and elsewhere, to LTM, highlights the relevance and timeous nature of this study.

CONTENTS

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Chapter 1
General Introduction

1.1 Fish stocks and fisheries management

Increases in human population size, dependency on marine resources and technological advancements in fishing gear have resulted in steady increases in fishing pressure over the last century. Consequently, an estimated 52% of world fish stocks are fully exploited and 25% overexploited or depleted (FAO 2006). As world population and the demand for marine resources increase, the problems of declining fish stocks are only going to worsen (Caddy and Griffiths 1995).

In South Africa, increases in fishing effort and inefficient regulations have resulted in a steady decline in fish stocks (Sauer *et al*. 2003) and the collapse of most linefish stocks (many of which are reef-associated), with the commercial extinction of *Polysteganus undulosus* (seventyfour). Analysis of catch-per-unit-effort (CPUE) data from the linefishery showed that catches are less than 10% of those reported at the beginning of the twentieth century (Griffiths 2000), and in 2000, the linefishery was declared to be in a state of crisis (Sauer *et al*. 2003).

Fishing can affect biomass (Buxton and Smale 1989), stock size structure (Roberts and Polunin 1991, Buxton 1993a), growth rate (Buxton 1987, Russ 1991), genetic diversity and population dynamics, at the species level, and community structure, species composition and biotic diversity at the ecosystem level (Buxton and Smale 1989, Burger 1990). Buxton (1993c) suggests harvesting of such species is analogous to harvesting capital (standing stock), as opposed to harvesting interest (surplus production) as in many of the pelagic fisheries. The effect is greatest on reef species, because of late attainment of sexual maturity, lower natural mortality and greater longevity (Buxton 1993a). As a consequence, they are affected by lower levels of fishing effort, are less resilient to such impact and recovery is slower than in species with more r-selected life histories (Russ 1991). Removal of larger, more predatory species is likely to affect biological interactions, such as predator-prey and competition interactions and may allow entry of smaller, less valuable species

(Hoggarth *et al*. 2006), referred to as "fishing down the food chain" (Pauly *et al*. 1998). Although the effects of fishing on fish stocks have been well documented (e.g. Russ 1991, Smith *et al*. 1991, Buxton 1993a, 1993c, Caddy and Griffiths 1995, Levin and Grimes 2002), monitoring of population dynamics is essential to provide an understanding of the magnitude and implications of these effects (Buxton 1993b).

Conventional management strategies, such as closed seasons, daily bag and minimum size limits, are "not practical to enforce" (Attwood and Bennett 1994) and, as a result, have failed to sustain many reef-associated fishes (Kerwath 2005). This failure of traditional management measures has brought about the need for the use of complementary management measures in conjunction with traditional bag and size limits, and the need to manage marine resources from an ecosystem perspective (Cochrane *et al*. 2004). Having realised that there may be few viable alternatives fisheries managers in numerous countries have turned to marine protected areas and Ecosystem Based Management (EBM) for management and protection of reef fish stocks (Roberts and Polunin 1991, DeMartini 1993, Penney *et al*. 1999, Russ 2002).

Marine protected areas (MPAs) have been advocated by numerous fisheries biologists (*inter alia* Buxton 1993a, Attwood and Bennett 1995, Roberts 1998, Zeller *et al*. 2003, Hilborn *et al*. 2004, Mann *et al*. 2006) as a complementary tool to traditional management measures, and an important tool for the protection of coastal and marine resources, particularly invertebrates and reef associated linefish species (Britz *et al*. 2001). MPAs can provide control or reference areas, against which exploited areas may be compared to assess the impacts of fishing or protection on population parameters (Griffiths and Wilke 2002, Hilborn *et al*. 2004). The numerous biological benefits of MPAs may include decreased fishing mortality (Russ 1991), facilitation of recovery of depleted stocks (Beger *et al*. 2003), enhancement of stocks within the MPA through direct protection (Bennett and Attwood 1991, Millar and Willis 1999), spillover of adults to adjacent fished areas (Bennett and Attwood 1991, Zeller *et al*. 2003), seeding of recruits into adjacent fisheries through larval dispersal (Tilney 1993, Tilney *et al*. 1996), increased biomass and size structure within the reserve (Buxton 1987, Buxton and Smale 1989, Russ 1991, Roberts and Polunin 1991, Willis *et al*. 2000), increased reproductive capacity, insurance against recruitment or management failure, prevention of bycatch and high-grading and protection of habitat (Gell and

Roberts 2003). MPAs should not, however, be seen as "a panacea for fisheries management problems", but rather as a complementary or alternative measure to traditional management tools, to be used as "one element in a broader package of measures" (Hilborn *et al*. 2004). Even after the establishment of an MPA, conventional management measures should remain in place in the adjacent exploited areas (Russ 2002). This is particularly important in situations where fishing effort becomes concentrated at the edges of the MPA.

Integrated Coastal Management (ICM) has also been advocated as a complementary measure to traditional fisheries management. ICM strives to maximise social and economic benefits, while focusing on optimal, rather than maximum, resource utilisation (Smith 2005), through local knowledge and stakeholder participation in decision-making. This is particularly important for the management of fisheries where resource use takes place at all socio-economic levels (i.e. subsistence, recreational and commercial levels). In highly urbanised areas, ICM may be complemented by implementation of smaller-scale subsidiary management plans, which may include codes of conduct for resource users and zonation of areas for different resource uses. For example, in Plettenberg Bay, South Africa, the development of such a Bay Management Plan (BMP) has been initiated, through collaboration with provincial and local government, local non-governmental conservation organisations, resource users and stakeholders from all socio-economic levels, to ultimately strive for sustainable resource use, with optimal social and economic benefits (Smith 2005).

The World Summit on Sustainable Development (WSSD) held in Johannesburg in 2002 encouraged the ecosystem approach to fisheries management to be implemented by 2010 (Turrell 2004). The Ecosystem Approach to Fisheries (EAF) is a form of fisheries governance framework, drawing from conventional fisheries management and EBM principles (FAO 2003, Garcia *et al*. 2003). The basis of EAF is the management of fishery resources with specific goals, to allow for the sustainable use of resources and meet the needs of the users, while maintaining the ecosystem complexity, interactions and processes necessary for conservation of proper ecosystem functioning (Garcia *et al*. 2003). In South Africa, a dedicated EAF Working Group oversees EAF progress and related issues (Shannon *et al*. 2006). Most research programmes in South Africa have been conducted on a short-term basis, and

such data cannot answer long-term ecological questions (Biggs *et al*. 1999). Long-Term Ecological Research (LTER) sites have therefore been promoted to improve our understanding of ecosystem function and strengthen ecological early warning capabilities (van Jaarsveld and Biggs 2000).

MPAs, ICM and EAF can, therefore, provide possible solutions to the failure of conventional fisheries management. However, there is a need for an assessment tool for collection and assessment of fishery-independent data, to be used in conjunction with, and to assess the effectiveness of, such management measures. Effective management requires baseline information on important fishery species, which requires, among other information, accurate estimates of population abundance and natural temporal variability (Zeller and Russ 2000). However, collection of data for such assessments can be complicated, expensive and often laborious (Die 1997). This is especially apparent in the South African linefishery.

1.2 Fishery assessments and monitoring

The complexity and multispecies nature of the South African linefishery makes data collection difficult and expensive. Management through conventional management measures and assessment through standard single-species production models, respectively, are therefore not suitable. Information on the life-histories of many South African linefish species was not available until recently, and for many species is still unavailable (Sauer *et al*. 2003). A major problem with multispecies fisheries is that of bycatch of non-target species (Caddy and Cochrane 2001) or species for which fishers or vessels hold no rights, which may constitute a large proportion of the total catch. A considerable volume of this bycatch, some of which may be of commercial value, is therefore not recorded or discarded at sea (Attwood *et al*. 1997). Furthermore, the linefishery is characterised by a high number of users, including full-time commercial components and shore-based, estuarine and boat-based recreational components, making enforcement, management and collection of catch and effort data difficult (Sauer *et al*. 1997). The high number of access points and a wide operational range of the fisheries further complicate these tasks (Sauer *et al*. 2003).

While it is widely understood that fishing has deleterious effects on fish stocks, there is often little data available on the pre-exploitation levels of most stocks (Baum and Myers 2004). Assessment of fish stocks relative to pristine levels requires knowledge of the stocks in their unexploited state. In the absence of this data, management must be based on stock levels that are not pristine, and may therefore be misleading (Myers and Worm 2003). This has become known as the 'missing baseline' problem. The magnitude of the problem increases as stocks become further depleted, and management is based on ever-decreasing estimated pristine levels, referred to as the 'shifting baseline syndrome' (Pauly 1995, Baum and Myers 2004). One of the main arguments for MPAs is that they provide a pristine control, against which exploited areas can be compared (Attwood *et al*. 1997, Gell and Roberts 2003). However, without pre-exploitation catch and effort data, the level of catch that constitutes pristine remains unknown (Bell *et al*. 1987). Furthermore, in South Africa, the establishment of reserves has often been determined by political or social pressures rather than scientific knowledge of ecosystem dynamics or biological requirements of target species. Therefore, there is often insufficient monitoring of populations in the area before closure, providing little 'before closure' data for comparison with 'after closure' data, for the assessment of reserve effectiveness (Millar and Willis 1999).

Fishery-independent assessment of stocks in temperate areas is complicated by the nature of the environment. Rough seas limit the number of sea-going days, and strong surge, wave action and poor visibility make underwater visual census (UVC) difficult (Mann *et al*. 2006). Low visibility also limits the size of the UVC census area (Mann 1992). Temperature fluctuations and habitat heterogeneity make it difficult to standardise conditions across samples from different areas or times, increasing the likelihood of confounding factors when comparing results (Mann 1992). In contrast, conditions in tropical areas are less adverse and variable, simplifying data collection, particularly UVC, and allowing more time in the field (Ebeling and Hixon 1991).

Fishery assessments are further complicated by the numerous sources of natural variability. Annual variability in recruitment, spawning migration patterns and mortality, and variability in abundance associated with density-dependent population growth result in high levels of true population temporal variability (Sale and Douglas 1984, Ault and Johnson 1998, Cowley and Götz 2007). Such variability may be

reflected in estimates of stock abundance taken over short periods. Therefore, to account for these problems, it is important that stocks are monitored over suitably long periods (Cowley and Götz 2007). This has necessitated a move towards the use of long-term monitoring, for the assessment of fish stocks and detection of changes in abundance over time.

Long-term monitoring (LTM) programmes are essential for marine conservation planning and implementation (McKenna and Allen 2005). One of the main aims of LTM is the conservation of biodiversity. Instability caused by the loss of one or a few species may result in the loss of further species, or even the collapse of an ecosystem (Bond 1989). Changes in measured variables that indicate or reflect possible species extinction may be identified through LTM, so that remedial action can be taken in time, to prevent further change (Underwood 1991). LTM can also be used to assess the effectiveness of management measures set in place (Vos *et al*. 2000). Attwood *et al*. (1997) suggested that LTM is an essential part of MPA management and should be included in the management plan of all MPAs.

Extractive resource use, such as fishing, bait-collection and shellfish harvesting, is one of the greatest anthropogenic impacts on the coastal marine environment. In addition, coastal construction, coastal mining, coastal industry, introduction of exotic species (Lombard *et al*. 2004), tourism and recreation, coastal shipping, climate change and associated changes in sea temperatures (Crawford *et al*. 1990) are further factors that affect the marine environment and, indirectly, fish resources (Caddy and Griffiths 1995). LTM of the inshore marine environment, its key communities and the effects of anthropogenic and environmental factors that influence these communities is, therefore, essential to ensure the persistence and conservation of these resources, without complete prevention of extractive resource use.

Evidence of the effects of climate change on fisheries has been well documented (e.g. Tian *et al*. 2006, Herrick *et al*. 2007), further highlighting the need for LTM (Goodwin 2007). LTM can provide an understanding of how climate change affects ecosystems and fisheries, and can provide management with an early warning system for possible population decline (Goodwin 2007). Climate change is expected to affect abundance, location, migratory patterns and production of fish stocks (Castro-Ortiz

and Lluch-Belda 2007, Hannesson 2007), and will have implications for species whose life cycles are associated with estuaries, due to expected changes in rainfall, salinities and temperatures (Meynecke *et al*. 2006). Different ecosystems and fisheries will respond differently to climate change, suggesting that LTM is necessary in different ecosystems and biogeographical regions (Stenevik and Sundby 2007).

The South African Government has recognised the value and need for LTM. This has resulted in the establishment of the South African Environmental Observation Network[1] (SAEON). The Elwandle Node of SAEON, established in 2006, will witness the establishment of a network of coastal LTM sites across the country, which will require a standardised protocol for LTM and assessment of coastal fish resources, to allow comparison across monitoring sites.

The problems in the South African linefishery, the need for LTM and the development of a network of LTM sites (SAEON) highlight the need for the development of a sampling protocol for LTM and assessment of reef fish communities, incorporating standardised methods, to allow for comparisons between monitoring sites, and with other studies and programmes of a similar nature (Sutherland 1996b, ICES 2006).

1.3 Aims and objectives

The overall aim of this study was to identify the most suitable methods and develop a sampling protocol to be used as a tool for monitoring and assessment of inshore reef fish community structure. This was achieved by addressing the following objectives:

1. reviewing available methods and selecting those suitable for assessing reef fish communities,
2. identifying an area suitable for assessment of selected methods,
3. selecting optimal techniques, through practical evaluation and statistical comparison of UVC and CPUE techniques,
4. developing a proposed sampling protocol, and
5. evaluating the proposed protocol.

[1] http://www.saeon.ac.za

It must be noted that this thesis concentrates on sample design, and the results are examined with a view to the suitability of different methods and evaluation of the protocol. The layout of the thesis and focus of each chapter are given in Figure 1.1.

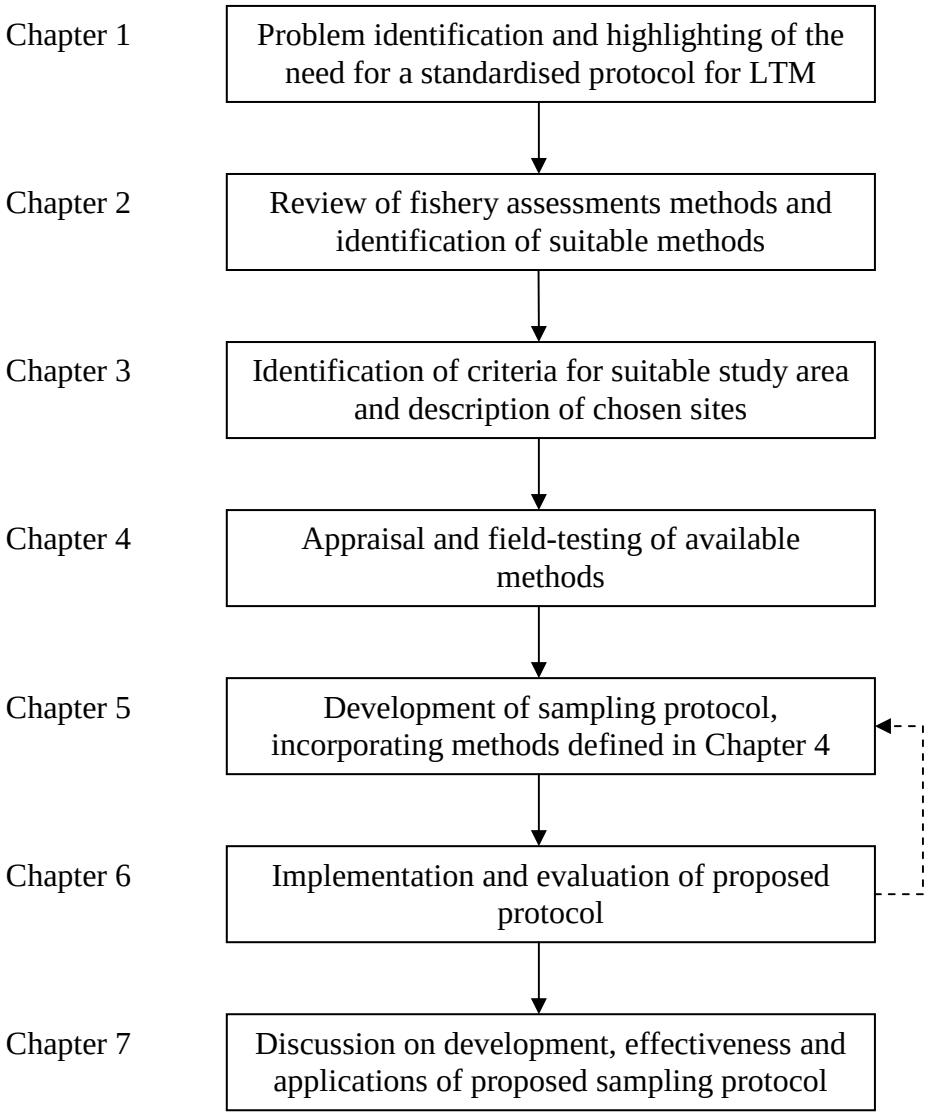

Figure 1.1: Flow diagram of thesis layout and the steps involved in the development of the sampling protocol for LTM and assessment of reef fish stocks. Each chapter was based on results from the previous chapter(s). The dotted arrow between Chapters 6 and 5 represents a feedback loop, through which possible changes in the proposed sampling protocol can be made.

Chapter 1 (General Introduction) outlines the status of fish stocks and problems with fisheries, and fisheries management, in South Africa. In this chapter, the difficulties associated with multispecies and temperate reef assessments are discussed. The lack of a standardised sampling methodology and the need for LTM and development of a suitable assessment tool are highlighted.

Chapter 2 (Review of Reef Fish Monitoring Methods) provides a review of available methods for monitoring reef fish communities, and discusses the associated problems.

Chapter 3 (Study Area) outlines the criteria required for a suitable study area and includes a description of the two chosen sites.

Chapter 4 (Comparison of Underwater Visual Census and Controlled Fishing Methods) provides an appraisal of available UVC and fishing methods for sampling temperate reef fish communities. Suitable methods were then chosen, tested in the field and compared statistically.

Chapter 5 (Development of a Sampling Protocol) incorporates the most suitable methods identified in Chapter 4, in the development of a proposed sampling protocol. The design process includes the determination of the sampling approach and methodology components, and defines the spatial and temporal scales of distribution of sampling effort.

Chapter 6 (Implementation and Evaluation of Proposed Protocol) includes a report on the implementation and results of the protocol, and a critical assessment of the methodology.

Finally, Chapter 7 (General Discussion) discusses the effectiveness and suitability of the sampling protocol, as well as the applications of the protocol outside of the study objectives. Recommendations for further research are also provided.

Chapter 2
Review of Reef Fish Monitoring Methods

Numerous methods are available for monitoring fish communities. It is important that the methods selected are appropriate to meet the objectives of, and provide the data required for, a LTM programme. Such data commonly include density or abundance, size structure and species composition. Methods must provide the lowest possible sampling variability, and be standardised to allow comparisons among different studies, monitoring programmes or monitoring areas. This required a review of available methods, to identify those appropriate for LTM. Furthermore, for monitoring and assessment of fish communities to be effective there are numerous associated problems that must be identified and overcome. The aim of this chapter was to review the suitability of available methods for, and problems associated with, reef fish monitoring and assessment.

2.1 Methods for monitoring fish communities

There are numerous methods used for assessing the status of fish stocks and determining the effects of fishing on fish resources. However, methods selected must be suitable to answer specific questions and meet the objectives of each study. It is important to identify suitable methods *a priori*, as changing sampling methods after a monitoring programme has been established negates the possibility of comparison of results from before and after the change (Sutherland 1996b, ICES 2006).

Data for fisheries assessments can be collected from the fishery (fishery-dependent data) or through controlled research surveys (fishery-independent data) (Samoilys and Gribble 1997).

2.1.1 Fishery-dependent data

Fishery-dependent data can be collected by the fishery, and are therefore cheap and can be collected for a long time-series (Penney *et al*. 1999). Such surveys provide information on catch, effort, gear types, fishing patterns and location of fishing grounds, which are important for understanding the impacts of fishing on the stocks

(Die 1997). Commercial fishery data can be obtained from commercial catch records (Crawford and Crous 1982, Penney *et al*. 1999) or trawl surveys (Griffiths 2000). However, such fishery-dependent data are limited to times and areas where the fleet fishes and effort is concentrated on specific fish groups (Penney *et al*. 1999) and areas of higher density (Saville 1977), and are, therefore, likely to be inaccurate (Die 1997). Furthermore, catch data are based on that recorded by the industry and may be inaccurate or untrustworthy (Die 1997). Recreational fishery data can be collected through roving creel surveys (Brouwer *et al*. 1997, Cowley *et al*. 2002), access point surveys (Brouwer and Buxton 2002) and daily catch cards (Hanekom *et al*. 1997, Penney *et al*. 1999). Roving creel surveys are suitable for collection of shore-based catch and effort data, but rely on reports of individual anglers (Brouwer *et al*. 1997). Access point surveys, suitable for assessment of skiboat catches, are inexpensive and allow measuring and accurate species identification, but provide no record of discarded catch (Brouwer and Buxton 2002). Catch cards are inexpensive and can provide large datasets, but are often inadequately completed and the accuracy of species identification is unknown (Hanekom *et al*. 1997). Fishery-dependent data are not available for protected areas, therefore preventing the possibility of comparison between protected and exploited areas (La Mesa and Vacchi 1999). Due to the nature of the fish processing onboard fishing vessels, catch data is commonly pooled by genus or family, or other groupings, such as the grouping of species of the family Sparidae into "redfishes" (Crawford and Crous 1982), making individual species assessments difficult or impossible.

2.1.2 Fishery-independent data

Fishery-independent surveys are more accurate and representative, as they allow even distribution of sampling effort over the study area as a whole (Die 1997), and are not biased by false recordings.

Destructive sampling

Destructive sampling techniques, including ichthyocides (Ackerman and Bellwood 2000, Willis 2001) and anaesthetics (Sayer *et al*. 1994), have been used in numerous studies to determine species composition, particularly in coral reef environments. Although such techniques are effective for assessing cryptic species, and provide greater species and family counts than non-destructive methods (Kulbicki 1998,

Willis 2001), they are non-selective, vary widely in effectiveness, provide inefficient sampling of highly mobile species (Buxton 1987) and cannot be used where sampling is to be repeated (Brock 1982, Buxton 1987), or in sensitive or protected areas (Thresher and Gunn 1986). Trawling is also commonly used in exploratory fishing to provide estimates of stock size (Kulbicki and Wantiez 1990, Swartzman *et al*. 1992, Francis 1995), and can be included in the destructive sampling methods, due to the damage caused to the substrate. Trawl surveys are expensive and time-consuming (in terms of vessel and manpower), and trawling cannot be conducted over areas of reef (Bodholt and Solli 1995). Due to high rates of mortality in the catch, trawl surveys are unsuitable for use in protected areas.

Acoustic surveys

Acoustic surveys are commonly used to provide abundance estimates for management (Tsimenides *et al*. 1995, Bodholt and Solli 1995), to detect changes in school size associated with exploitation (Azzali *et al*. 1995a, b, Reid *et al*. 2000), or in resource appraisal of virgin stocks (Saville 1977). However, the method is most suitable for offshore pelagic shoaling species and is unsuitable for demersal species, and provides no information on species composition (Saville 1977).

Underwater visual census

Underwater visual census (UVC) techniques have been used for estimation of reef fish abundance since the 1950s (Brock 1954). UVC is increasingly cited as a means by which to collect both qualitative and quantitative fishery-independent data on density (Buxton 1987, De Girolamo and Mazzoldi 2001), species diversity, species richness, length-frequency distributions (Brock 1982, Jennings and Polunin 1995, Kulbicki 1998), population dynamics, ecology (Samoilys 1997), community structure and behaviour (De Girolamo and Mazzoldi 2001). The advantages of this method have led to its use in numerous coastal, MPA and stock monitoring programmes (Samoilys 1997, Millar and Willis 1999, Barrett and Buxton 2002).

Kulbicki (1998) suggested that "UVC at present remains by far the best method available" for estimating density and biomass of reef fishes. The intimacy of direct observation affords researchers the opportunity to focus on particular species, and to assess habitat condition (e.g. siltation or anchor damage) *in situ*, whereas remote

methods may not. UVC techniques are relatively inexpensive (Watson and Quinn 1997), can provide a rapid assessment of relative abundance or density, and can be extrapolated to obtain estimates of absolute abundance (Sale and Douglas 1981). UVC can provide estimated length-frequency distributions (Barrett and Buxton 2002), and through the application of length-weight relationship models, can be used to estimate biomass (Jennings and Polunin 1995, Russ and Alcala 1996). UVC is particularly suitable when a long time series of data is unavailable, and the non-destructive nature of the method makes it suitable for use in LTM programmes, in particular those in protected or sensitive areas.

Some authors (Brock 1954, Brock 1982, Andrew and Mapstone 1987, Sale and Sharp 1983, Thresher and Gunn 1986, Kulbicki 1998) warn researchers that the validity of results from numerous UVC-based studies may be questionable. Inherent bias and sources of inaccuracy associated with UVC methods include underestimation of densities of cryptic (Kulbicki 1998, De Girolamo and Mazzoldi 2001) and highly abundant species (Richards and Schnute 1986), and between- and within-observer error (Watson and Quinn 1997). UVC is also limited by the depth and bottom time constraints of SCUBA (Samoilys 1997), and the distribution and behaviour of study subjects may be affected by the presence of the observer (Cowley and Naesje 2004). Therefore, Willis *et al*. (2000) suggested the use of a surface-tendered or remote sampling method in conjunction with UVC techniques, against which results of UVC may be compared or verified.

Underwater video assessment

Numerous studies have also made use of underwater video for assessment of reef fish populations, either in the form of stationary baited video cameras (Willis and Babcock 2000, Willis *et al*. 2000) or diver-based video transects (Alevizon and Brooks 1975, Potts *et al*. 1987, Parker *et al*. 1994). Technological advances have allowed the estimation of abundance and assessment of community structure through the use of remotely operated vehicles (ROVs) (Adams *et al*. 1995), manned submersibles (Langton and Uzmann 1989) and remote underwater cameras (Willis *et al*. 2000). However, these methods require expensive equipment and trained operators, and are not readily available, making them unsuitable for use in LTM programmes, particularly in developing countries, where finances for LTM are likely to be limiting.

Mark-recapture

Mark-recapture using hook-and-line fishing has been widely used to determine movement patterns of selected fish species (Attwood and Bennett 1994, Brouwer 2002, Cowley *et al*. 2002, Griffiths and Wilke 2002, Zeller *et al*. 2003), determine age and growth parameters (Buxton and Allen 1989, Cowley 2000), and estimate abundance (Parker 1990, Cowley and Whitfield 2001, Bergstedt *et al*. 2003) and capture probabilities (Ricker 1975). However, the method has been little used to estimate abundance of marine species, as certain assumptions of tag-recapture cannot realistically be satisfied (Thresher and Gunn 1986, Zeller and Russ 2000). Mark-recapture assumes that the population under study remains closed and does not experience recruitment, mortality, immigration or emigration (Cowley and Whitfield 2001); assumptions which may be unrealistic for reef fishes. Mark-recapture also assumes that all individuals have equal catchability (Thresher and Gunn 1986); however, due to the nature of hook-and-line fishing and hook size-selectivity, this assumption is also unlikely to be met (Buxton and Allen 1989, DeMartini and Lau 1999). The possibility of incidental mortality associated with tagging and tag-induced mortality make the technique less suitable for use in marine reserves, particularly for ongoing monitoring (Bell 1983, Willis *et al*. 2000). Tag loss is a further problem as this may bias results. Furthermore, to estimate abundance mark-recapture requires the collection of large datasets.

Mark-resighting

An additional technique that has been used to sample reef fish populations is tag-resighting, which makes use of conventional capture and tagging, but where ‘recaptures’ are made by underwater observation of tagged individuals (Zeller and Russ 1998, 2000, Chapman and Kramer 2000). However, as this method employs the use of tagging, many of the problems associated with mark-recapture are applicable also to mark-resighting. The method is also restricted by the depth and bottom time constraints of SCUBA.

Controlled fishing

CPUE fishing is commonly used in recreational and commercial fishery assessments to provide an index of abundance (Bannerot and Austin 1983), and is effective for use in LTM (Millar and Willis 1999, Attwood 2003). CPUE surveys provide an effective

means for monitoring temporal variability, particularly for assessment of the effects of fishing on fish populations, and assessment of the effectiveness of marine reserves, and have therefore been used in numerous such programmes (Bennett and Attwood 1993, Underwood 1991, Edgar and Barrett 1997, Zeller and Russ 1998, Millar and Willis 1999, Cowley *et al*. 2002).

Actual handling of the fish during research fishing surveys allows accurate measurement of length or weight (as opposed to estimation thereof during UVC), assessment of fish condition, recording of morphometric measurements and collection of DNA samples (finclips). Furthermore, the duration and depth sampled are not restricted by the constraints imposed by SCUBA, and fishing can be conducted at night, in conditions where it may be difficult to sample using other methods and under conditions of poor visibility (Perrow *et al*. 1996).

As hook size is highly selective, fishing surveys are suggested to provide skewed estimates of length-frequency distributions (Perrow *et al*. 1996), as fishing selects for larger individuals (Willis *et al*. 2000). However, fishing can provide a good representation of the length-frequency distribution of fishes available to the fishery i.e. “fish of harvestable size” (Zeller and Russ 2000). Fishing is also highly species selective, dependent on bait type and size, and cannot sample strictly corallivorous or herbivorous species that do not take baited hooks (Perrow *et al*. 1996). Descriptions of species composition are therefore better suited to UVC.

As an index, CPUE assumes constant catchability of individuals (Arreguin-Sanchez 1996). Buxton and Allen (1989) caution researchers that line fishing fails the assumption of equal catchability, as this may vary according to the level of fishing pressure or because of density-dependent competition for food, particularly in areas of high fish density (Millar and Willis 1999). CPUE performs best as an index of abundance when results are determined for single species (Richards and Schnute 1986). When captured, fish are subjected to stress and possible injury or incidental mortality as a result of barotrauma injuries or damage to the gills or viscera that may be caused by complete hook ingestion (Willis *et al*. 2000). However, hook-and-line fishing is relatively inexpensive, requires simple, inexpensive equipment and less skilled personnel, and a large sample size can be easily achieved. Importantly, the

method allows comparisons with other studies where fishing has been (or is being) used exclusively (Perrow *et al*. 1996). The principal advantages and disadvantages of the different methods are summarised in Table 2.1.

Table 2.1: Summary of advantages and disadvantages of each method.

Method	Advantages	Disadvantages
Fishery-Dependent		
Catch and effort data	Inexpensive[1] Long time series available[1]	Limited to times and areas of fishery[2] Catch for similar species pooled[3]
Roving creel surveys	Suitable for long areas of shoreline[4]	Reliant on angler reports[4] Data unavailable for MPAs[5]
Access point surveys	Allows accurate identification of species[6] Allows measurement of fish lengths	No record of discards[7] Data unavailable for MPAs[5]
Catch cards	Provide large sets of data[8] Inexpensive	Inadequately completed[8] Species identification problems[8]
Fishery-Independent		
Ichthyocides/ anaesthetics	Increased detection of cryptic species[9]	Highly variable effectiveness[10] Unsuitable for use in MPAs[11]
Trawling	Suitable for soft substrate benthic species Depth not constrained by SCUBA	Unsuitable for reef areas or MPAs[12] Extensive damage to the environment[13]
Acoustic surveys	Provides rapid assessment of stock size	Unsuitable for demersal species[2] No information on species composition[2]
UVC	Non-destructive, suitable for MPAs[14] Not size- or species-selective[15]	Constrained by SCUBA limitations[14] Diver presence may affect fish behaviour[16]
Underwater video/ submersibles	Depth may not be constrained by SCUBA Can revisit data in controlled environment	Requires expensive equipment Lower species estimates than UVC[17]
Mark-recapture	Can provide estimate of total mortality Can provide information on territoriality	Fails assumption of equal catchability[18] Tag associated and incidental mortality[19]
Mark-resighting	Same advantages as mark-recapture	Same disadvantages as mark-recapture Constrained by SCUBA limitations
Controlled fishing	Not constrained by SCUBA limitations[20] Allows exact measurement of lengths[20]	Size- and species-selective[19] Possibility of incidental mortality[19]

1 – Die (1997), 2 – Saville (1977), 3 – Crawford and Crous (1982), 4 – Brouwer *et al*. (1997), 5 – La Mesa and Vacchi (1999), 6 – Penney *et al*. (1999), 7 – Brouwer and Buxton (2002), 8 – Hanekom *et al*. (1997), 9 – Kulbicki (1998), 10 – Buxton (1987), 11 – Thresher and Gunn (1986), 12 – Bodholt and Solli (1995), 13 – Hixon and Tissot (2007), 14 – Samoilys (1997), 15 – Watson and Quinn (1997), 16 – Cowley and Naesje (2004), 17 – Tessier *et al*. (2005), 18 – DeMartini and Lau (1999), 19 – Willis *et al*. (2000), 20 – Perrow *et al*. (1996)

2.2 Problems with reef fish monitoring and assessment

This section reviews problems associated with data collection and analysis, for monitoring and assessment.

Pseudoreplication

Pseudoreplication is defined as "the use of inferential statistics to test for treatment effects with data from experiments where either treatments are not replicated (though samples may be) or replicates are not statistically independent" (Hurlbert 1984). Pseudoreplication results in a lack of independence of errors in each sample, and consequently prohibits us from knowing ά (alpha, the probability of a type I error), in which case interpretation may become subjective (Hurlbert 1984). Pseudoreplication may also lead to a decrease in statistical power, which is the probability of detecting a specified difference between treatments (Vos *et al*. 2000). In ecological experiments pseudoreplication is commonly due to the spatial distribution of samples not being independent (Hurlbert 1984). Consequently samples may appear more similar than they actually are, as a result of spatial autocorrelation (McArdle *et al*. 1990).

Lack of comparability

Assessments of biological resources between study areas or time periods often lack comparability (Willis *et al*. 2003). An example of such a study is provided by Sluka *et al*. (1994), in which the aim was to compare grouper densities between Exuma Cays Land and Sea Park (ECLSP), Bahamas and the Florida Keys National Marine Sanctuary (FKNMS). Results showed significant differences in density and length-frequency distribution of species between the two areas. However, sampling was conducted by snorkelling during summer (May/June) at ECLSP, and on SCUBA during winter (February) at FKNMS. The results lack comparability because sampling was conducted at different times (seasons) and with different methods in each study area. Therefore, it is disputable that the observed differences were the result of the different areas.

This may be of particular concern when comparisons are made across an MPA boundary, in which fishery-dependent data are collected from exploited areas, and

compared with fishery-independent data from within the MPA (Buxton 1987). The results of such comparisons may be confounded by factors such as (i) anglers in exploited areas targeting fishing spots known to offer higher catch rates, and (ii) differences in skill level between trained research anglers and recreational anglers. Examples of such comparison are given by Buxton (1993a) and Cowley *et al*. (2002). This lack of comparability suggests that comparisons should be restricted to data collected by fishery-independent controlled fishing only, from the protected and exploited areas (Attwood 2003).

A further problem is lack of comparability between study areas due to habitat type, depth or topographic complexity (Willis *et al*. 2003). In such cases, causal relationships drawn between fishing pressure and density or CPUE must be interpreted with caution. Such problems are common in comparisons made between a protected area, and an exploited area some distance away, in which habitat, area history or larval supply may be considerably different (Russ 2002).

Insufficient sampling

Malone (2003) suggested that one of the greatest problems in coastal monitoring programmes, particularly in the southern hemisphere, is undersampling. This results in low statistical power, which increases the probability of a type II error (i.e. not detecting a difference between treatments when a difference exists) (Cohen 1973). Such an error may be costly in an environmental monitoring programme, as the effect may only be detected once it is too late, or extremely costly, to rectify (Fairweather 1991). Furthermore, insufficient samples provide poor representation of community structure.

Use of a single method

Accurate estimates are critical in ecological studies of reef fish density; however, assessing the accuracy of a density estimate is complicated by an absence of a standard with which results from different methods can be verified (Thresher and Gunn 1986). In the absence of such a standard, the use of more than one method is advantageous for two reasons. Comparison of the results obtained from each method allows verification of results obtained from the other methods, which in turn provides insight into which methods may be more suitable (Haggarty and King 2006).

Variability

Although numerous studies have focused on spatial patterns of abundance, there are still a number of problems associated with the description and interpretation of temporal variation in reef fish communities (Thompson and Mapstone 2002). There are also numerous sources of variability, which may include real change in abundance (Thompson and Mapstone 2002), variation in the difference between true and estimated abundance (Stewart-Oaten *et al*. 1995), temporary, localised or small scale shifts in the distribution of abundance of individuals, sampling error and variable sightability or mobility when UVC methods are used (Thompson and Mapstone 2002). These sources of variability must be considered and eliminated wherever possible before making inference about differences in population abundance. LTM programmes aimed at detecting natural temporal change must therefore minimise sampling-associated variability, by incorporating methods that provide the lowest variability (Lohr 1999, Willis *et al*. 2000).

In order to monitor fish stocks effectively, it is necessary to distinguish between natural variability (i.e. change in abundance associated with environmental, climatic or oceanographic change) and change as a result of changes in fishing pressure or management regime (Garcia *et al*. 2003). Natural variability can be distinguished from that associated with fishing pressure, or changes in fishing pressure or management regime over time, by comparison of variability in abundance estimates between exploited areas (subject to natural and fishing-related variability) and protected areas (subject to natural variability only). Therefore, to detect fishing-associated variability over time, sampling protocols must be suitable for monitoring both protected and exploited areas.

The problems presented above have illustrated the need for the development of a standardised protocol for the purpose of assessment (Sakagawa 1995, Colvocoresses and Acosta 2007). The first step was to identify suitable methods, appropriate to meet the objectives of a LTM programme. As pointed out in the previous discussion, there are numerous methods available for assessing the state of fish stocks and the effects of fishing on fish communities. However, it appears that UVC and CPUE fishing are most suitable and were, therefore, included for assessment in this study.

Having selected suitable methods to be investigated, the next step was to identify a general study area in which to conduct the study. This also had to be appropriate for meeting the objectives of a LTM programme in the area, and allow for comparison of available methods and testing of the proposed sampling protocol (Turpie *et al*. 2000). It was then necessary to compare different techniques within each of the selected methods, so that the optimal techniques could be identified and included in the study. It was also necessary to select an appropriate indicator species.

Chapter 3
Study Area

3.1 Identifying a suitable study area

The inshore marine environment of the South African coastline is comprised of three distinguishable biogeographical regions (Fig. 3.1) (after Hockey and Buxton 1989). The subtropical region extends from Mozambique in the north-east, southwards approximately to Port St Johns on the south-east coast, and is characterised by high ichthyofaunal species richness, particularly of Indo-Pacific species (Turpie *et al*. 2000). The cool temperate region, characterised by low species richness, and particularly low endemic species richness (Turpie *et al*. 2000), extends from Cape Point in the south, northwards to Namibia. The warm temperate region forms the transition between these biogeographical regions, and is characterised by increasing species richness from west to east (Turpie *et al*. 2000). Within this region, overall species richness is intermediate of the subtropical and cool temperate regions, but with particularly high richness of southern and South African endemics (Turpie *et al*. 2000), many of which are important to the recreational and commercial line fisheries (Buxton 1993b, Attwood *et al*. 2002).

The widespread dissimilarities in oceanographic processes (Harris 1978) and ecological components (Branch and Branch 1981) among these three biogeographical regions suggest the need for the establishment of LTM sites within each region. Furthermore, to maximise geographic representivity, monitoring sites should be situated near the centre of each region (Turpie *et al*. 2000). Turpie *et al*. (2000) divided the South African coastline into 52 50-km sections, starting with section one at the Namibia border on the west coast, and ending with section 52 at the Mozambique border on the east coast, and suggested that conservation importance be focused in the centre of the warm temperate region, because of the high number of endemics in the area. The centre of the region falls within sections 26 and 27.

Monitoring, in which detection of natural temporal variability is envisaged, must include sites within protected and exploited areas, to allow comparison of exploited

area fish communities with those in nearby protected areas, to separate the effects of natural and fishing associated temporal variability. Therefore, the study area had to include a large reef complex, exploited by recreational and commercial line fisheries, which was in close proximity to a large, well-established MPA. Furthermore these areas had to be of similar depth, spatial extent, habitat type and distance offshore, to allow comparisons of fish abundance and community structure. The areas had to be large enough to allow comparison of methods and development and testing of sampling strategies. LTM study areas should be situated near the centre of the MPA, to minimise anthropogenic influences and allow monitoring of environmental change.

Within the warm temperate biogeographical region, an area was identified that included a large well-established MPA, and an adjacent exploited area for which there has been discussion of an integrated coastal management plan, including a Bay Management Plan, which may include the closure of an area to extractive resource utilisation (Smith 2005).

The Tsitsikamma Coastal National Park (TNP) MPA (Figs 3.1) is situated in section 27 of the coastline, in the centre of the warm temperate biogeographical region (after Turpie *et al*. 2000), and is the largest, and oldest, 'no-take' MPA in South Africa, and one of the largest single-unit 'no-take' MPAs in the world (SANParks 2007). The MPA was proclaimed in 1964 (proclamation 324 in Government Gazette 936 December 1964 and National Parks Act of 1962), and now protects 59 km (straight line distance) of the coastline, from the Groot River in the east to the Groot River in the west (Hanekom *et al*. 1997), and seven percent of the rocky shoreline of the warm temperate biogeographical region (Lombard *et al*. 2004). The MPA extends 0.8 km offshore between Groot River (west) and the Bloukrans River mouth, east of which it extends to 5 km offshore to a depth of approximately 100 m (Tilney *et al*. 1996), between the Bloukrans River and the Groot River (east), covering approximately 36 845 ha (Robinson and de Graaf 1994). The reef fish communities here are suggested to have recovered from the effects of extractive exploitation, and are assumed to be in pristine condition (Attwood *et al.* 1997). Due to its accessibility and geographic position in the centre of the biogeographical region, and the pristine condition of its fish stocks, the TNP was selected as a suitable location in this region for the establishment of a LTM study area for reef fish communities. Furthermore,

although there is a long-term shore-based fish monitoring programme in place in the TNP (Cowley *et al*. 2002), there is currently no off-shore reef fish monitoring programme, providing the opportunity for such a programme to be initiated.

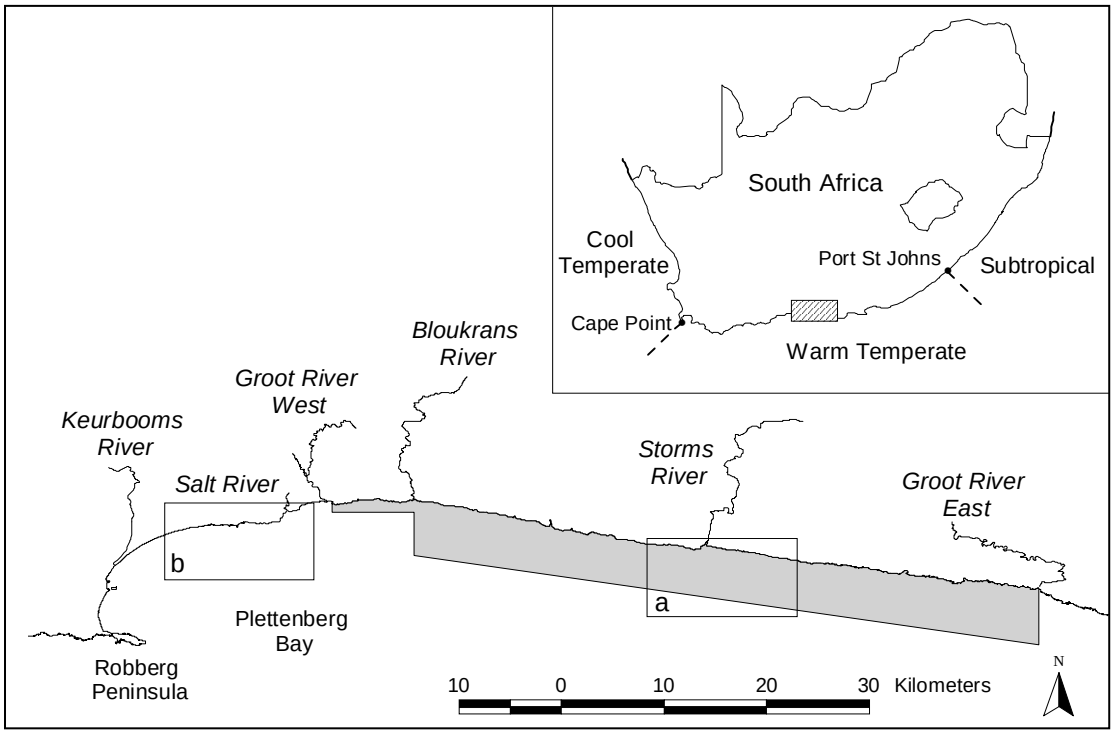

Figure 3.1: General study region, showing protected (a) and exploited (b) study areas. The shaded section represents the TNP MPA. The blocks labelled (a) and (b) show the positions and extents of Figures 3.2 and 3.3, respectively. The inset shows the three biogeographical regions of the South African coastline (after Hockey and Buxton 1989). The dashed lines represent approximate boundaries between biogeographical regions. The position of the general study area is indicated by the hashed block in the inset.

Approximately 3 km to the west of the western boundary of the TNP, is a large expanse of contiguous nearshore shallow reef, open to extractive resource use, including boat- and shore-based angling and spearfishing. This reef complex with boat launching facilities at Plettenberg Bay (PB) was the exploited study area chosen for the current study.

The protected and exploited study areas selected are in close enough proximity to be influenced by the same environmental and oceanographic phenomena, and for

biogeographical and habitat changes not to influence comparisons, but sufficiently spatially separated to be independent of one another (Stewart-Oaten *et al*. 1986). They differed only in the level of fishing intensity.

3.2 Description of study area

The coastline in the study area is characterised by several headlands, with associated bays (Martin and Flemming 1986), and is dominated by steeply shelving, exposed cliffs (Tilney *et al*. 1996). The shoreline is rugged, consisting of steep, rocky ridges, which extend into the subtidal (Hanekom *et al*. 1989), and is exposed to strong wave action (Cowley *et al*. 2002).

The east and south-east coasts of South Africa are dominated by the Agulhas Current, a typical well-defined south-westerly western boundary current. Inshore of this on the east coast, water movement over the continental shelf is influenced mainly by the wind, while off the south coast the shelf is wider, and dominated by south-westerly swell (Martin and Flemming 1986). Here, coastal trapped waves are the dominant process influencing net water movement (Tilney *et al*. 1996). As a result there is often an inshore counter current moving eastward along the south-east coast. There are two distinct current patterns in the area. During winter, water movement is predominantly longshore barotropic oscillation, generated by coastal trapped waves (Tilney *et al*. 1996), associated with a lowering of the thermocline (downwelling) and, commonly, increased visibility. During summer, water movement is dominated by baroclinic crossshore and longshore surface currents, associated with upwelling, decreases in sea surface temperature and decreases in visibility (Tilney 1993). Sea temperatures in winter remain relatively constant between 15 and 18° C, while those during summer range between 9 and 25° C, with decreases associated with easterly winds and increases associated with long periods of westerly winds (Hanekom *et al*. 1989).

The counter current inshore, together with longshore drift generated by wave action, transports sediment and plankton in a predominantly north-easterly direction (Schumann 1987). This eastward movement of inshore water, along with Ekman veering associated with easterly winds, results in net littoral movement along the

south coast in an easterly direction (Martin and Flemming 1986). As a consequence, sand and sediment are transported eastward past the Robberg Peninsula (Fig. 3.1) towards a mud depocentre in the middle of Plettenberg Bay. Inshore of this depocentre are bands of sandy mud, muddy sand, fine sand and fine-medium sand; the latter being texturally and compositionally indistinguishable from adjacent beach sands. This fine-medium sand is found along the coastline from the Robberg Peninsula eastwards to the eastern end of Nature's Valley beach, and is interspersed by patches of course sand and rock from Keurbooms Village to Nature's Valley (Fig. 3.3) (Martin and Flemming 1986, Smith 2005).

The Tsitsikamma coastal shelf has a smooth sediment surface in the east, but towards the west is dominated by a continuous bedrock terrace of low-relief rock, partially covered by unconsolidated sediments (Martin and Flemming 1986). Due to the dominance of coastal trapped waves, there is potential for temporal change in the spatial distribution of this unconsolidated sediment. Anecdotal evidence for this was provided by divers descending onto sand, on areas shown by side scan sonar (SSS) to be of rocky reef.

3.3 Sample site selection

Tsitsikamma

Suitable sites had to be identified within the protected and exploited areas, in which all sampling would take place. There is extensive literature suggesting that isolated reefs may not be representative of the respective study area (Ault and Johnson 1998), and may receive little or highly variable recruitment. Therefore, to be representative of a large number of habitats (profile and depth ranges) and to minimise the effects of chance disturbance, the sampled area of reef should be continuous with a greater reef complex (Ault and Johnson 1998). Suitable study areas were, therefore, identified on large expanses of contiguous reef within the protected (TNP) and exploited (PB) areas.

Within the TNP, areas of contiguous reef were initially identified using bathymetry and physiography data from SSS, captured by Schumann *et al*. (1982) and Flemming

et al. (1983). This data was digitised by scanning A4 hardcopies, and then georeferenced in ArcView 3.2 (Environmental Systems Research Institute). The bathymetry and physiography data were then converted to shape files using drawing tools in ArcView. The SSS was of low resolution and was, therefore, used only to determine the spatial extent of reef area within each site, rather than to provide accurate mapping of the seafloor. Therefore, to aid identification of apparently suitable sites, two additional methods were employed.

Areas of potentially suitable depth and substrate, determined from the display on the boat's echo-sounder, were viewed by a SeaViewer remote camera lens (SeaView Video Technology, Inc.), which was lowered from the boat to approximately 1 m above the seafloor. Substrate type was determined as sand or rock, by the view displayed on a small monitor on the boat, attached to the lens by cable. Locations appearing to have suitable rocky substrate were then assessed *in situ* by divers using SCUBA. Although the lowering of the camera lens and the use of SCUBA restricted the spatial extent that could be assessed at each locality, results from these dives confirmed those obtained by the SSS data and those from the vessel transects. The Rheeders Reef complex, immediately to the east of Storms River Mouth, was the selected study site in the protected area (Fig. 3.2).

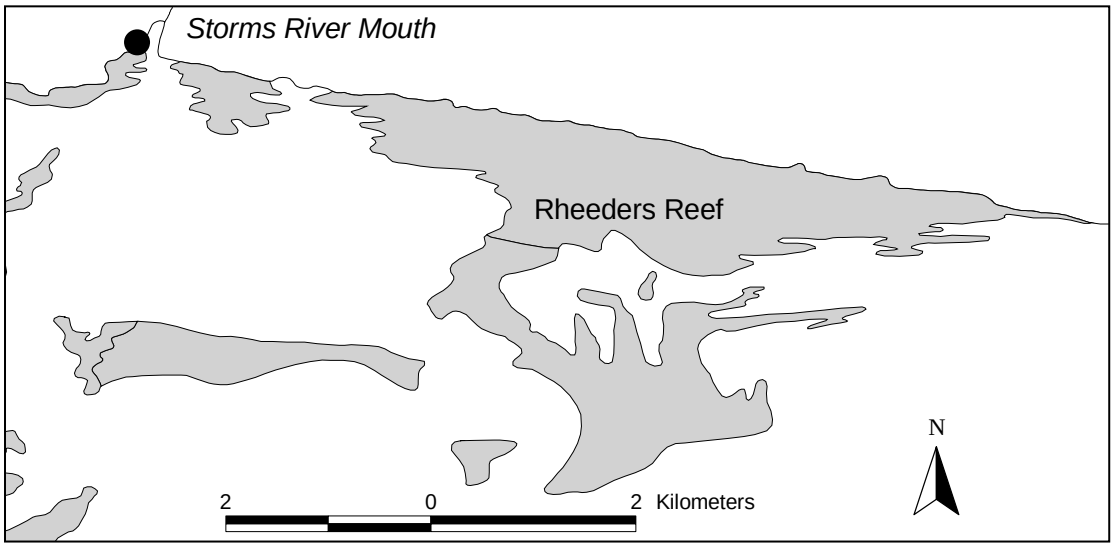

Figure 3.2: Map of the Rheeders Reef complex, showing reef areas (grey), and areas of sand and gravel (white). Data based on SSS survey data (Schumann *et al*. 1982, Flemming *et al*. 1983). The black dot shows the location of the launch site.

This site is situated near the centre of the TNP (i.e. far from the influences of edge effects), in close proximity to the conservation offices, and offshore of an inaccessible section of coastline characterised by steep cliffs, and has, therefore, probably not been subject to fishing activities over the past 40 years, with the exception of some research fishing (Buxton 1993a, Smith 2005).

Plettenberg Bay

For initial identification of a suitable study site within PB, low resolution bathymetry and physiography data from SSS (Schumann *et al*. 1982, Flemming *et al.* 1983, Martin and Flemming 1986) were once again scanned into electronic form and converted to shape files in ArcView. Additional depth data were provided by low resolution mapping from vessel transects (Smith 2005), and substrate was verified by remote camera lens and observational SCUBA dives.

A suitable area of contiguous rocky reef was identified to the west of Nature's Valley (Fig. 3.3). The eastern edge of the reef was approximately 35 km west of the study area within the TNP and separated from the western boundary of the TNP by approximately 3 km of open sand. The large area of nearshore reef of suitable depth for SCUBA observations, and similar depth and profile to the TNP, suggested that the area was suitable for implementation of the proposed sampling protocol.

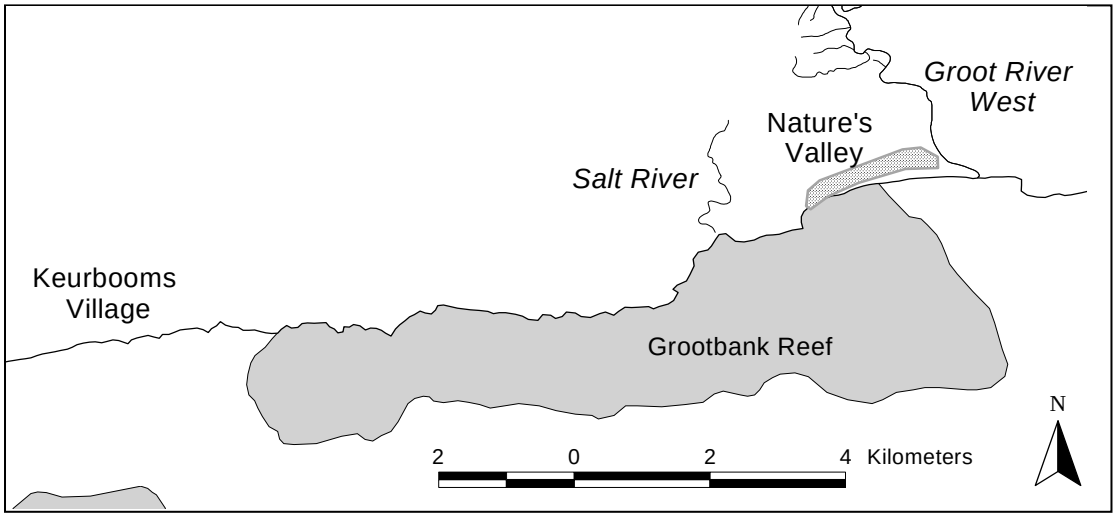

Figure 3.3: Map of the Plettenberg Bay study area showing areas of reef (grey) and surrounding sand and gravel (white). Adapted from Martin and Flemming (1986).

Fishing pressure in PB is comprised of commercial deck boat, recreational, commercial and charter skiboat fishing, and recreational shore-angling and spearfishing (Smith 2005). Effort is considerably higher for the commercial deck boats during winter. These boats target mainly shallow water hake (*Merluccius capensis*), but also silver kob (*Argyrosomus inodorus*) and geelbek (*Atractoscion aequidens*) when their numbers are high. Smith (2005) showed that between 2002 and 2004, the recreational, commercial and charter skiboats exerted an estimated total of 890 boat days•year^{-1}, or 3560 fisher days•year^{-1}, with seasonal peaks from December through January (summer holiday) and in April (Easter holiday). The main target species are silver kob, Garrick (*Lichia amia*), geelbek and hake, while catch composition is dominated by hake, carpenter (*Argyrozona argyrozona*), silver kob, roman (*Chrysoblephus laticeps*) and geelbek.

Chapter 4
Comparison of Underwater Visual Census and Controlled Fishing Methods

4.1 Introduction

In Chapter 2, UVC and controlled fishing were identified as the most suitable methods for monitoring fish populations and community structure change over time. The next step was to refine each of these methods in the light of the research sites identified. It was also necessary to identify a suitable indicator species for use in the study.

4.1.1 Identification of suitable methods

Within UVC and controlled fishing there are numerous techniques, each with their associated advantages and disadvantages, and suitability for different applications (Colvocoresses and Acosta 2007). Numerous criteria influence the choice of method and their suitability can only be determined by statistical comparison and evaluation.

Underwater visual census techniques

Numerous UVC techniques have been described for censusing reef fish populations (Thresher and Gunn 1986). However, most ecological studies involving UVC, particularly those conducting comparative assessments of reef fish communities inside and outside MPAs (Roberts and Polunin 1991), have made use of transect counts (Brock 1954, Brock 1982, Buxton 1993b, La Mesa and Vacchi 1999), point counts (Thresher and Gunn 1986, Miller and Ambrose 2000) or rapid visual census counts (Jones and Thompson 1978, Bortone *et al*. 1989) made by divers using SCUBA (Kulbicki 1998), with transect counts being the most commonly used (Thresher and Gunn 1986, Sale 1991) (Table 4.1).

Strip transects involve a diver traversing a predetermined distance and recording individuals within a specified path width (Thresher and Gunn 1986). Line transects (or distance sampling, Thresher and Gunn 1986) differ from strip transects, only in

that the diver records the estimated distance from a central transect line to each individual, as opposed to recording individuals within a specified strip width (Thresher and Gunn 1986). Belt transects differ only slightly from strip transects, in that two parallel transect lines are laid along the substrate, forming a passage, within which observed fish are recorded, but without the subjective estimation of strip width (Barrett and Buxton 2002). Use of the strip transect method excludes time consuming setting of transect lines or the consequent effects thereof on fish behaviour, and allows comparison with results obtained during previous UVC studies in the study area (Buxton and Smale 1989, Burger 1990).

Table 4.1: Selected studies making use of transect and point count UVC techniques.

Reference	Strip Transects	Line Transects	Instantaneous Area Point Counts	Interval Counts (Point)
Brock (1954)	x			
Brock (1982)	x			
Kimmel (1985)	x			
Bohnsack and Bannerot (1986)				x
Bortone *et al*. (1986)	x			
Clarke (1986)				x
Sanderson and Solonsky (1986)	x			
Thresher and Gunn (1986)	x	x	x	
Bortone *et al.* (1989)	x			x
Buckley and Heuckel (1989)	x			
Cole *et al.* (1990)	x			
Samoilys and Squire (1994)	x			
Jennings and Polunin (1995)			x	
Rakitin and Kramer (1996)	x			
Russ and Alcala (1996)	x			
Cheal and Thompson (1997)	x			
Kulbicki and Sarramegna (1999)	x	x		
La Mesa and Vacchi (1999)	x			
Cole *et al.* (2000)	x			
Willis *et al*. (2000)	x			
Zeller and Russ (2000)	x			
Pet-Soede *et al.* (2001)		x		
Barrett and Buxton (2002)	x			
Zeller *et al.* (2003)	x			
Baron *et al.* (2004)	x			x
Götz (2005)			x	
Smith (2005)			x	

Point counts can be divided into two types; instantaneous area point counts, in which the diver records all subjects within a specified radius in as short a time as possible (Bohnsack and Bannerot 1986) and interval counts in which the diver records all subjects passing through the census area in a specified time (Bortone *et al*. 1986). Density estimates made using the latter method are largely dependent on the swimming speed of the subjects, which may vary largely between sampling sites or seasons and among species. Results from this method are therefore likely to be highly variable (Thresher and Gunn 1986). The use of instantaneous area point counts minimises these problems and allows for comparison with results from other studies conducted in the area (Buxton and Smale 1989, Smith 2005).

The rapid visual census, and variations thereof, has also been used estimate abundance of reef fishes. However, this method involves the diver recording all subjects observed, while swimming along a random path for a predetermined duration (Bortone *et al*. 1989), providing no information on the area censused, making density estimates from such studies questionable.

From the above discussion, it appears that strip transects and instantaneous area point counts are most suitable for monitoring. These techniques are both commonly used and provide useful estimates of density, diversity and length frequency distributions of reef fishes (Watson and Quinn 1997). Point counts are particularly suitable for small or heterogenous habitats and artificial reefs, allowing comparison between large and small reefs (Bortone *et al*. 1989), and have been shown to provide higher density estimates and precision than transect counts (Thresher and Gunn 1986, Watson and Quinn 1997). Transects are suitable for assessing large expanses of contiguous reef (Sale 1991) and species with non-random distribution, which may be common among reef fishes (Kulbicki and Sarramegna 1999). The simplicity and "standardised protocol" of transects allow comparison between divers, sites and species, and over time (Bortone *et al*. 1989). Transects are preferred by numerous authors for quantitatively assessing fish assemblages (Brock 1982, DeMartini and Roberts 1982, Kimmel 1985, Sanderson and Solonsky 1986, Bortone *et al*. 1989).

The implementation of multiple UVC techniques is not feasible in LTM programmes, where financial and time constraints limit sample size, or in the high energy marine

environment off the South African south coast, where days at sea are limiting and low visibility conditions common. It is recommended that effort should rather be concentrated on a single technique.

Controlled fishing

Although numerous studies have made use of CPUE fishing, the duration and number of anglers used per fishing station in different studies have been inconsistent; for example 15 – 60 minutes with one angler (Zeller and Russ 1998), 30 minutes with two anglers (Smith 2005) and 60 – 75 minutes with three anglers (Haggarty and King 2006). Furthermore, numerous studies fail to make reference to the number of anglers used (Willis *et al*. 2000), providing no information on absolute effort. This lack of standardised optimal fishing station effort has limited comparability of results (Haggarty and King 2006), thus highlighting the need for a standardised fishing station effort to allow comparison between studies. Angling, to provide CPUE data, is commonly only conducted for a short duration; however, assessments aimed at determining the effort required to provide the lowest sampling associated variability would require considerably longer duration.

For a given number of fishing stations, greater effort would provide a greater absolute catch, and consequently more representative species composition and community structure. However, the maximum effort per fishing station is governed by financial, time and manpower constraints. Excessive effort would result in increased financial or manpower requirements, and/or a decrease in the number of stations that could be feasibly sampled per unit time (i.e. per day or field trip). For the same reason, there is a further trade-off between station effort and the minimum number of fishing stations required for statistical strength. Fishing stations should, therefore, be of sufficient effort to provide reasonable estimates of abundance and representation of size and species composition, but without excessive cost and required time, and at the same time provide minimum within site variability

4.1.2 Indicator species

Indicator species should be selected on the basis that their relationship between the indicator variable (e.g. mean length) and the population status is understood (Vos *et al*. 2000). Indicator species should be of public interest (i.e. keystone species), or of

commercial or economic value (e.g. targeted by a fishery) (Keough and Quinn 1991). Highly mobile or uncommon species are not suitable as indicators, as variability in estimates of such species is likely to be high (Green 1979). Indicator species should also be well-represented in the samples.

Chrysoblephus laticeps (roman) is a small to medium sized sparid, endemic to South Africa, with a maximum fork length of 512 mm, and weight of approximately 4.2 kg (Mann 2000). It is an important fishery species, targeted by spear-, recreational, charter-boat and commercial fishers. Due to its unique colouration, the species is easily identifiable even by the non-specialist. Roman has been targeted in the South African line fishery since about 1898 (Crawford and Crous 1982). Within the chosen exploited study area (PB) roman is targeted by all fishery sectors (Smith 2005).

Furthermore, roman has been the subject of much research, with focus on its biology (Buxton 1987, van der Elst 1993), life-history (Buxton 1987, 1989, 1993a), feeding (Buxton 1984), abundance (Buxton and Allen 1989, Buxton 1993b) and movement patterns (Buxton and Allen 1989, Griffiths and Wilke 2002, Kerwath *et al*. 2007), and the effects of fishing on its life-history parameters (Buxton and Smale 1989, Götz 2005). Roman was therefore deemed suitable as an indicator species for this study.

4.1.3 Measures of variability

In order to use variability to monitor population change, it must be accurately and precisely measured. Suitable measures of variability should, therefore, be independent of the mean population estimate and sample size, and utilise the data with the highest resolution (McArdle *et al*. 1990). Numerous measures are available for estimating variability, including the range (Rosner 2000), 95% confidence intervals, standard deviation (Rosner 2000), D_{max}/D_{min} (Stewart-Oaten *et al*. 1995), the standard deviation of the natural logarithms of successive population estimates ($SD[\ln(x_i)]$) (Connell and Sousa 1983), and the coefficient of variation (CV) (e.g. Haldane 1955).

Although such a wide range of measures is available for estimating variability, all have biases and many are unsuitable for comparisons of groups with different arithmetic means. Range is highly sensitive to outliers (Rosner 2000). Hurlbert (1984) suggested that 95% confidence intervals provided misleading representation of

variability, and that standard deviation (SD) provided a better estimate. However, due to the nature of its calculation, SD is dependent on the mean (Rosner 2000), and is therefore not suitable for comparing variability between groups with different means. D_{max}/D_{min} is calculated by dividing the largest population estimate by the smallest, and provides an estimate of variability in results, based on this ratio, but provides no information on the estimates between these values, and is badly biased by outliers.

The two most commonly used measures, SD[ln(x_i)] and CV, are independent of the mean and are therefore suitable for comparisons between groups having different mean values, such as those from transect and point counts, protected and exploited areas or fishing stations of varying effort (Stewart-Oaten *et al*. 1995, Lohr 1999).

SD[ln(x_i)]

SD[ln(x_i)] is the most widely used measure of variability (McArdle *et al*. 1990), and is calculated as the SD of the natural logarithms of successive population estimates (x_i). Although independent of the mean, estimates of variability using this measure are affected by spatial variability, and tend to overestimate true temporal population variability (Stewart-Oaten *et al*. 1995). Where counts or population estimates include zero values, SD[ln(x_i)] is undefined (natural log of zero = undefined).

Coefficient of variation (CV)

CV is unaffected by zero counts, and therefore does not require transformation of data. CV is slightly biased when sample size is low. However, Haldane (1955) provided a correction for this bias (CV'), which takes into account sample size.

CV is suitable for comparing samples with different arithmetic means as it accounts for the higher variability that is expected with a greater mean, and is therefore unaffected by the mean (Rosner 2000). For the same reason, CV was suitable for comparing variability associated with fishing stations of variable effort, in which mean catch numbers are expected to vary in relation to effort (Rosner 2000).

4.1.4 Aims

The aim of this chapter was to identify the optimal methods for inclusion in the design of a sampling protocol for LTM. This was achieved by:

1. comparing point and transect count UVC techniques, in terms of efficiency, variability, bias and required sample size, and
2. determining the optimal angler effort, in terms of efficiency, overall catch, overall CPUE, overall time required and variability.

4.2 Methods and materials

4.2.1 Study area

Data for the calculations were obtained from UVC counts and controlled fishing in the protected and exploited areas described in Chapter 3 (Figs 3.2 and 3.3). Depths at both sites ranged from 18 to 25 m, and substrates (determined during preliminary SCUBA dives and from the display on the boat's echo-sounder) were of similar profile, rugosity and habitat type. Data were obtained seasonally from winter 2005 to summer 2006/2007.

4.2.2 Allocation of sampling sites

Sampling fixed sites, as opposed to random sites within each area, reduces overall variability (Stewart-Oaten *et al*. 1995) by excluding spatial variability otherwise introduced by sampling a different set of sites on each occasion (Thompson and Mapstone 2002). Therefore, to exclude the effect of spatial variability, fishing stations were conducted at the same site within each study area (Ault and Johnson 1998).

4.2.3 General methods

Diving and controlled fishing were conducted from a 6 m ski-boat, anchored on a fixed locality in each study area. The sites were located using a Garmin GPS12 handheld GPS, with an estimated position error of 5 m. When anchoring, wind, current and swell directions were taken into account, so that the boat was positioned over the same spot on each occasion.

Underwater visual census

Diver training

Before entering the field, it was important that the observer was trained in fish length estimation, to minimise within-observer error (Samoilys 1997, Thompson and Mapstone 1997, Kulbicki 1998). Therefore, prior to the initiation of UVC sampling for this experiment, the author (the sole observer for this experiment), underwent diver training. This involved underwater estimation of the lengths of model fish, cut from high density polyethylene, ranging in length from 2 to 65 cm (TL). From an opaque bag, a second diver (positioned three to five metres from the observer) produced a single model fish of random length, which was held stationary for 2 to 3 seconds at a position to either side or above the second diver, or moved through the water in a "mock swimming" motion, during which time the observer would estimate its length, to the nearest cm (TL). Once the lengths of 50 model fish had been estimated, these lengths were compared to actual lengths using a paired t-test (Samoilys 1997). This process was repeated until there was no significant difference between estimated and actual lengths of the model fish.

Strip transect counts

Once at the dive site, the anchor was deployed. Two divers descended on SCUBA, following the anchor rope to the substrate. Once on the bottom, the divers swam a distance of 5 m from the anchor in a random direction, before beginning the count. This five-metre section acted as a buffer to avoid effects of diver presence and anchor deployment on fish behaviour. From this point diver one swam in a straight line at a swimming speed of approximately 8 $m \bullet min^{-1}$ (Cheal and Thompson 1997), recording all individuals observed in a strip of six to ten metres wide (according to visibility). Lengths of all roman (the indicator species) observed were estimated and assigned to length classes of 5 cm increments (Mann *et al*. 2006), and recorded on Perspex slates. La Mesa and Vacchi (1999) suggested a swimming speed of $5m \bullet min^{-1}$; however, bottom time was restricted by depth and De Girolamo and Mazzoldi (2001) suggested that an increased swimming speed improved results obtained for epibenthic species, such as those investigated during the current study. Fishes that passed the diver from behind were not recorded, to avoid the possibility of counting the same individuals more than once (Froeschke *et al*. 2006).

The strip width was limited to a minimum of three and a maximum of five metres on either side of the diver depending on the visibility (Mann *et al*. 2006), which is commonly between 3 and 5 m in the two study areas (Smith 2005). Dives in which visibility was less than 3 m were aborted (Brock 1954, Ebeling and Hixon 1991). Diver two followed diver one, releasing a graduated line from a dive reel, and alerted diver one once a distance of 50 m had been traversed (Mapstone and Ayling 1993, La Mesa and Vacchi 1999, Zeller and Russ 2000). Visibility was estimated at the end of each transect by recording the maximum distance of observable substrate along the graduated line. The area censused by each replicate was calculated using twice the visibility as strip width, multiplied by the transect length (50 m). The duration of each transect was approximately six minutes.

Point counts

For the purpose of comparison, the point count method followed that described by Smith (2005). Point counts involved two observers descending slowly together. Diver two remained on the substrate at the shot line, while diver one swam to a point 10 m from diver two. This point acted as the centre of the point count census area. A distance of 10 m allowed a 5 m buffer and a maximum point count radius of 5 m. Diver one rapidly scanned an area of 3 to 5 m radius (depending on visibility) and recorded numbers of all species observed on a Perspex dive slate in as short a time as possible (Barrett and Buxton 2002). Once again, lengths of all roman observed were estimated and assigned to length classes of 5 cm increments. Diver one then returned along the line and again swam 10 m from diver two but in the opposite direction, where the counting process was repeated. Diver one repeated this until four areas (constituting four replicates) had been censused, at 180º, 90º and 270º from the direction of the initial swim. Visibility was recorded for each replicate, following the same procedures as described for the transect counts. The area censused on each replicate was calculated using the formula for the area of circle, using the estimated visibility as the radius. Each point count replicate took approximately four minutes.

Limiting data to those dives in which transect and point counts were both conducted allowed direct comparison of the two techniques. Four transect and four point count replicates were conducted on each dive. For each census technique, all replicates

conducted per dive were pooled, so that each dive constituted a single sample. All counts were made by a single observer to avoid between-observer error.

Controlled fishing

Within each area, the vessel was anchored on the same GPS co-ordinates. Angling was conducted for up to five hours (10 angler hours) per station at each site, to allow comparison of the variability obtained for varying fishing effort. Two anglers, fishing simultaneously, used a standardised hook-and-line configuration to avoid bias introduced by the use of different tackle. This included a single barbless 4/0 VMC sport circle hook, baited with pilchard (*Sardinops sagax*) and chokka squid (*Loligo vulgaris reynaudii*) and a 170 g sinker on each line. Circle hooks were used to decrease the incidence of gut-hooking (Zimmerman and Bochonek 2002, Cooke and Suski 2004) and post-release mortality (Falterman and Graves 2002, Prince *et al*. 2007), and barbs were removed to avoid unnecessary injury (Parsons *et al*. 2003) and facilitate hook-removal (Schaeffer and Hoffman 2002). Once at the surface, fish were brought onboard the vessel in a PVC fish sling, equipped with a measuring tape. Swim bladders of fish exhibiting signs of barotrauma were deflated by careful insertion of a 15-gauge hypodermic needle under a scale, through the body wall at a position where the swim bladder adheres to the abdominal wall (Buxton 1990, Bruesewitz *et al*. 1993, Keniry *et al*. 1996, Collins *et al*. 1999). The hook was removed and the fish was measured to the nearest millimetre, fork length (FL) and total length (TL), before being released.

It is suggested that the severity of the injury or stress inflicted on a captured fish is a function of the duration that the fish spends at low pressure, i.e. in shallow water or on the surface (Smith 2005). Therefore all fish were processed (deflated, hook removed and measured) and returned to the water in as little time and with as little handling stress as possible, to maximise the chances of survival. Processing time for all fish was kept below 30 seconds. To avoid between-angler variability, all fishing for this experiment was conducted by the same two anglers. All diving and fishing was restricted to daylight hours (08:00 to 17:00) to minimise effects of crepuscular activity.

4.2.4 Data analysis

Underwater visual census

Density and species density were calculated as the number of individuals and species, respectively, observed per 100 m^2. Data were then adjusted for effort, by calculating the mean number of individuals and species recorded per replicate by each technique, by converting each count to standardised point count and transect census areas. These census areas were calculated using a point count census area radius and half transect strip width of 3 m (based on 3 m visibility). This provided a point count census area of 28 m^2 and a transect area of 300 m^2. Data were also adjusted for time by calculating the mean number of individuals recorded by each census technique per hour, based on average census durations of six minutes per transect and four minutes per point count. All comparisons were made using the non-parametric Wilcoxon Matched Pairs (Rank Sum) Test, as the assumptions of normality of distribution (Chi-square test) and homogeneity of variance were not met (Hartley F-max and Levene's test). All analyses were run in Statistica 7.1 (StatSoft, Inc.).

Variability was measured using CV and SD[ln(x_i)] of density estimates (fish•100 m^{-2}), and compared between transect and point counts, to determine which UVC technique provided the lowest sampling-associated variability. Variability in estimates of species density (species•100 m^{-2}), number of fish per replicate, number of species per replicate and number of fish per hour of census were also compared.

Sample size required was calculated as the sample size required to detect a 10% change in the mean estimate, at a significance criterion of 0.05 (Bausell and Li 2002), with a power of 80% (Fairweather 1991, Peterman and M'Gonigle 1992, Rosner 2000, Lenth 2001).Required sample size was determined by power analysis, according to the equation provided by Kapadia *et al*. (2005) for a one sample, two-tailed test, which takes the form:

$$n = \left[\frac{\sigma\left(z_{1-\alpha/2} + z_{1-\beta}\right)}{\mu_a - \mu_0} \right]^2,$$

where n is the number of samples required, σ is the standard deviation of the estimates, $z_{1-\alpha/2}$ is the Z value corresponding to the significance criterion (α) of 0.05

for a two-tailed test (obtained from statistical tables, Rosner 2000) with n - 1 degrees of freedom, $z_{1-\beta}$ is the Z value that corresponds to a power of 0.8 (obtained from statistical tables, Rosner 2000), and $\mu_a - \mu_0$ is the difference between the means required to be detected (Kapadia *et al*. 2005).

Controlled fishing

To determine fishing station effort that provided the lowest variability in CPUE, fishing results from each station were divided into periods of one angler hour (i.e. two anglers fishing simultaneously for 30 minutes). Catch from each consecutive angler hour within each day was added cumulatively to provide estimates of abundance for stations of effort ranging from one to 10 angler hours. CPUE was calculated by dividing the overall catch for cumulative angler hours at each station, by the number of angler hours exerted, to provide an index of abundance based on fish•angler hour^{-1}.

Measures of variability

Variability between counts made by transects and point counts, in which counts were based on census areas of different sizes, was compared using SD[ln(x_i)] and CV. Based on the previous discussion, CV and SD[ln(x_i)] were also chosen to measure catch variability for comparison of fishing station effort. However, as CV is slightly biased, CV adjusted (CV', Haldane 1955) was also used.

SD[ln(x_i)] is calculated as the standard deviation (SD) of the natural logarithms of successive population estimates (x_i) (Connell and Sousa 1983). CV is calculated by dividing the SD of a set of population estimates by the mean, and takes the form:

$$CV = [SD(x_i)]/\bar{x},$$

where SD = standard deviation of successive population estimates, x_i = population estimate or sample number i, and $\bar{x}$ = mean population estimate (McArdle *et al*. 1990). CV' is calculated in the same way as CV, but takes into account sample size, and takes the form:

$$CV' = \left(1 + \frac{1}{4n}\right) \times \left(\frac{SD(x_i)}{\bar{x}}\right),$$

where CV' = CV adjusted for bias according to Haldane (1955), SD = standard deviation, $\bar{x}$ = mean population estimate, and n = sample size (Haldane 1955).

Comparison of length-frequency distributions

Length-frequency distributions of roman were compared between UVC and controlled fishing. Lengths for each size class estimated during point and transect counts were pooled, and those measured during controlled fishing were assigned to the same length classes of five cm increments.

4.3 Results

4.3.1 Diver training

During diver training, the second set of 50 estimated lengths correlated well with actual lengths ($r^2 = 0.987$, $p < 0.01$) (Fig. 4.1).

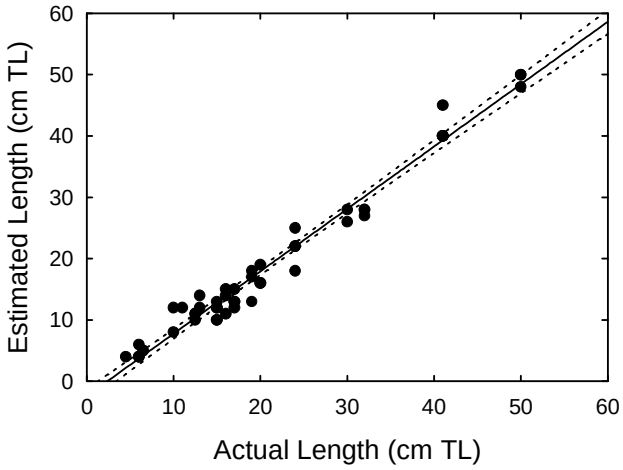

Figure 4.1: Estimated versus actual lengths (cm TL) of model fish for diver training.

4.3.2 Underwater visual census

For this experiment, 44 dives were conducted; 33 in the protected area and 11 in the exploited area. Both transect counts and point counts were conducted during 22 of the dives in the protected area and nine in the exploited area, allowing direct comparison of variabilities. A total of 3729 fishes representing 20 species were recorded during the point counts, and 7913 fishes representing 24 species during transect counts. Species observed using each technique are presented in Appendix I.

Efficiency

Results of the Wilcoxon Matched Pairs Test (Table 4.2) showed that densities (fish•100 m^{-2}) recorded during point counts were significantly higher than those recorded during transect counts, in the protected ($p<0.001$) and exploited ($p<0.01$) areas. Similarly, species densities (species•100 m^{-2}) were significantly higher for point counts than transect counts, for the protected ($p<0.001$) and exploited ($p<0.01$) areas.

Numbers of individuals recorded per replicate (all replicates per dive pooled), adjusted for point count radius and half transect strip width of 3 m, were significantly lower for point counts than transect counts, in the protected ($p<0.001$) and exploited ($p<0.01$) areas. Similarly, numbers of species recorded per replicate were significantly lower for point counts than transect counts, in the protected ($p<0.001$) and exploited areas ($p<0.01$). When adjusted for census duration, the numbers of fish recorded per hour of transect census were significantly higher than those recorded per hour of point count census, for the protected ($p<0.001$) and exploited sites ($p<0.01$).

Table 4.2: Results of comparisons of density (fish•100 m^{-2}), species density (species•100 m^{-2}), fish per replicate, species per replicate and fish per hour (±SD), from transect and point counts

	Protected					Exploited				
	Transect		Point		p	Transect		Point		p
Density	11.41	(2.59)	35.07	(16.87)	0.001	10.13	(3.15)	36.13	(13.11)	0.01
Species density	1.43	(0.24)	7.16	(2.27)	0.001	2.06	(0.43)	11.21	(5.24)	0.01
Fish/replicate	34.24	(7.76)	9.92	(4.77)	0.001	30.39	(9.44)	10.21	(3.71)	0.01
Species/replicate	4.28	(0.73)	2.02	(0.64)	0.001	6.18	(1.28)	3.17	(1.48)	0.01
Fish/hour	342.4	(77.64)	148.75	(71.54)	0.001	303.9	(94.42)	153.2	(55.58)	0.01

Variability

For density, species density, fish per replicate, species per replicate and fish per hour, variability (in terms of CV and SD[$\ln(x_i)$]), was higher for point counts than for transect counts conducted on the same dives, for both the protected and exploited areas (Figs 4.2 and 4.3).

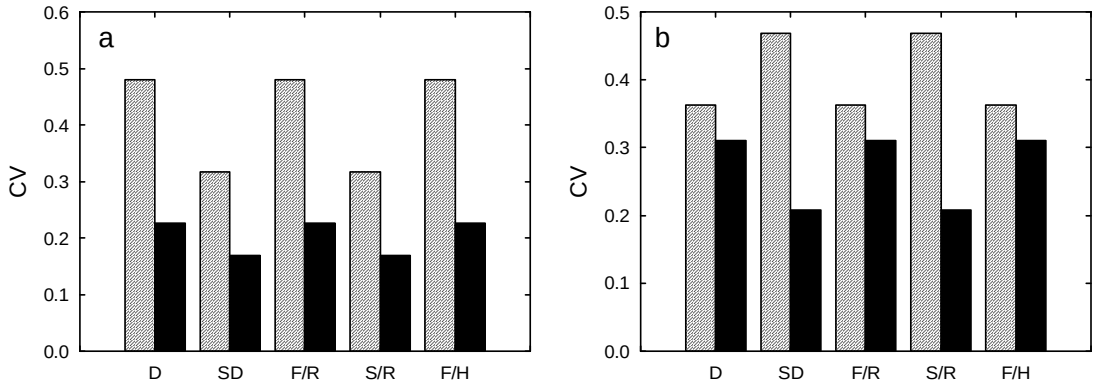

Figure 4.2: Variability (CV) for density (D; fish•100 m^{-2}), species density (SD; species•100 m^{-2}), fish per replicate (F/R), species per replicate (S/R) and fish per hour (F/H), in the TNP (a) and PB (b). Grey bars represent point counts and black bars transect counts.

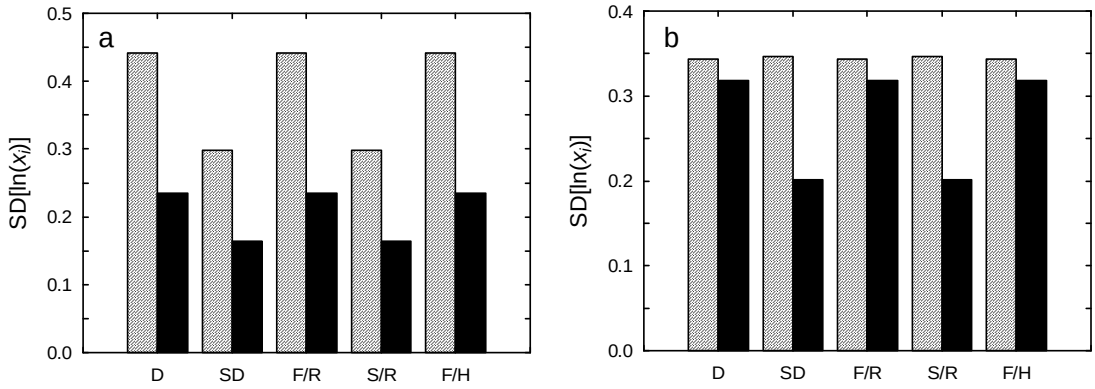

Figure 4.3: Variability (SD[ln(x_i)]) for density (D; fish•100 m^{-2}), species density (SD; species•100 m^{-2}), fish per replicate (F/R), species per replicate (S/R) and fish per hour (F/H), in the TNP (a) and PB (b). Grey bars represent point counts and black bars transect counts.

Point counts required less time per replicate than strip transects, but covered a smaller census area under all visibility conditions encountered during this experiment (Fig. 4.4). Therefore, strip transects probably provide more representative estimates of community structure. Furthermore, when adjusted for time required per replicate, transects covered a greater area per unit time than point counts, allowing a greater overall area to be sampled in a given time.

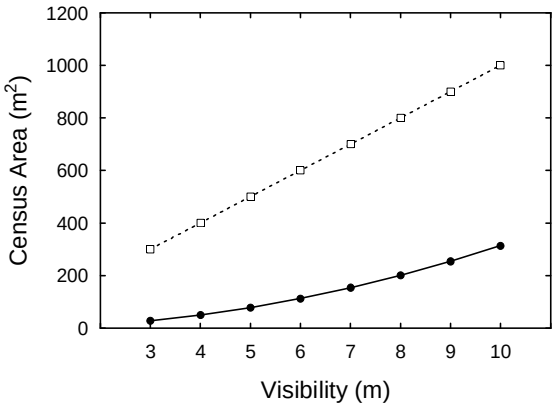

Figure 4.4: Area (m^2) censused by strip transects (dashed line) and point counts (solid line) under varying visibility (m).

Results of the power analysis showed that the sample size required to detect a 10% change in the mean UVC count, with a statistical power of 80% (or 0.8), was 28 for point counts, and 11 for transects, at the 5% level of significance.

4.3.3 Controlled fishing

A total of 583 fishes and 18 species were captured in 208 angler hours in the protected area (2.80 fish•angler $hour^{-1}$), while 201 fishes and 11 species were captured in 104 angler hours in the exploited area (1.93 fish•angler $hour^{-1}$). For the two areas combined, a total of 21 species were captured, with eight species common to both areas. Species that were captured are listed in Appendix II.

Sampling efficiency

As expected, mean catch (per station) increased with station effort, for roman and for all species combined, in the protected and exploited areas (Fig. 4.5). The magnitude of the increase was not linear and decreased as station effort increased, particularly for roman. Roman catch approached asymptotic at a lower station effort in the exploited area than the protected area. Conversely, the mean cumulative CPUE (fish•angler hour) decreased with increased effort, for roman and for all species combined, in both areas (Fig. 4.6). The rate of decrease in CPUE with increased effort was rapid for roman in both areas, but was more gradual for all species combined. Despite an increase from one to two angler hours, CPUE of all species in the exploited area showed a gradual overall decrease with increased effort.

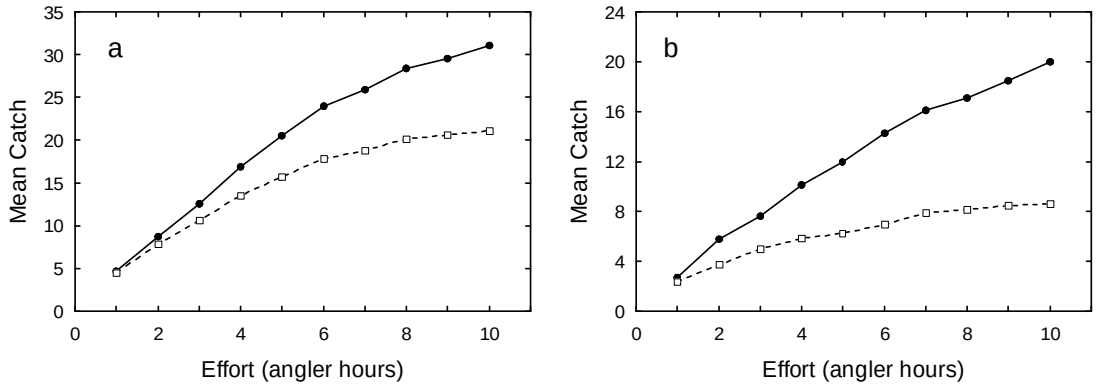

Figure 4.5: Mean catch (fish per station) versus fishing station effort (angler hours), for the protected (a) and exploited (b) areas. The solid line represents all species combined and the dashed line represents roman catch only. Standard deviations were omitted to aid clarity of presentation. (n = 8 for each effort).

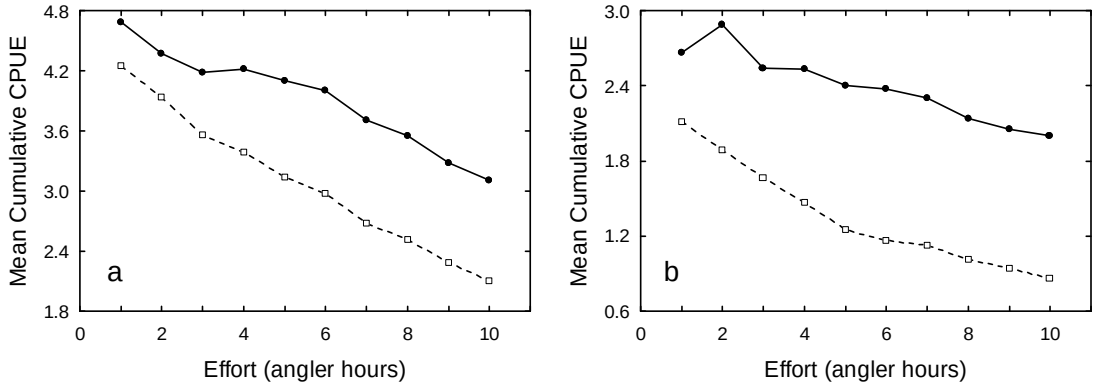

Figure 4.6: Mean cumulative CPUE (fish•angler hour^{-1}) versus fishing station effort (angler hours), for the protected (a) and exploited areas (b). The solid line represents CPUE of all species combined and the dashed line represents roman CPUE. Standard deviations were omitted to aid clarity of presentation. (n = 8 for each effort).

Mean contribution of each consecutive angler hour to overall roman catch at each station is presented in Figure 4.7. In the protected area (TNP), the catch decreased rapidly from a maximum after one angler hour. In the exploited area (PB), there was a gradual decrease in mean contribution per angler hour until three angler hours, after which there was a sharp decline to five angler hours, with approximately 60% of the roman catch for each station being captured within the first three angler hours, and the remaining approximately 40% being captured in the subsequent seven angler hours.

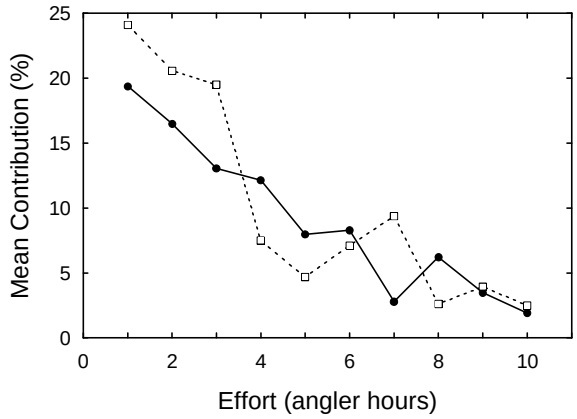

Figure 4.7: Mean roman contribution (% of overall roman catch per station) for each consecutive angler hour, in the TNP (solid line) and PB (dashed line). Standard deviations were omitted to aid clarity of presentation. (n = 8 for each effort).

Variability

Variability associated with fishing stations of varying effort are displayed in Figure 4.8. Variability (CV and SD[ln(x_i)]) for the TNP, as well as the two areas combined, decreased gradually with increased effort (Fig. 4.8). Overall, variability in PB also decreased with increased effort, but showed a sharp decrease from one to two angler hours, after which the decrease was gradual.

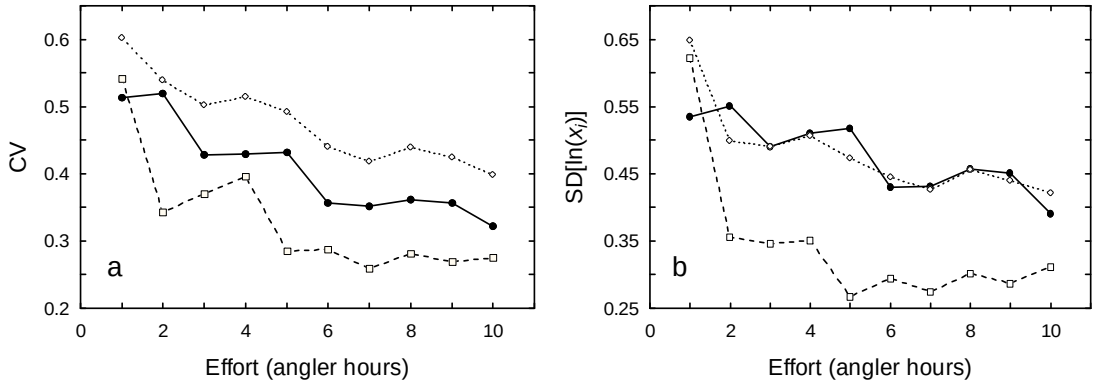

Figure 4.8: Variability associated with fishing stations of varying effort, measured using CV (a) and SD[ln(x_i)] (b). Data are for summer and winter combined, for the TNP (solid line), PB (dashed line) and the TNP and PB combined (dotted line).

Variability estimated by CV and CV' provided near identical results. The results of CV' were therefore excluded. Variability associated with varying fishing effort, measured in terms of CV and SD[ln(x_i)], was compared between summer and winter

for the TNP and PB (Fig. 4.9). In the TNP, variability (CV and SD[ln(x_i)]) in summer decreased gradually with increased effort, and was considerably higher than in winter. Variability (CV and SD[ln(x_i)]) decreased sharply in winter in the TNP, from one to four angler hours, after which it showed little change. In PB, variability (CV and SD[ln(x_i)]) was higher for the first two angler hours in winter, after which values decreased to below summer values. Summer variability (both measures) was lowest after two angler hours, after which it increased gradually with effort. Winter variability in PB decreased sharply with increased effort, to minima after seven hours.

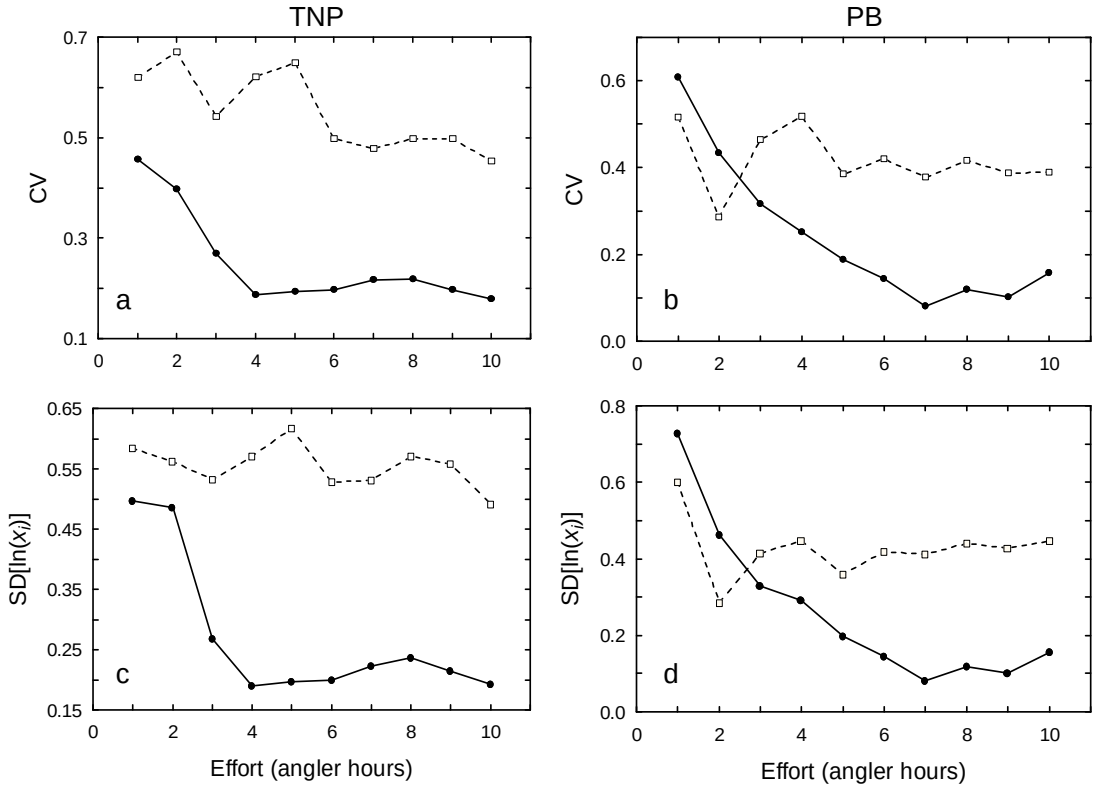

Figure 4.9: Comparison of seasonal variability associated with varying fishing effort (angler hours), measured as CV in the TNP (a) and PB (b), and as SD[ln(x_i)] in the TNP (c) and PB (d). The solid line represents winter and the dashed line summer.

Comparison of length-frequency distributions

Length-frequency distributions of roman obtained during UVC (point and transect data pooled) were evenly distributed across length classes. However, larger length classes were disproportionately represented by the fishing data, with length classes up to 15 cm not represented at all (Fig. 4.10).

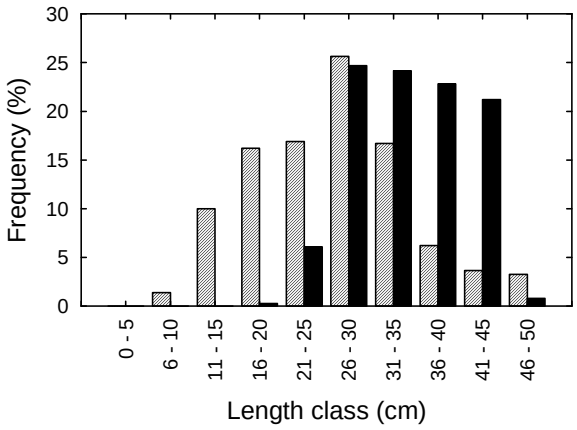

Figure 4.10: Length-frequency distributions for roman, estimated during UVC (hatched bars, n = 1011) and measured during controlled fishing (solid bars, n = 377).

4.4 Discussion

4.4.1 Underwater visual census

Density estimates were significantly higher for point counts than transect counts. This agrees with results obtained by Bortone *et al*. (1989). Buxton (1987) found that transects underestimated density. The result is likely due to the effect of diver presence on fish behaviour during point counts, in which the diver remains stationary, allowing curious species to approach (Kulbicki 1998). Thresher and Gunn (1986) and Samoilys and Carlos (1995) found no significant differences in density estimates between point counts and strip transects. However, their point count census area size was not limited by visibility, which was greater than 15 m throughout their studies. The effect of diver presence is likely to decrease with distance from the diver, therefore, being greater when the point count radius is smaller.

Transects covered a larger census area per replicate than point counts, and the numbers of individuals and species recorded per replicate were significantly higher for transect counts than point counts. Transects are, therefore, likely to provide more representative estimates of community structure. Furthermore, when adjusted for census duration, the numbers of fish recorded per unit of time were significantly higher for transect counts. This is consistent with results obtained by Bortone *et al*. (1989). Transects during this study took approximately six minutes per replicate, and point counts approximately four minutes. Although point counts covered a smaller

area, the disproportionate time required per point count replicate may be due to difficulties associated with recording an increasing number of individuals as fishes are constantly attracted to the stationary diver, as well as the time required to distinguish between fishes already present and those entering the census area after the census has started. Furthermore, the diver must rotate to detect all individuals in the census area, and if done for each species, may be time consuming, particularly in areas where fish density and species richness are high.

Transect counts provided lower variability in estimates of density than point counts (fish•100 m^{-2}). This is contrary to results obtained by Watson and Quinn (1997) and Thresher and Gunn (1986), who found that point counts provided lower variability than transects. This discrepancy in results may be due to a smaller census area during the current study. Miller and Ambrose (2000) found that transects provided more accurate estimates of density in the intertidal zone than quadrats, as quadrats were more likely to fall entirely within or between patches. This is analogous to point counts, which are likely to provide lower within site variability, due to more homogenous habitat within each census area, but less accurate estimates overall than transects, which are more likely to cross different habitat types. Variability was also lower for species density (species•100 m^{-2}), number of fish per replicate, number of species per replicate and fish per hour from transects than those from point counts.

Bias may be caused by the effect of diver presence during point counts. Furthermore, fishes entering or leaving the census area during point counts may not always be detected, as some part of the census area is always behind the observer. This is not the case with transect counts. Bias may be introduced in transect counts by the non-random movement of fishes, i.e. net movement towards or away from the observer (Watson *et al*. 1995). Potential inaccuracy is introduced in transect and point counts, by subjective estimation of strip width and point count radius, respectively. However, the problem is exaggerated in point counts, as the distance estimated (i.e. point count radius), and consequently any error in distance estimation, is squared in the calculation of the area of a circle (Greenwood 1996). This is not the case with the calculation of the area of a rectangle, which can be shown by a simple example. An error in strip width estimation of 20% (e.g. a transect width of 5 m censused on either side of the diver, erroneously estimated to be 4 m), will result in a 20% decrease in

the area thought to be censused by the transect count, which relates to a 25% increase in that density estimate. However, a similar error in estimation of point count radius results in a 36% decrease in the area censused by the point count, and a consequent 56% inflation of the density estimate. Areas estimated during point counts are, therefore, likely to be less accurate, suggesting that density estimates provided by point counts may incur more bias than those from transects.

Power analysis showed that a sample size of 28 point counts was required to detect a 10% change in the mean count, while just 11 transect samples would be required. This further highlights the greater efficiency of transect counts over point counts.

Although point counts provided higher density estimates, which are suggested to be more accurate (Colvocoresses and Acosta 2007), strip transects provided lower variability (Figs 4.2 and 4.3), while at the same time a higher number of individuals and species per replicate. Because the study focused rather on relative density and representivity than absolute abundance, transects were more suitable and, therefore, are recommended for LTM.

4.4.2 Controlled fishing

LTM programmes commonly aim to detect inter-annual change, in which case data collected in summer and winter may be pooled. Therefore, optimal fishing station effort was based on data from the two seasons combined. The decreases in variability (CV and SD[ln(x_i)]) with increased effort, for the TNP and areas combined, suggested that a greater effort would be most suitable (Fig. 4.9). However, the magnitude of the decreases in most cases was low, suggesting that the decreased variability may not be worth the increased time and cost. Furthermore, CPUE decreased with increased effort (Fig. 4.6). Therefore, a reduced effort per station would provide a greater overall CPUE, but a reduced overall catch (Fig. 4.5). In order to maximise overall catch and CPUE for a given overall effort, a larger number of fishing stations of reduced effort would be necessary, and would provide a greater overall catch and thus a more representative sample of the community. In addition, the contribution of roman catch for consecutive angler hours in the TNP and PB decreased rapidly with increased effort (Fig. 4.7). In PB, approximately 60% of the total roman catch for each station was captured within the first three angler hours, suggesting that

controlled fishing for roman (as an indicator species) would be most efficient if based on stations of three angler hours or less. From a logistical point of view, reduced fishing station effort would allow for a more flexible sampling protocol. For example, fishing stations could be conducted between dives on diving days, without limiting the number of dives possible on a given day. Lower effort, for a given sample size, would also decrease the overall time required to complete all stations, possibly decreasing temporal variability within the results.

Variability in PB decreased sharply with increased effort from one to two angler hours, after which further decreases associated with increased effort were only slight. CV decreased by 37% from one angler hour to two, but required a further five angler hours for a further 15% decrease, to reach a minimum. Similarly, SD[ln(x_i)] decreased by 43% from one to two angler hours, but required an additional three angler hours to reach its minimum, a further decrease of only 14% (Fig. 4.9). This suggested that an increase in effort from one to two angler hours, to provide considerably lower variability, would be worth the decrease in CPUE and increased time and cost required, but that further decreases in variability were insufficient to merit further increase in angler effort. Two angler hours was therefore deemed the most optimal effort.

The seasonal differences in variability, found in both the TNP and PB, suggested that a seasonal sampling protocol was necessary.

The protocol adopted for this experiment made use of a single site in each study area, and is thus subject to pseudoreplication. However, this approach was selected to address specific aims. Sampling the same site repeatedly, within each study area, as opposed to sampling different sites on each occasion, excluded spatial variability and provided direct comparisons of transect and point counts, and fishing stations of varying effort, in which variability was attributable almost exclusively to each technique. A larger number of sites (i.e. repetitive sampling at numerous sites) would have increased statistical power. However, the nature of the experiment and the absolute duration (in hours) of fishing at each site per day, the overall dive time required for all sampling at the two sites and the limited number of available seagoing days in the temperate marine environment in which the study was conducted, limited

the number of sites to just two. Ultimately, the results provide a complement to the theoretical comparisons of the two UVC techniques, allowing identification of the more suitable technique, for use in this study area. Furthermore, in the absence of available literature on the optimal effort to be exerted at each fishing station, the results have provided an indication of a suitable angler effort. Further work is, however, necessary with an increased number of sampling sites.

Calculations for optimal fishing station effort were based solely on CPUE estimates at each site; however, it is recommended that further analyses be done to determine fishing station effort that provides suitably low variability in estimates of diversity, and representation of community structure.

4.4.3 Comparison of length-frequency distributions

As a result of hook-size selectivity (Perrow *et al*. 1996, DeMartini and Lau 1999), smaller size classes were under-represented by controlled fishing when compared with UVC. UVC provided a length-frequency distribution representative of length classes across the range of roman sizes and is, therefore, suitable for assessments of population size structure. Controlled fishing did not provide a representative illustration of the size structure of the population, and is, therefore, not suitable as a health index for a species or ecosystem. However, the length-frequency distribution provided by controlled fishing is representative of the population available to the fishery (Zeller and Russ 2000) and is suitable for assessments of the effects of fishing on population size structure.

4.4.4 Conclusions

Although there are numerous methods available, the most commonly used methods to provide density or abundance estimates of temperate reef fish populations are UVC (Brock 1954, Thresher and Gunn 1986, Willis *et al*. 2000) and controlled CPUE fishing (Bannerot and Austin 1983, Richards and Schnute 1986, Millar and Willis 1999, Bennett and Attwood 1993). Haggarty and King (2006) found that estimates from UVC compared well with those from CPUE fishing. Although the two methods provided inconsistent length-frequency distributions, both estimates were suitable for their respective applications. Therefore, based on the strengths of UVC (direct observation, non-selectivity and greater species representivity), and controlled CPUE

fishing (measured lengths, unconstrained by depth and bottom time, low cost), it is recommended that both methods should be included in a LTM sampling protocol.

Transect counts were superior to point counts, in terms of sampling efficiency, variability, bias and required sample size. In terms of controlled fishing, the optimal fishing station effort (for offshore angling in exploited and protected areas) was two angler hours, providing low variability while at the same time low overall cost and time required for sampling.

Chapter 5
Development of a Sampling Protocol

5.1 Introduction

The overall aim of this thesis was to develop a sampling protocol that is suitable for LTM and assessment of temperate reef fish communities. The sampling protocol, therefore, had to be designed to:

1. detect natural temporal change in population abundance and community structure, to allow detection of change in protected areas that may be associated with climate change or anthropogenic influences,
2. detect change in the exploited area that may be associated with fishing pressure, or changes in fishing pressure or management regime, and
3. detect differences between protected and exploited areas.

The first objective required a protocol that could assess parameters at the community level, such as species richness, species composition, diversity, evenness, species dominance and community similarities and dissimilarities.

The second and third objectives required sampling that allowed for statistical comparison of species-level indicators, such as mean densities, mean CPUE, mean lengths and length-frequency distributions. The use of abundance indicators to investigate the effects of fishing is common (*inter alia* Bennett and Griffiths 1986, Burger 1990, Bennett and Attwood 1993, Buxton 1993a, Rakitin and Kramer 1996, Willis *et al*. 2000, Brouwer and Buxton 2002). However, due to the multispecies nature of reef fish communities and the fact that different groups of species may be sampled best with different methods (Lincoln-Smith 1989), it was not possible to compare all the above-mentioned parameters for every species. One or more suitable indicator species therefore had to be identified.

All three objectives required the use of methods that provided the minimum possible sampling-associated variability and bias. These were identified in Chapter 4. In

addition, the problems associated with statistical power, variability, error and pseudoreplication (Cohen 1973, Hurlbert 1984, Stewart-Oaten *et al*. 1986, Forbes 1990, Peterman 1990, Fairweather 1991, Faith *et al*. 1991) had to be considered, in order to provide a sampling protocol that is statistically powerful, free of sampling error and pseudoreplication, and minimises variability and bias.

The aim of this chapter was, therefore, to determine suitable parameters, such as sample size, census area size, spatial and temporal extent of sampling, and spatial allocation of sample units, to develop a standardised sampling protocol that minimised the above-mentioned issues, but also considered available time, manpower and financial resources. In order to achieve this, the design of the sampling protocol had to consider each component separately, on a step-by-step basis (Fig. 5.1).

The first step was to identify a suitable study area, after which the species- and community-level indicators to be measured had to be determined. Once these had been selected, the sampling approach, sampling method and scale components could be considered. Once the components within these three sections had been determined, suitable methods were identified for data analysis.

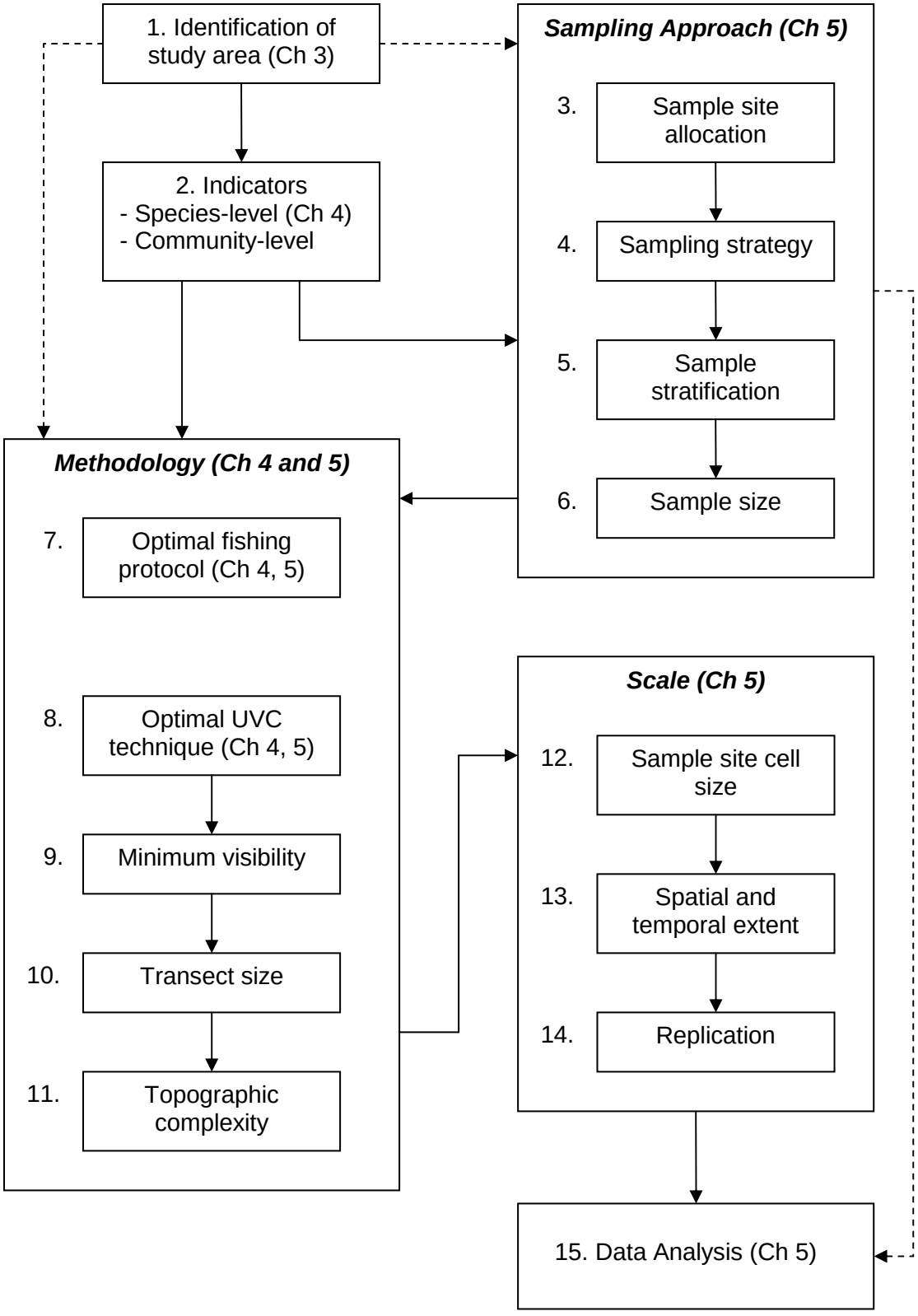

Figure 5.1: Steps in the development of the proposed sampling protocol. Solid arrows illustrate the direct order in which components were considered, while the dashed arrows represent the order in which the different sections of protocol design were considered.

5.2 Development of a protocol

5.2.1 Identification of suitable study areas

Before the sampling protocol could be designed, a suitable monitoring site had to be identified. This was achieved in Chapter 3.

5.2.2 Selection of indicators

Ecosystem based management (EBM) is increasingly using ecological indicators to aid management, as they can be used to assess the current status or condition of a stock or ecosystem, simplify complex biological information to allow ease of communication and detection of trends, and monitor progress towards management and ecological goals (Pajak 2000). However, the choice of which indicator variables or species to measure is not a simple one (Keough and Quinn 1991, Degnbol and Jarre 2004).

A suitable indicator is one that is easier, cheaper or more accurate to measure, or that shows an earlier response to an impact (e.g. a protected area becoming open to exploitation) or changes in environmental variables (e.g. increase in sea temperatures) (Vos *et al*. 2000). Indicators should also convey as much information as possible with regards to the effects of anthropogenic impacts and environmental change on the health of the reef community (Green 1979, Hodgson 1999). Pajak (2000) provided a good synthesis of possible indicators.

Species-level indicators

Chrysoblephus laticeps (roman) was selected as an indicator species in this study, as it is targeted by commercial, recreational and charter-boat fisheries, it is well-represented in UVC counts and catches in the area, and the species has been well studied (Chapter 4). The indicator variables selected to study roman included mean density from UVC, mean CPUE, mean length and length-frequency distribution in the protected (TNP) and exploited (PB) study areas.

Community-level indicators

Community-level indicators used to detect temporal changes associated with climate change should be able to detect changes in community structure and species composition. Species richness is important, for example, climate change and associated increases in sea temperatures are likely to result in a southerly extension of the geographical distribution of more tropical species. Such evidence already exists as recent recordings of tropical species, such as blackedged butterflyfish (*Chaetodon dolosus*) and whitespotted butterflyfish (*Chaetodon kleinii*) from PB, more than 500 km out of their previously described distributional ranges (Heemstra 1986). Furthermore, Green (1979) makes reference to numerous studies that have found species richness to be a better indicator of biological change than diversity indices.

It is important that the relationship between the indicator and environmental change or anthropogenic impact be understood, to show causality between predictor and dependent variables (Vos *et al*. 2000). Washington (1984) and Keough and Quinn (1991) suggested that many diversity measures have no biological meaning and that in many cases, there may be little reason to measure diversity, other than for political or social reasons. There is no reason to expect a relationship between measures of diversity and community stability (Keough and Quinn 1991) or environmental quality (Green 1979) and, in communities where species are not competitively equal, disturbance may even increase diversity. Furthermore, due to the nature of diversity calculations, information is lost, by way of a reduction in resolution (Green 1979). Although diversity indices are criticised by numerous authors (Green 1979, Washington 1984, Keough and Quinn 1991), such measures are commonly used in ecological studies and monitoring programmes. Despite its dubious suitability, diversity was included as an index in the current study, but only to allow for comparison with results of similar studies. Diversity measures were not included to compare communities from the protected and exploited study areas, but rather to detect changes in the community structure within each study area over time. Furthermore, Warwick and Clarke (1991) suggested that multivariate analysis (MVA), such as non-metric multidimensional scaling (MDS), is superior to univariate measures, such as diversity. Such analyses are more sensitive to change, and therefore provide an earlier response to environmental change, and were therefore chosen as the indicators in this study to detect community change.

5.2.3 Seasonality

In order to account for seasonal trends associated with oceanographic conduction, the protocol included both summer and winter sampling periods.

5.2.4 Sample site allocation

There were three options for allocation of UVC and fishing sites within each study area, each season. These were: a) sample the exact same sites for UVC and fishing, b) select UVC and fishing sites randomly or c) exclude any site that was already sampled by either method in that season. Diving and fishing the same sites in each study area would allow direct comparison of results from the two methods, e.g. community structure, diversity, species richness, density and CPUE. However, there is a lack of information on the effect of a baited hook on fish behaviour (Millar and Willis 1999). Thompson and Mapstone (2002) found that there was an effect of diver presence on fish behaviour up to three days after a dive. Totally random sampling would allow some sites to be sampled by both methods (with the possibility of diver or baited hook effects) but without removing the spatial variability resulting from sampling different sites with different methods, and preventing direct comparison of UVC and controlled fishing results. Therefore, it was decided not to dive sites that had been fished in that sampling season (or vice versa) due to possible effects of baited hooks on fishes observed, or of diver presence on the behaviour of fish around a baited hook. Furthermore, it was decided that a single site would not be sampled by the same method in consecutive seasons, to avoid the possibility of pseudoreplication or temporal autocorrelation. However, sampling a single site in consecutive seasons with alternate methods was considered because it was assumed that the effect of diver presence and baited hooks would be negligible after six months (Thompson and Mapstone 2002).

5.2.5 Sampling strategy

Ecological studies focusing on LTM and detection of temporal variability are commonly based on one of two sampling strategies. Fixed sampling sites (e.g. fishing stations or UVC sites) may be revisited during each consecutive sampling phase, or a new subset of sites may be randomly allocated on each occasion (Thompson and Mapstone 2002).

Revisiting fixed sites

Revisiting the same set of sites on each occasion allows for the exclusion of spatial variability (or change in spatial variability associated with sampling a different set of sites each season) (Greenwood 1996, Thompson and Mapstone 2002), thus increasing the statistical power of the experiment to detect temporal change (Vos *et al*. 2000).

However, it is possible that chosen sites may not be representative of the greater study area, suggesting that extrapolation of results to the entire study area should be interpreted with caution (Nowlis and Friedlander 2004). Chance disturbance, which may occur at one or a number of fixed sites, may result in erroneously high estimates of spatial or temporal variability, further suggesting that revisiting fixed sites is less suitable than random reallocation (Hurlbert 1984).

Further bias may be incurred with a revisiting strategy, as a result of trampling effects (Vos *et al*. 2000). Such bias may include anchor damage, cumulative capture-related mortality or habituation of subject individuals to diver presence. Evidence to suggest an effect of diver presence or baited hooks on fish behaviour in subsequent samples has been shown by Thompson and Mapstone (2002).

Furthermore, when revisiting fixed sites, the sample size is limited to the number of sites, whereas with random reallocation, the overall sample size increases with every sampling season or visit. This provides an increasing number of records (e.g. UVC counts at a greater number of depths), ultimately strengthening analyses on the effects of such variables on fish numbers.

Stratified random sampling with reallocation

Although overall variability may be decreased with a revisiting approach, due to the exclusion of spatial variability, and because the methods employed during this study (a single UVC count or fishing station) were likely to cause little trampling effect, a stratified random approach with reallocation of sites each season was preferred for a number of reasons:

1. It was decided that each site would not be sampled by both methods in a given season. Therefore, if the same fixed sites were revisited each season, it would

mean that all dive sites and fishing sites would be permanently spatially exclusive and consequently non-comparable. Therefore by randomly reallocating new sites each season (without replacement within season – to avoid pseudoreplication), all UVC sites and fishing stations would be sampled without the effect of a baited hook or diver presence on UVC or fishing results, respectively, while still allowing comparison of fishing and diving results.

2. GPS accuracy and variable water depth would make relocation of bottom waypoints (such as the starting point of a fixed strip transect) from the surface difficult (Götz 2005). Similarly, one cannot be sure that the exact location has been sampled on each occasion while fishing.
3. Furthermore, the high-energy nature of the sea conditions on the South African south coast negates the possibility of setting permanent sampling lines or marker buoys, particularly for detection by divers underwater. Coupled with this, relatively low gradient contiguous reef (such as that in both study areas) excludes the possibility of using physical landmarks to locate such waypoints, and having to locate these points while underwater would waste available bottom/nitrogen time during dives, which is already likely to be limiting, particularly in poor visibility.
4. Random reallocation allows the use of certain statistical analyses based on random sample allocation (Vos *et al.* 2000).
5. Reallocation of sites decreases the possibility of selecting non-representative or poorly representative sites, although reallocation may (in some seasons) provide less representative sites by chance (Vos *et al.* 2000).
6. Reallocation avoids the effects of chance disturbance at one or a number of fixed sites (Ault and Johnson 1998). Furthermore, although by chance, a site suffering disturbance may be selected in future seasons, if the disturbance is apparent or obvious, or at least known about, then such sites may be excluded from those available, prior to selection. A permanent change or disturbance at a fixed site would mean that that site would no longer be useful for comparison and all data collected up to that point would be of little value.
7. Random reallocation of samples in successive phases allows for the spatial spreading and dilution of any trampling effect or capture related mortality.

5.2.6 Sample stratification

Stratification refers to the spatial or temporal subdivision of sampling, according to different habitat types, such as by substrate type or depth, as well as different time periods, such as time of day, tidal cycle or season (Vos *et al*. 2000). Stratification allows more definitive identification of the factors associated with, and the causes underlying, environmental change and variability (Vos *et al*. 2000). Because variability within each stratum is likely to be lower than in the study area as a whole, stratification over heterogenous habitat generally results in lower overall variability and, consequently, more precise estimates (Green 1979). Furthermore, stratification provides results specific to each stratum, such as shallow low profile reef, and allows for extrapolation over all strata sampled (Green 1979, Lohr 1999).

Stratification provides more precise variability estimates specific to each stratum, which ultimately provides more insight into causal factors affecting abundance and biomass (Lohr 1999). Failure to stratify over highly heterogenous "treatments" will result in excessive variability in results, which may be difficult or impossible to distinguish from temporal variability.

Because detection of differences in species and community level parameters, caused by fishing in the exploited area, was one of the objectives of the sampling protocol, and because the benefits of protection have been well documented in the literature (Buxton and Smale 1989, Garcia-Rubies and Zabala 1990, Bennett and Attwood 1991, Russ and Alcala 1996, Willis *et al*. 2000, Parsons *et al*. 2003), stratification over protection status was most important.

Depth has been shown to be a major factor affecting the distribution of fish abundance (Thresher 1983, Roberts and Ormond 1987, McCormick 1994, Friedlander *et al*. 2003) and sizes (Götz 2005). UVC dives and fishing stations were therefore stratified over depth, which was classed as a categorical variable (from the mapping data in ArcView), but measured as a continuous variable.

Reef profile, or vertical relief, has also been shown to be a major predictor of abundance (Carpenter *et al*. 1981, Grigg 1994, Friedlander and Parrish 1998, Ohman and Rajasuriya 1998, Almany 2004, Gratwicke and Speight 2005a). Samples were,

therefore, also stratified over reef profile, which was calculated using the slope function in ArcView, and defined as either high or low.

From the preliminary sampling to compare methods, it was evident that variability was lower in winter than in summer. It was, therefore, necessary to stratify sampling over season, to provide seasonal estimates of abundance, consequently lowering overall variability.

Although substrate has been shown to affect fish abundance (Guidetti 2000, Gratwicke and Speight 2005a), samples were not stratified over substrate, as the study focused on reef-associated species.

Time of day has also been shown to affect fish abundance (Colton and Alevizon 1981). However, due to typical South African weather and sea conditions (e.g. strong wind and rough seas), time at sea is often limited. Therefore, it was impractical to stratify strictly over time of day. Time of day was therefore treated as a random effect. Similarly, temperature and visibility were treated as random effects.

5.2.7 Sample size

Green (1979) suggests that the best sample size is the largest. However, in any LTM programme there is a trade-off between the minimum sample size required for statistical strength, and the maximum number of samples that can be conducted within the financial, time and manpower constraints of the programme.

There is also a trade-off between the number of samples and the size of each sampling area. In this case, it is better to have more samples of smaller size than fewer large samples, to sample a given total area, particularly when sampling a population with non-random distribution (Green 1979, Sutherland 1996b).

Increased sample size is beneficial in that most statistical analyses are more robust in the event of violations of assumptions if they are based on data with a greater sample size, as a result of a greater number of error degrees of freedom (Green 1979). Furthermore, the central limit theorem suggests that as sample size increases, the distribution of estimates tends towards the normal (Rosner 2000).

It is also important that the sample size is sufficient to provide statistical power necessary to detect changes in population estimates of specific magnitude, at a certain level of significance. Statistical power is the ability of the experiment to detect a predetermined level of change (Vos *et al*. 2000), and is inversely related to the probability of making a type II error (i.e. failing to detect a change or impact when such a change or impact has occurred) (Keough and Quinn 1991). The power of an experiment is dependent on the significance level set (i.e. alpha), the sample size, the effect level to detect and the natural variability in the population (Keough and Quinn 1991). Calculations to determine the sample size required, to obtain a predetermined level of power, are provided in the literature (e.g. Greenwood 1996, Lohr 1999, Bausell and Li 2002, Kapadia *et al*. 2005).

A power analysis was conducted to determine the sample size required by point count and transect count UVC techniques (Chapter 4), to determine which method was more efficient. Results of this power analysis showed that 11 transect count samples were required to detect a change of 10% in the mean population estimate, with a power of 80%. Twelve samples per study area would have provided three samples per stratum (stratified over depth and profile), per study area, i.e. 24 samples per season overall. However, a sample size of 30 is suggested to be a suitable minimum, as the *t*-statistic with a sample size of 30 or greater is suggested to approach that of the normal (Lohr 1999). Therefore, it was decided to increase the number of samples to 32 (i.e. four per stratum or 16 per area), for increased statistical strength. A sample size of 32 was also selected for the controlled fishing.

It was decided that two transects would be conducted during each dive (each counted by a different diver), to increase the overall area sampled without having to increase sample size. Furthermore, having two divers counting on a dive reduced between diver error (i.e. the subjectivity in abundance estimates was not restricted to that of a single observer, but rather of two observers, therefore reducing diver associated error). This can also reduce time required per dive if time becomes limiting, as each diver can sample one of the transects, allowing the two transects to be completed concurrently, which would reduce bottom time, allowing more dives to be conducted in a given time, if necessary.

5.2.8 Fishing techniques

Optimal fishing station effort was investigated in Chapter 4. This study showed that two angler hours per station provided low variability with low required time and cost. Two angler hours per station was therefore selected as the sampling effort for the proposed LTM sampling protocol.

Because of uncertainties regarding the area fished by a baited hook and the behaviour of fishes in the presence of one or more baited hooks, it remains unclear whether the absolute 'effort' applied by a single angler fishing for two hours, or two anglers for one hour, or even four anglers for 30 minutes is equal (Millar and Willis 1999). However, in the absence of evidence suggesting these might not exert equal effort it was assumed that exerted effort would remain constant with a change in the number of anglers, provided the overall number of angler hours was kept constant. Therefore, although the optimum fishing station effort of two angler hours was calculated based on two anglers (i.e. absolute duration of one hour), the distribution of effort chosen for the proposed LTM protocol (although still two angler hours) was four anglers fishing for 30 minutes. This was based on the fact that four anglers fishing for 30 minutes would require significantly less time to complete the required 16 fishing stations than that for two anglers to complete 16 one-hour stations. This decrease in time required to sample each station, and thus the overall time required to sample all fishing stations, without decreasing the effort at each station, could have the added benefit of decreasing the temporal variability within each sampling phase.

In South Africa, the Occupational Health and Safety Act (Act No. 85 of 1993) states that scientific diving (including research diving for UVC) requires a minimum of four appropriately trained personnel (see Appendix III) to be present during dive operations. This suggests that there would be a minimum of four researchers available on any field trip. Fishing stations making use of four anglers for 30 minutes would therefore be a more efficient use of resources than two anglers for one hour, provided the same four individuals could be used for diving and fishing.

Circle hooks were used in the sampling to determine optimal fishing station effort. Although they are expensive, there is extensive literature that advocates the use of circle hooks in release fisheries (*inter alia* Prince *et al*. 2002, Skomal *et al*. 2002,

Anon. 2003). The benefits of circle hooks include increased post-release survival (Prince *et al*. 2007), increased occurrence of 'jaw-hooking' (Falterman and Graves 2002), decreased incidence of gut or throat hooking (Zimmerman and Bochonek 2002) and decreased hook-related bleeding or injury (Domeier *et al*. 2003), with no associated decrease in catch rates (Cooke and Suski 2004). However, Cooke and Suski (2004) warn that circle hooks may not be superior to straight shank "J" hooks for all species, and recommend the use of circle hooks only for species for which there is scientific evidence of their benefit. This should be determined prior to implementation of LTM, for all target species.

Barbs should be removed to decrease time and injury associated with hook-removal (Schaeffer and Hoffman 2002) and the point of the hook should not be offset, as this has been shown to increase the incidence of deep hooking (Prince *et al*. 2002). The use of barbless circle hooks with no offset point was, therefore, included in the proposed protocol.

The hook size selected for this study was 4/0, as this was shown by Götz (2005) to be the most suitable for roman, the indicator species. The use of inconsistent hook sizes in different areas would prevent comparison between such areas. However, it may be necessary to determine experimentally an optimal hook size for each LTM programme, as this is likely to vary with area or indicator species. Alternatively, it may be better to use a range of hook sizes, including hooks of size suitable for a wide range of species and areas.

Pilchard and squid were used in conjunction during preliminary sampling. However, Götz (2005) found no significant differences in catch between pilchard and squid baits. During implementation and assessment of the proposed protocol, bait would be restricted to pilchard, as this was found to be suitable for roman in the Goukamma MPA (Götz 2005). Pilchard is likely to be suitable for most warm temperate reef areas in South Africa, due to the high contribution of roman and other sparids with similar feeding biology, throughout this region (Götz 2005, Smith 2005, Mann *et al*. 2006). However, it is recommended that the suitability of different bait types be tested for each LTM study area.

5.2.9 Selection of UVC technique

The sampling efficiencies (in terms of number of fish observed per replicate and unit time, and number of species observed per replicate), variability, bias and required sample size of point count and transect count UVC techniques were investigated in Chapter 4. Transects were found to be superior to point counts in all aspects, and were therefore chosen as the preferred technique for the proposed LTM protocol.

5.2.10 Minimum visibility

Counts during preliminary sampling were restricted to dives during which visibility (estimated as the horizontal distance along the transect line from where the shot line anchor first became visible) was at least 3 m. However, visibility was regularly observed to be spatially variable, even within individual transect counts. A minimum of 3 m was therefore insufficient, as there were patches where visibility decreased below this, encroaching on the census area. It was, therefore, decided that minimum visibility should be greater than 3 m to allow for patchy areas of lower visibility, to prevent the census area from being encroached upon. During preliminary sampling in the protected area, 94% of dives during which counts were made (n = 33) had visibility of 4 m, suggesting a loss of only 6% of possible samples if minimum visibility to allow sampling was increased to 4 m, compared to a loss of 19% of otherwise suitable conditions if minimum visibility was restricted to 5 m (estimated on only 81% of dives) (Fig. 5.2).

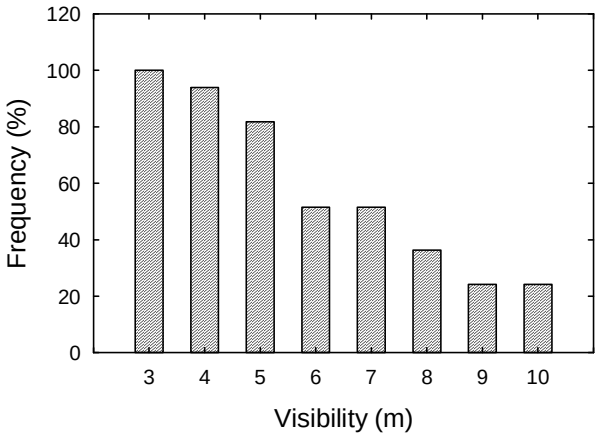

Figure 5.2: Cumulative percent frequencies for visibility (metres) during 33 preliminary sampling dives in the TNP, during the period June 2005 to December 2006.

Therefore, for LTM it was suggested that sampling be restricted to dives during which visibility is at least 4 m, to allow at least 1 m of visibility outside of the census area, for two reasons: 1) so that areas of patchy visibility do not encroach on the census area, and 2) so that fishes entering the census area from the sides of the transect can be detected and recognised as such, and excluded from counts (Froeschke *et al*. 2006).

5.2.11 Census area size

Watson and Quinn (1997) suggested that for UVC the number of sample sites and the size of each site for estimating abundance are not important, and rather that the estimation of variance is a function only of the total area sampled, suggesting that variability is a reflection of the heterogeneity of the total sampled area, as if it were a single unit. However, where fishes are not randomly distributed (as with many reef fishes), census area size and sample size are criteria that must be considered.

When sampling a species with non-random spatial distribution, a small census area should provide lower variability and may, therefore, be more suitable to allow detection of temporal variability (Green 1979, Ault and Johnson 1998). However, the census area must be large enough to be representative of the habitat, abundance and distribution of target populations, and to avoid too many zero counts (Zeller and Russ 1998), i.e. UVC counts should be high enough to be meaningful, to allow variability to be attributed to a cause rather than chance (Sutherland 1996a, b).

Although a wide range of transect census area dimensions has been used, a transect length of 50 m appears to be the most commonly used for reef fishes (Jones 1988, Grigg 1994, Samoilys 1997, Ohman and Rajasuriya 1998, Barrett and Buxton 2002, Froeschke *et al*. 2006). Zeller and Russ (2000) suggested 50 m for mobile (as opposed to cryptic) species. La Mesa and Vacchi (1999) used 50 m x 5 m transects for estimating abundance of *Epinephelus* and *Diplodus* species. Samoilys and Carlos (1995) and Mapstone and Ayling (1993) proposed 50 m x 5m for fish greater than 11 cm FL. Transects of 50 m were used to compare transect and point count UVC techniques in this study (Chapter 4) and provided satisfactory counts for most species.

It was also important to select a suitable transect width, as this has been shown to affect the precision and accuracy of UVC counts (Zeller and Russ 2000), and the use

of variable strip width (variable distance counts) prevents comparisons of variability with other studies (Mann *et al*. 2006). A wider strip would provide a larger census area and a resultant higher number of observations. However, subject detectability decreases with increased strip width, particularly in poor visibility (Thresher and Gunn 1986). The maximum strip was governed by predominant visibility, which, in the current study area, is commonly no more than 4 m. Therefore, a strip width of 3 m on either side of the diver was chosen, providing a transect census area of 50 m × 6 m.

5.2.12 Measures of topographic complexity

Numerous measures of topographic complexity have been used (*inter alia* Risk 1972, Underwood and Chapman 1989, McCormick 1994). However, inconsistency in measures has prevented comparison between studies, thus highlighting the need for a standardised method. This would allow different users to employ similar or exact methods to allow comparison between studies and allow a single user to compare two areas with different ranges of topographic complexity or different complexity maxima and minima.

The most commonly used measure of topographic complexity is the chain link method (Luckhurst and Luckhurst 1978, Sale and Douglas 1984, Connell and Kingsford 1998, Friedlander and Parrish 1998, Ferreira *et al*. 2001, Almany 2004), described by Risk (1972), in which the ratio of contour length to linear distance is calculated by measuring the actual length of chain required to cover a given linear distance over the reef. However, the method is most commonly used for small surface areas, such as quadrats, and is therefore not suitable for determining topographic complexity of the census area covered by a 50 m transect (Bell and Galzin 1984). A chain of sufficient length to measure the actual contour distance of a 50 m transect would be unmanageable for divers underwater. Furthermore, the method is time-consuming and would increase required dive time (Risk 1972).

Two measures of topographic complexity were proposed during this study. The profile of each UVC site would be calculated as the sum of squares of the differences between consecutive depth readings (McCormick 1994) taken by divers at 10 m intervals along the 110 m spanning the two 50 m transects and the two 5 m buffer sections between the shot line and the start of each census area. Due to the nature of

the calculation, this method is able to distinguish between a level seafloor and one that deviates from level, providing information on the angle of slope, but is unable to distinguish between a sloping seafloor with even surface and a level seafloor with uneven surface. This concept is illustrated simply in Figure 5.3.

The solid line represents a sloping seafloor with an even surface (seafloor A), while the dashed line represents a level (i.e. no net slope overall) seafloor with uneven surface (seafloor B) and the dotted line represents a level seafloor of even surface (seafloor C).

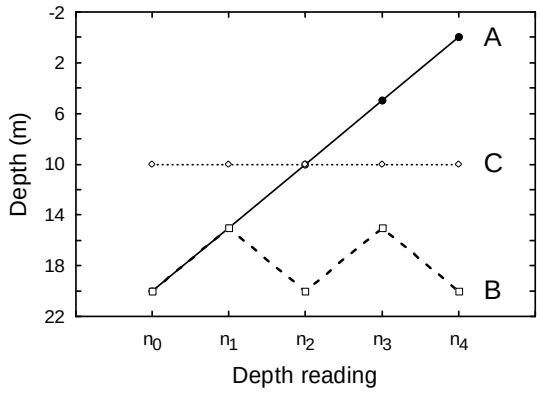

Figure 5.3: Schematic representation of seafloor A (sloping with even surface) represented by the solid line, seafloor B (level with uneven surface) represented by the dashed line, and seafloor C (level with even surface) represented by the dotted line.

Differences between consecutive depth readings were calculated as $n_1 - n_0$, $n_2 - n_1$, $n_3 - n_2$ and $n_4 - n_3$. Values were calculated as -5, -5, -5 and -5 for seafloor A, -5, 5, -5 and 5 for seafloor B, and 0, 0, 0 and 0 for seafloor C. Therefore, profile, calculated according to McCormick (1994), provided values of 100, 100 and 0, for seafloors A, B and C, respectively, illustrating how the method can distinguish between seafloors of different slope (seafloors A and C), but not between a sloping even seafloor (A) and a level uneven seafloor (B). The chain link method (described by Risk 1972) would also not have been able to distinguish between seafloors A and B.

This suggested that a second measure of topographic complexity would be necessary, referred to in this study as rugosity. The SD of the depth readings recorded along the 110 m transect lines would provide no information on the spatial arrangements of

depths, i.e. depth readings relative to adjacent readings (McCormick 1994). Therefore, rugosity was calculated as the absolute value of the SD (as opposed to sum of squares) of the differences between consecutive depth readings. This is based on the fact that an even seafloor of any slope would provide a lower rugosity value than a level uneven seafloor, and that the magnitude of unevenness would be reflected in the calculated value. This can again be illustrated using the example in Figure 5.3.

Rugosity was calculated, using the SD of the differences between consecutive depth readings, as 0, 5.77 and 0 for seafloors A, B and C, respectively, illustrating the ability of this method to distinguish between seafloors of variable surface evenness, thus providing a suitable measure of rugosity, as opposed to seafloor slope.

Although there are other more complicated and expensive methods available to calculate or estimate profile, such as those described by Brock *et al*. (2004) and Kuffner *et al*. (2007), dive time during the current study was limited, and the method for calculating profile (McCormick 1994) and that developed for calculating rugosity would provide simple, yet suitable, time efficient estimates.

5.2.13 Size of sample site cell

It was important that each sample be sufficiently spatially segregated to avoid pseudoreplication or spatial autocorrelation and that each sample site be selected randomly. The two study areas had to be divided into units, or cells, of equal size, within which a single sample would be conducted. Each cell constituted a possible sample site. Cells had to be large enough to ensure that sampling could be conducted within its boundaries, to avoid pseudoreplication or spatial autocorrelation (McArdle *et al*. 1990). At the same time the cells had to be small enough to provide minimal within-cell spatial variability (i.e. minimum habitat heterogeneity), and to maximise the number of cells available within the overall study area.

Minimum cell size for diving was governed by the length of the census area, GPS error and displacement of the shot line anchor after deployment (which was governed by current strength and water depth, but determined to be no greater than 10 m). The census area required a length of 55 m (50 m transect + 5 m buffer area). GPS error for the Garmin GPS 60 was determined as approximately 5 m (Götz 2005). Therefore, a

minimum total distance of at least 70 m (10 + 55 + 5) was required from the centre of the cell to its boundary, for diving, in the event of all displacements being linear in the same direction.

Minimum cell size for fishing was governed by GPS error (5 m), displacement of the anchor after deployment (also assumed to be no greater than 10 m) and the length of anchor rope deployed, which is influenced by depth, wind strength, current strength and swell size. However, the angle between the anchor rope and the sea surface during preliminary sampling was never less than 45 degrees, and thus the maximum horizontal displacement of the boat from the anchor position was never greater than the maximum water depth of approximately 35 m (Pythagoras's Theorem). Therefore, for fishing stations, a minimum distance of 50 m (5 + 10 + 35) from the centre of the cell to the boundary was required if all displacements happened in the same linear direction.

Furthermore, Kerwath *et al*. (2007) found the linear extent of roman (the chosen indicator species) home range to be no more than 100 m, suggesting a minimum cell size for sampling of 100 m x 100 m. Cell size was therefore set at 150 m x 150 m, based on the requirements for diving.

5.2.14 Temporal and spatial extent of sampling

Variability in results will be affected by the temporal and spatial extent of sampling. Therefore, when designing monitoring experiments, it is important that suitable spatial and temporal extents (and scales) are chosen to meet the objectives of the study (McArdle *et al*. 1990).

Sampling at a single site on a number of occasions will provide an estimate of temporal variability, which is spatially pseudoreplicated. Similarly, sampling at a number of sites, but on only one occasion, will provide an estimate of spatial variability that is temporally pseudoreplicated (Stewart-Oaten *et al*. 1986). It is, therefore, important that the spatial and temporal extents of samples are sufficient to provide representative results (McArdle *et al*. 1990).

Temporal scale

Sampling, to detect changes in a mean estimate, should be conducted frequently over an extended period. Estimates made infrequently or over a short term may erroneously indicate changes in the mean estimate, when estimates are simply fluctuating around a constant mean (Underwood 1991). Conversely, it is also important that samples are sufficiently temporally separated to avoid temporal autocorrelation; a factor of particular consideration when sampling involves revisiting fixed sites (McArdle *et al*. 1990).

Diurnal, daily, monthly, seasonal and annual temporal variability each result from different sources and it may, therefore, be sensible to focus on only one or two temporal scales in which the project has invested interest (Stewart-Oaten *et al*. 1995). A suitable temporal unit of measurement for LTM of reef fishes is suggested to be the generation time of the species of interest (Connell and Sousa 1983, McArdle *et al*. 1990). Therefore, in monitoring programmes aimed at detecting annual temporal change, specifically in species with a single annual spawning season, a suitable temporal scale would be annual. However, from the literature and preliminary sampling it appeared necessary to stratify sampling over season to provide variability estimates for winter and summer (see section 5.2.3).

Spatial scale

The spatial scale of sampling should be related to the range of movement of individuals of the study species (Stewart-Oaten *et al*. 1995). Studies by Buxton and Allen (1989), Kerwath *et al*. (2007) and Griffiths and Wilke (2002), focusing on the movement patterns of roman, found maximum movements of 2 km and 4 km, and a modal displacement of 6 km, respectively, suggesting that a study area of linear length between 2 km and 6 km would be suitable for monitoring of this species, and probably other reef-associated species as well.

The maximum extent of the study area, however, should be kept as small as possible, to minimise spatial variability. Therefore the maximum extent is limited by the maximum spatial variability that can be allowed, while the minimum extent is governed by that which allows sufficient independent samples and which is representative of the overall region or study area.

The spatial extent had to allow sufficient cells for future seasons and years, and additional cells that may be necessary in the event of sand covering areas of rock, as a result of longshore drift (Schumann 1987). Evidence for this was provided by the difference in substrate type determined by SSS (Schumann *et al*. 1982, Martin and Flemming 1986) and observation dives made during the current study. Approximately 30% of the cells in the TNP appearing from the SSS data to be reef were shown to be covered by sand during observation SCUBA dives. Therefore, assuming an extra 30% of cells were required each season for cells covered by sand, and a further 30% for cells that are too deep or inaccessible, a total of approximately 52 cells per study area per season (i.e. 104) per year would be required. The spatial extent of each study area, therefore, had to be large enough to encompass a maximum of approximately 104 cells.

5.2.15 Replication

Replication increases precision by reducing the effects of random variability in estimates and supplies an error term (i.e. a variability estimate), which can be used to determine whether differences are significant or not (Hurlbert 1984). However, the cost of replication is usually excessive. Therefore, LTM studies must be designed in such a way that the experiment has the power (statistical and diagnostic) to detect change and meet the objectives, within the given time and financial constraints (Hurlbert 1984).

5.2.16 Data analysis

Data analysis methods were only selected once the sampling approach, methodology and scale components had been determined or selected.

Species-level analysis

UVC counts commonly produce highly variable results (Samoilys and Carlos 2000, Mann *et al*. 2006) and are, therefore, unlikely to meet the assumptions of homogeneity of variance required for parametric statistics. Comparisons of abundance (observed density and CPUE) between protected and exploited areas should, in this case, be compared using non-parametric Mann-Whitney U tests.

A stock density index was chosen as an indicator of population status (ICES 2006), to show the effect of fishing on the sizes of individuals captured. The index is calculated as the ratio of individuals of fork length (FL) greater than a certain specified length to the total number of individuals captured within each study area (Rochet and Trenkel 2003). A specified FL of 270 mm was selected as this is the FL that corresponds approximately to the minimum legal size (300 mm TL) for roman (Buxton 1993a). Length-frequency distributions were also chosen to illustrate differences in size structure.

Generalised linear models (GLM) were chosen to identify and separate the effects of selected environmental variables (continuous and categorical) on UVC counts, controlled fishing catch and size of individuals captured, in the two study areas. Because GLMs exclude the covariate effects of other (measured) factors and provide results based on what is predicted to be the effect of each factor independently, they provide the ability to link observed trends to causal factors, and therefore provide stronger diagnostic power (Vos *et al*. 2000). GLMs were, therefore, suitable to determine which variables should be recorded or monitored in the LTM programme. Count data, such as that obtained from diving surveys and controlled fishing, may include frequent zero counts (Seavy *et al*. 2005). GLMs apply linear regression techniques to nonlinear data with heterogenous variances (Willis *et al* 2000), and are suitable for such analyses.

Community analysis

For comparison of diversity from UVC counts and CPUE fishing, diversity indices selected were Margalef's (1958) overall diversity index (*d*), the Shannon-Wiener (Shannon 1949) diversity index (*H'*) and Pielou's (1966) evenness (*J*). Margalef's *d* is a measure of the number of species present relative to the total number of individuals in that sample. The Shannon-Wiener diversity index takes into account the relative proportions of each species, and Pielou's *J* is a measure of how evenly the individuals are distributed among the different species. Although diversity measures have received much criticism, the chosen methods appeared most suitable (Washington 1984). Non-parametric MVA was, therefore, also chosen for analysis of fish community data from UVC and controlled fishing surveys. Such analyses would determine whether MVA was suitable for LTM within the protected area or within the

exploited area (in which the protected area would act as a control, against which the exploited area would be compared).

5.3 Proposed protocol

Based on the above review and preliminary investigation (Chapter 4) the following protocol was developed for monitoring of temperate reef fish communities in the warm temperate biogeographical region:

1. The protected and exploited study areas were identified in Chapter 3.
2. Roman was selected as an indicator species due to the level of information on the species, particularly the effects of fishing on its life-history parameters (Chapter 4).
3. It was decided that sample sites should not be dived and fished in the same season, or sampled by the same method in consecutive seasons.
4. Preliminary results showed that sampling was necessary in summer and winter (Chapter 4).
5. Samples would be randomly reallocated during each season, i.e. no fixed site sampling.
6. Samples were to be stratified evenly over protection status, depth, profile (calculated in ArcView) and season.
7. Sample size was set at 16 UVC and 16 fishing station samples per study area, per season, determined by power analysis.
8. Fishing was set at two angler hours per station, with four anglers fishing for 30 minutes, using barbless 4/0 circle hooks baited with pilchard.
9. Transect counts were selected in favour of point counts, based on variability and efficiency.
10. Minimum visibility for UVC was set at 4 m, below which dives would be aborted.
11. The transect census area size was set at 300 m^2 (50 m × 6 m).
12. Profile and rugosity would be measured as the sum of squares and SD of the differences between consecutive depth readings, respectively.

13. The size of each sample site cell was set at 150 m × 150 m, based on minimum requirements of UVC counts.
14. The temporal extent was not limited, as the programme was designed for LTM, but sampling was to be conducted seasonally. The spatial extent of each study area was large enough to encompass approximately 104 sample cell sites, extending no more than 3 km along the shoreline.
15. Treatments could not be replicated due to the scale of the sampling objectives.
16. GLMs and non-parametric Mann-Whitney U tests were selected for comparison of species abundance, and stock density ratios, length-frequency histograms and GLMs were selected for comparisons of size structure. Diversity, species richness and MVA were selected for comparisons of community structure.

The next step was to test the protocol during preliminary sampling in the field, in order to determine the logistical and financial feasibility and efficiency of its design, and to identify any possible fatal flaws or shortcomings.

Chapter 6
Implementation and Evaluation of Proposed Protocol

6.1 Introduction

Greenwood (1996) highlighted the importance of preliminary sampling, to assess a proposed sampling protocol and determine potential problems, before full-scale implementation. Preliminary sampling can also be used to determine the sample size required for certain predetermined statistical power and whether stratification is necessary (Green 1979).

The aim of this chapter was to assess the effectiveness and suitability of the proposed sampling protocol. This was achieved by addressing the following objectives:

1. implementing the proposed sampling protocol in the protected (TNP) and exploited (PB) study areas,
2. determining whether the methods and analyses employed were able to detect spatial differences between protected and exploited study areas, known from previous studies to exist (Burger 1990, Smith 2005),
3. determining whether the protocol was suitable for LTM in the protected area (to detect natural change) and identifying suitable community-level indicators,
4. determining whether the protocol was suitable for LTM in the exploited area (using the protected area as a control) and identifying suitable species-level indicators,
5. determining what environmental variables should be measured,
6. determining whether habitat stratification is necessary, and
7. determining whether sample size is sufficient to provide the desired precision.

6.2 Methods and materials

6.2.1 Site identification and mapping

Side scan sonar data, a remote underwater camera lens and observation dives on SCUBA were used to provide a comprehensive map of depth and substrate type.

These assessments were used to identify suitable sampling sites within the two study areas, and to define the lateral, offshore and inshore extents of each study area, which were governed by the extent of reef area, depth and boat accessibility inshore, respectively.

The chosen sampling sites, within each of the reef complexes identified, were topographically surveyed by means of vessel transects conducted with a handheld GPS and the boat's echo-sounder. This provided depth and substrate data for a series of GPS coordinates, taken while the boat followed a tight grid (lines approximately 20 m apart). Survey speed was maintained at approximately 10 $km \bullet h^{-1}$ to reduce GPS error (Götz 2005). The depth and physiography data were imported into ArcView to display the mapping points spatially. A study area polygon was created to encompass each study area. The spline tension method in Spatial Analyst 2.0a (Environmental Systems Research Institute) was used to interpolate depth data within each polygon, from which continuous seafloor maps were created for each study area (Fig. 6.1).

The protected study area (TNP) encompassed a 2.9 km^2 area of reef. During the study period, visibility in the protected study area ranged between 4 and 10 meters and temperature ranged between 19° C and 21° C. The exploited study area (PB) encompassed a 3.2 km^2 area of reef. Visibility in the exploited area during the study ranged between 8 and 10 m and water temperature between 19° C and 24° C. Reef profile and complexity were similar in the two study areas, and depths ranged between three and 36 m in both study areas.

6.2.2 Stratification and randomised sample site allocation

A grid with cells of 150 m x 150 m was overlaid onto the study area polygon. Each cell constituted a possible sample site. Each row and column was numbered so that each cell had a unique bi-coordinate number. Sampling was stratified over profile (calculated as slope in ArcView for each grid cell) and depth (determined from the continuous seafloor), which were categorised as low or high profile and shallow or deep, respectively, thereby providing four strata within each study area. Each bi-coordinate cell number was then numbered with a positive integer, and cells were chosen using a random number generator so that all cells had an equal chance of being selected and that no subjectivity was placed on cell site selection (Greenwood

1996). Four diving and four fishing cells were randomly assigned to each stratum, within each study area. Extra cells were subsequently added to each stratum, for use in the event that one or more of the four cells in each stratum could not be sampled, for reasons such as being too close inshore to approach with the boat (i.e. in the surfzone), depths in excess of safe diving limits or substrate appearing as sand on the echo-sounder. GPS coordinates for the centre of each sample cell were obtained from the grid in ArcView and entered as waypoints into a handheld GPS. Sites were sampled in a random order but such that each stratum was represented only once in every four consecutive samples of each method. All fishing and diving was conducted from a 6 m skiboat.

6.2.3 Underwater visual census

Underwater visual census sites were located using the handheld GPS. When the boat was on position, the depth and substrate were determined using the boat's echo-sounder and, if suitable for diving, a shot line was deployed. The shot line consisted of a 5 kg anchor attached to a 15 mm diameter nylon rope of variable length (adjusted according to depth) and a surface marker buoy. The boat was not anchored. Two divers descended the shot line together, and once at the bottom attached a dive reel to the shot line anchor, then swam away from the anchor. Transects were conducted approximately parallel to the coastline to avoid excessive depth ranges on each transect. Counting commenced once the divers had moved a distance of 5 m from the anchor. This five-metre section acted as a buffer for effects of shot line deployment and diver presence on fish behaviour. One diver swam ahead in a straight line, recording on a Perspex slate all fishes observed within a six-metre wide strip (i.e. 3 m on either side of the diver). Only fishes inside the six-metre strip and in front of the diver were recorded. Fishes that passed the diver from behind were not counted, as it could not be determined whether these individuals had already been recorded (Froeschke *et al*. 2006). The second diver reeled out 50 m of transect line and once diver one had traversed 50 m (i.e. 55 m from the shot line anchor) sampling was terminated. While returning along the transect line, diver one recorded depths every 10 m with a dive computer (SCUBAPRO UWATEC Xtender). At each point, habitat was recorded as rock or sand. Visibility was estimated as the distance from which the shot line anchor first became visible. A second transect was then conducted in the same way as the first, but in the opposite direction from the shot line, with diver two

recording and diver one reeling out line. Again depth and habitat were recorded every 10 m, giving a total of 12 depth recordings, from which profile, rugosity and substrate type were calculated. Each dive took approximately 16 minutes to complete.

6.2.4 Controlled fishing

Fishing sites were located using the handheld GPS and when on position, depth and substrate were determined using the boat's echo-sounder. If depth and substrate were suitable, the anchor was deployed. Fishing commenced only once wind and current drift had resulted in the boat reaching a stationary position, and it was certain that the anchor was not dragging. In the event of the anchor dragging it was lifted and an attempt was made to re-anchor on the centre of the same cell and, if this was unsuccessful, the fishing station was moved to the next randomly chosen cell within that stratum.

Four anglers each used a standardised hook-and-line configuration (as described in Chapter 4), baited with pilchard. A total effort of two angler-hours was exerted at each of the sixteen stations within each study area (i.e. four anglers for 30 minutes).

6.2.5 Data analysis

Abundance

Mean density of fish from UVC counts (expressed as fish per 100 m^2) and mean CPUE (fish per angler hour) from fishing stations were compared between the protected and exploited areas to determine if there were significant differences. F-tests for UVC and fishing data showed that variances from the two study areas, for all species, were not equal. Density and CPUE data were therefore compared using non-parametric Mann-Whitney U tests.

Diversity and species richness

Margalef's (1958) *d* is calculated as:

$$d = \frac{(S-1)}{\log(N)},$$

where *d* is Margalef's overall diversity index, *S* is the number of species in the sample and *N* is the number of individuals in the sample.

Shannon's (1949) *H'* is calculated as:

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

where *H'* is the Shannon-Wiener diversity index and p_i is the proportion of the total count of each sample represented by the i^{th} species (Shannon 1949). Pielou's (1966) *J* is calculated as:

$$J = \frac{H'}{\ln S},$$

where *J* is Pielou's evenness, *H'* is the Shannon-Wiener diversity index (Shannon 1949) and *S* is the total number of species (Pielou 1966). Indices were calculated in PRIMER 6.1.6 (Plymouth Marine Laboratories).

Diversity indices were compared using the non-parametric Mann-Whitney U test, as a prior F-test had shown inequality of variances between the two study areas, for all three diversity measures. Analyses were run in Statistica 7.1.

Two measures of species richness were used in comparisons between protected and exploited areas; the overall total number of species observed in all UVC counts and captured in all fishing stations, and the mean number of species observed per count and captured per fishing station.

Size and age structure

A stock density index was used to assess the effects of fishing on population size structure (ICES 2006). This was calculated as the ratio of individuals greater than 270 mm FL to the total catch. This was calculated for the indicator species (roman) and all species combined. Due to characteristically greater lengths, all shark species were excluded from this analysis. Migratory species were also excluded. Length-frequency distributions were compared between protected and exploited areas to further assess the effects of fishing on population size structure. Age-frequency distributions and average age were also compared between protected and exploited areas.

Community analysis

Fish community data from UVC and controlled fishing surveys were analysed using non-parametric MVA run in PRIMER 6.1.6. Count data were root-root transformed and catch data were root transformed, before creating Bray-Curtis similarity matrices for density and CPUE, respectively, using the Resemblance function in PRIMER (Bray and Curtis 1957). Such transformation was required to account for high variabilities associated with the count or catch of a few very abundant species (Clarke and Warwick 1994). Cluster dendograms were produced using a group average hierarchical sorting strategy according to Cormack (1971). Density and CPUE results for the 32 diving and fishing stations, respectively, were compared using analysis of similarities (ANOSIM), to test for differences in community structure between the protected and exploited areas. Non-metric multi-dimensional scaling analysis (MDS) was applied to aid illustration of results. The SIMPER (similarity percentage breakdown) analysis was conducted to determine which species contributed most to the observed clustering (Clarke and Warwick 1994). Dominance curves (ranked species abundance curves) were plotted to rank species in order of importance.

Generalised liner models

Generalised linear models (GLMs) were used to identify and separate the effects of selected environmental variables on observed density, CPUE and size of individuals captured, in the two study areas, and to determine which environmental variables required recording. Before each GLM was performed, Akaike's Information Criterion (AIC, Akaike 1973) was used to determine which combination of measured variables provided the best fit to the data (Johnson and Omland 2004). The AIC combines a negative log-likelihood, which measures the lack of model fit to the observed data, and a bias correction factor, which increases as a function of the number of model parameters and takes the form:

$$AIC = -2\ln\left[L\left(\theta_p|y\right)\right] + 2p,$$

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

Vos *et al*. (2000) suggested that only variables that provide an early warning system or help to identify causes of environmental change should be measured, rather than what they referred to as *datakleptomania*, "for an unspecified better understanding of the system". Furthermore, Crawley (1993) suggested there should be fewer than n/3 factors in a GLM (where n is sample size). Therefore, prior to performing GLM calculations, all continuous variables recorded during sampling were entered into a correlation matrix to determine whether there were correlations between any of the measured variables, in which case one of the 'redundant' variables in each correlation could be excluded, in order to simplify the model (Crawley 1993).

For LTM, it is important to run GLMs on main effects (variables) and on secondary interactions between main effects. In this study, GLMs were run on the main effects to determine which variables should be measured for LTM. Between effects were less important, but were run anyway (separately to main effects) to provide an indication of whether the sampling strategy was suitable. Because GLMs based on small sample sizes are compromised by statistical over-fitting when too many factors are modelled, only those between effects that included protection status were included in the GLMs (the effect of protection status was of greatest focus). Furthermore, it was known that the study areas were of similar rugosity, profile and depth ranges, and temperature and visibility ranges were narrow. All GLM analyses were run in Statistica 7.1.

For the UVC data (based on the number of individuals observed per dive), the response codes were count variables. Therefore, the distribution of the data were assumed to be Poisson, for which the commonly used link function is the log-link (McCullagh and Nelder 1995):

$$\eta = \log \mu ,$$

where η is the linear predictor and μ is the population mean. To model the effects of the measured variables on UVC data, factors were included in a GLM of the form:

$$\begin{aligned}\log(Count) = \beta_0 + \beta_1(Depth) + \beta_2(Temperature) + \beta_3(Rugosity) + \beta_4(Profile) \\ + \beta_5(Substrate) + \beta_6(Visibility) + \beta_7(TimeOfDay) + \beta_8(Status) + \varepsilon\end{aligned}$$

where the β_i values were the estimated coefficients (McCullagh and Nelder 1995).

Time of day and protection status were treated as fixed effects, while depth, temperature, rugosity, profile, substrate and visibility were treated as random effects. Time of day was determined by dividing the day into three three-hour sessions, morning (08:00 – 11:00), midday (11:00 – 14:00) and afternoon (14:00 – 17:00), and recorded as the session in which the dive (or most thereof) was conducted (Smith 2005). Protection status referred to protected (TNP) or exploited (PB) areas. Depth for each UVC count was recorded as the depth at the shot line, and temperature as the average recorded by the dive computer every 30 seconds throughout the dive. Rugosity and profile were calculated as the SD and the sum of squares (McCormick 1994), respectively, of the differences between consecutive depth readings, recorded every 10 m along the transect line (Chapter 5). Substrate was calculated as the percentage of depth readings taken over rock, as opposed to sand or rock/sand. Visibility (in metres) was estimated as the horizontal distance along the transect line, from where the shot line anchor first became visible. GLMs for UVC data were only run for the four most abundant species, and all species combined.

For catch data, response codes were again count variables, and the distribution assumed to be Poisson, with a log-link function (McCullagh and Nelder 1995). To model the effect of the variables on catch, factors were included in a GLM of the form:

$$\log(Catch) = \beta_0 + \beta_1(Depth) + \beta_2(TimeOfDay) + \beta_3(Status) + \varepsilon,$$

where the β_i values were the estimated coefficients (McCullagh and Nelder 1995). For the fish length data the response codes were continuous variables. The distribution of the data was therefore assumed to be normal. The commonly used link function for the normal distribution is the identity link (McCullagh and Nelder 1995):

$$\eta = \mu,$$

where η is the linear predictor and μ is the population mean. To model the effects of the measured variables on fish lengths, factors were included in a GLM of the form:

$$Forklength = \beta_0 + \beta_1(Depth) + \beta_2(TimeOfDay) + \beta_3(Status) + \varepsilon,$$

where the β_i values were the estimated coefficients (McCullagh and Nelder 1995).

Time of day and protection status were again treated as fixed effects, and determined in the same way as for the UVC data. Depth was treated as a random effect, and measured using the boat's echo-sounder. Effort was not included as a factor, as equal effort (two angler hours) was exerted at each station. GLMs for the fishing data were only run for the indicator species (roman) and for all species combined.

Power analysis

Power analyses were conducted on the UVC and fishing data from each study area, to determine the sample size required to detect a change in mean count and catch, of magnitude equal to the differences in mean estimates between protected and exploited sites. The required precision level selected was set at 0.8 and the significance criterion at 0.05 (Bausell and Li 2002). Required sample size was calculated for a one-tailed two sample test, using the equation presented in Kapadia *et al*. (2005):

$$n = \frac{2\sigma^2 \left(z_{1-\alpha/2} + z_{1-\beta}\right)^2}{\Delta^2},$$

where n is the number of samples required per group, σ^2 is the variance pooled between the two groups (protected and exploited), $z_{1-\alpha/2}$ is the Z value corresponding to the significance criterion (α) of 0.05 for a one-tailed test (obtained from statistical tables, Rosner 2000) with ($n_1 + n_2 - 2$) degrees of freedom, $z_{1-\beta}$ is the Z value that corresponds to a power of 0.8 (obtained from statistical tables, Rosner 2000), and Δ is the difference between the mean values of the two groups (Kapadia *et al*. 2005).

6.3 Results and discussion

6.3.1 Implementation and assessment of protocol

The sampling protocol proposed in Chapter 5 was successfully implemented in the two study areas. A total of 16 dives and 16 fishing stations were conducted within each study area between December 2006 and January 2007. Sea conditions allowed all sampling to be conducted within six days in each area. The spatial distributions of these samples within each area are illustrated in Figure 6.1.

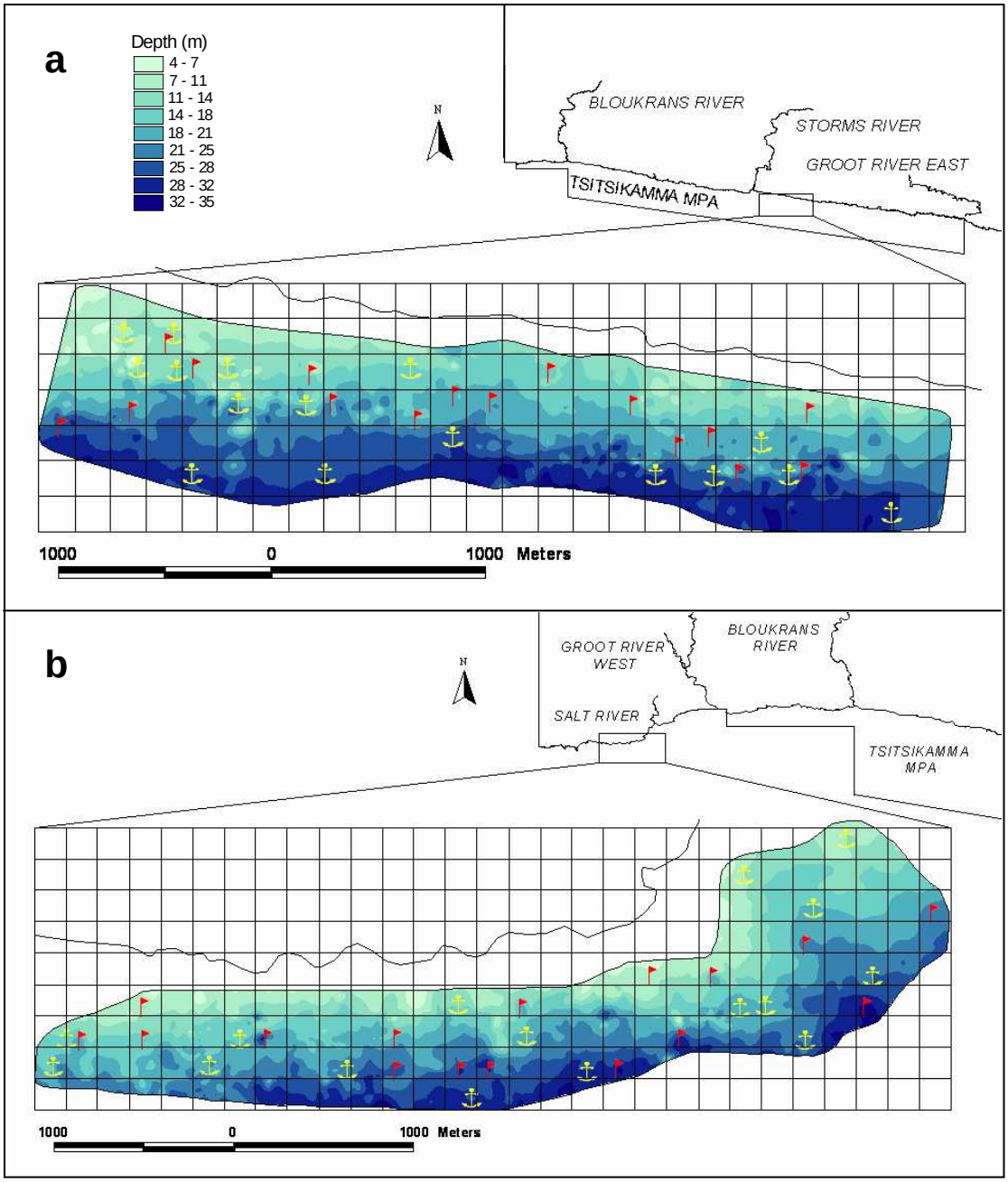

Figure 6.1: Continuous seafloors extrapolated from depth data obtained during mapping, showing the spatial distribution of dive sites (represented by flags) and fishing stations (represented by anchors) in the protected (a) and exploited (b) study areas.

6.3.2 Between areas comparison

Underwater visual census

During the 32 dives, a total of 3 209 individuals were recorded, representing 34 species and 11 families (Table 6.1).

Table 6.1: Mean densities (± SD) of fish species recorded during UVC counts, expressed as fish•100 m^{-2}. Species with a * showed significant differences in density between protected and exploited areas (Mann-Whitney U test).

FAMILY/Species	Common name	Protected (TNP)	Exploited (PB)	p-value
DASYATIDAE				
Dasyatis chrysonata	Blue stingray	0.042 ± 0.114	0.104 ± 0.201	0.522
ARIIDAE				
Galeichthys feliceps	White seacatfish		0.021	
SERRANIDAE				
Acanthistius sebastoides	Koester		0.063 ± 0.181	
Epinephelus marginatus	Yellowbelly rockcod	0.021		
SPARIDAE				
Cymatoceps nasutus	Black musselcracker		0.021	
*Diplodus sargus capensis**	Blacktail	2.354 ± 2.927	7.604 ± 5.747	0.002
Pachymetopon aeneum	Blue hottentot	4.875 ± 5.470	4.938 ± 5.736	0.880
Pachymetopon grande	Bronze bream	0.042 ± 0.114		
Rhabdosargus holubi	Cape stumpnose	0.250 ± 0.375	0.542 ± 0.851	0.611
Argyrozona argyrozona	Carpenter	0.021	0.021	
Chrysoblephus cristiceps	Dageraad	0.229 ± 0.675	0.167 ± 0.211	0.327
Porcostoma dentata	Dane	0.021	0.021	
Boopsoidea inornata	Fransmadam	3.438 ± 2.247	8.188 ± 7.046	0.076
Gymnocrotaphus curvidens	Janbruin	0.417 ± 0.494	0.667 ± 0.951	0.720
Petrus rupestris	Red steenbras	0.146 ± 0.210	0.271 ± 0.408	0.692
*Chrysoblephus gibbiceps**	Red stumpnose	0.021	0.604 ± 0.990	0.005
Chrysoblephus laticeps	Roman	2.292 ± 1.229	1.854 ± 1.355	0.624
Lithognathus mormyrus	Sand steenbras	0.563 ± 2.250		
Cheimerius nufar	Santer		0.333 ± 0.609	
*Spondyliosoma emarginatum**	Steentjie	0.542 ± 1.067	4.396 ± 6.138	0.048
Sarpa salpa	Strepie	4.479 ± 12.273	2.125 ± 8.324	0.955
Rhabdosargus globiceps	White stumpnose	0.250 ± 0.775		
Diplodus cervinus hottentotus	Zebra	0.229 ± 0.338	0.458 ± 0.515	0.274
PARASCORPIDIDAE				
Parascorpis typus	Jutjaw	0.104 ± 0.160	0.083 ± 0.192	0.611
SCIAENIDAE				
Umbrina canarienses	Baardman		0.021	
CHAETODONTIDAE				
Chaetodon dolosus	Blackedged butterflyfish		0.021	
Chaetodon marleyi	Doublesash butterflyfish	0.042 ± 0.114	0.271 ± 0.408	0.175
OPLEGNATHIDAE				
Oplegnathus conwayi	Cape knifejaw	0.333 ± 0.571	0.333 ± 0.558	0.851
CARANGIDAE				
Trachurus trachurus	Maasbanker	0.042 ± 0.167		
CHEILODACTYLIDAE				
Chirodactylus grandis	Bank steenbras		0.021	
Cheilodactylus pixi	Barred fingerfin	2.438 ± 2.362	1.646 ± 1.285	0.720
Cheilodactylus fasciatus	Redfingers	0.167 ± 0.298	0.229 ± 0.675	0.611
Chirodactylus brachydactylus	Twotone fingerfin	4.625 ± 3.218	3.833 ± 4.000	0.309
TRIAKIDAE				
Triakis megalopterus	Spotted gullyshark	0.021		

The family Sparidae dominated, with 56% (n = 19) of all species observed and 78% (n = 2 514) of all individuals observed. Sparids contributed 80% of species and 79% (n = 2 350) of individuals to the 10 most abundant species. *Boopsoidea inornata* (fransmadam) was the most abundant species, with 17% (n = 558) of all individuals observed. Mann-Whitney U tests showed that observed densities of *Diplodus sargus capensis* (blacktail), *Chrysoblephus gibbiceps* (red stumpnose) and *Spondyliosoma emarginatum* (steentjie) were significantly higher in the exploited area, but showed no significant differences in observed densities for any other species. Observed densities (fish•100 m^{-2}) from the protected and exploited sites are displayed in Table 6.1.

GLM analysis showed that fransmadam counts were significantly higher within the exploited area than the protected area (Fig. 6.2a).

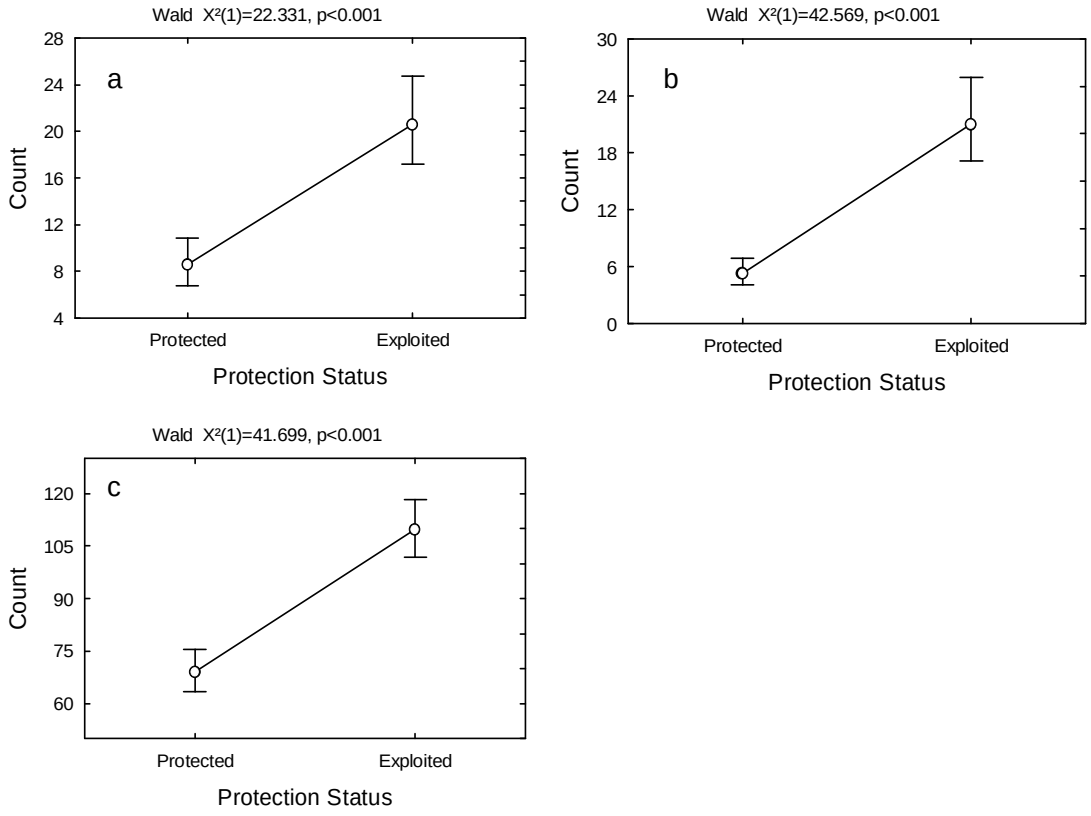

Figure 6.2: Results of GLM analysis of the effect of protection status on mean UVC count (± SD) of fransmadam (a), blacktail (b) and all species combined (c).

This result was consistent with that obtained by Buxton and Smale (1989) in the TNP and the nearby exploited reefs of PB, and by Götz (2005) in the Goukamma MPA and

the adjacent fished reefs, but was contrary to that obtained by Smith (2005) in the TNP and PB. This discrepancy with the results from Smith (2005) may be a result of Smith using point counts, as opposed to transect counts used in the current study and by Buxton and Smale (1989). Counts of blacktail were also significantly higher in the exploited area than the protected area (Fig. 6.2b), a result also found by Smith (2005) in the same area. Counts of all species combined (Fig. 6.2c) were also significantly higher in the exploited area. This is contrary to results obtained by most studies, when comparing protected and exploited areas using UVC (e.g. McCormick and Choat 1987, Polunin and Roberts 1993, Froeschke *et al*. 2006). However, this may be explained by the high proportions of smaller, generalist species in the exploited area, including fransmadam, blacktail, *Pachymetopon aeneum* (blue hottentot), steentjie and *Sarpa salpa* (strepie), where such species may be allowed to proliferate in the absence of larger, predatory species (evidence for which is supported by results from the controlled fishing).

Protection status had no significant effect on roman density (p=0.624 Mann-Whitney U, p=0.124 GLM). This is likely due to similar numbers of roman within each area, but gives no consideration to the mean sizes of individuals. Although not significantly different between protected and exploited areas, the observed density of roman in the current study (2.29 ± 1.23 fish•100 m^{-2}) was comparable to densities observed by Buxton and Smale (1989) (2.30 ± 1.6 fish•100 m^{-2}) and Burger (1990) (2.10 ± 1.03 fish•100 m^{-2}) for the protected area. Similarly, observed density of roman in the exploited area (1.85 ± 1.35 fish•100 m^{-2}) was comparable to that obtained by Burger (1990) (1.64 ± 0.75 fish•100 m^{-2}). Results were, however, substantially lower than estimates of density provided by Smith (2005) for the protected (4.26 ± 0.03 fish•100 m^{-2}) and exploited areas (2.79 ± 0.13 fish•100 m^{-2}). This is likely a result of the attraction effect of diver presence on the abundance of fishes during a stationary point count (Kulbicki 1998), as used by Smith (2005).

Controlled fishing

During the 32 fishing stations, a total of 384 individuals were captured, representing 22 species and 10 families. As with the UVC data, Sparidae was the most speciose and abundant family, with 55% (n = 12) and 87% (n = 333), respectively. Sparids represented the top five most abundant species, with roman contributing an

overwhelming 57% (n = 220) to the overall catch. CPUE (expressed as fish•angler hour^{-1}) from the protected and exploited sites are displayed in Table 6.2.

Table 6.2: Mean CPUE (fish•angler hour^{-1} ± SD) of all fishes captured in the protected and exploited study areas. Species with a * showed significant differences in CPUE between protected and exploited areas (Mann-Whitney U test).

FAMILY/Species	Common name	Protected (TNP)	Exploited (PB)	p-value
CARCHARHINIDAE				
Carcharhinus brachyurus	Copper shark	0.031	0.188 ± 0.403	0.356
TRIAKIDAE				
Mustelus mustelus	Smooth-hound shark	0.469 ± 1.522		
SPHYRNIDAE				
Sphyrna mokarran	Hammerhead	0.031		
TRIGLIDAE				
Chelidonichthys kumu	Bluefin gurnard	0.031		
SERRANIDAE				
Epinephelus lanceolatus	Brindle bass		0.031	
Acanthistius sebastoides	Koester	0.094 ± 0.272		
POMATOMIDAE				
Pomatomus saltatrix	Elf	0.031	0.031	
HAEMULIDAE				
Pomadasys olivaceus	Piggy	0.063 ± 0.171	0.031	0.763
SPARIDAE				
Diplodus sargus capensis	Blacktail	0.031	0.031	
Pachymetopon aeneum	Blue hottentot		0.063 ± 0.171	
Rhabdosargus holubi	Cape stumpnose		0.031	
Argyrozona argyrozona	Carpenter	0.031		
Chrysoblephus cristiceps	Dageraad	0.875 ± 2.070	0.094 ± 0.202	0.638
Boopsoidea inornata	Fransmadam	0.281 ± 0.446	0.531 ± 0.645	0.207
Petrus rupestris	Red steenbras	0.063 ± 0.171	0.031	0.763
Pagellus bellottii natalensis	Red tjor-tjor	0.031		
Chrysoblephus laticeps	Roman	4.219 ± 3.683	2.656 ± 1.805	0.291
Cheimerius nufar	Santer		0.563 ± 0.854	
*Spondyliosoma emarginatum**	Steentjie	0.031 ± 0.125	0.719 ± 1.354	0.013
Sarpa salpa	Strepie		0.125 ± 0.500	
SCIAENIDAE				
Atractoscion aequidens	Geelbek	0.500 ± 1.111		
CARANGIDAE *Trachurus trachurus*	Maasbanker	0.063 ± 0.250		

Mann-Whitney U tests showed that catch of steentjie was significantly higher in the exploited area, but showed no significant differences in catch for any other species. Although not significantly different, fransmadam catch in the exploited area was almost twice that in the protected area. Götz (2005) found that fransmadam CPUE was significantly higher at sites outside of the Goukamma MPA. The mean protected

area CPUE of roman in the current study (4.22 ± 3.68 fish•angler^{-1}•hour^{-1}) was comparable with that obtained by Smith (2005) (4.60 ± 3.64), although the exploited area CPUE in the current study (2.66 ± 1.80), was approximately twice that found by Smith (2005) (1.31 ± 2.56). This is likely a result of sampling in the current study being restricted to summer, while that of Smith (2005) was collected in both summer and winter.

Results of the GLM analyses of protection status on mean catch and mean fork length showed the opposite trend to the UVC results. Mean catch of roman (Fig. 6.3a) and all species combined (Fig. 6.3b) were significantly higher in the protected area.

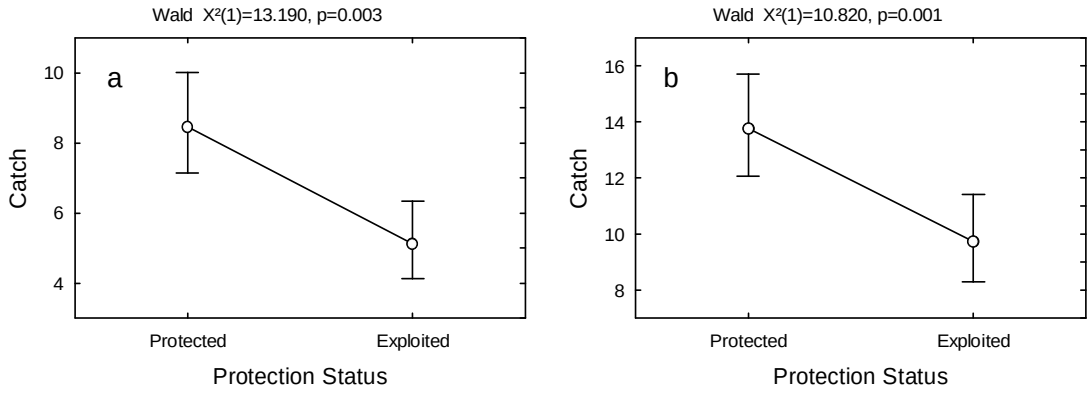

Figure 6.3: Results of GLM analyses of the effect of protection status on mean catch (± SD) of roman (a) and all species combined (b).

Götz (2005) suggested there were two community types represented in the Goukamma study area, one dominated by fransmadam (outside the protection of the MPA), and a second dominated by roman (within the MPA), and that each species was found to dominate in the absence of the other. Furthermore, that the two species share similar diets, including crustaceans, echinoderms, polychaete worms and molluscs (van der Elst 1993, Heemstra and Heemstra 2004), suggests the two species may be in competition, with fransmadam density dependent on that of roman. Fransmadam are also common prey items for larger predatory species, such as *Petrus rupestris* (red steenbras) (van der Elst 1993, Heemstra and Heemstra 2004), shown to be significantly more abundant within the TNP than in PB (Burger 1990), and *Atractoscion aequidens* (geelbek), *Mustelus mustelus* (smooth-hound shark) and *Carcharhinus brachyurus* (copper shark) (van der Elst 1993, Heemstra and Heemstra

2004), which were commonly captured in greater numbers within the protected area than the exploited area (personal observation, this study). The fact that fransmadam density was significantly higher in the exploited area and roman catch was higher in the protected area suggests that there may be one community type dominated by roman in the protected area, and another dominated by fransmadam in the exploited (due to low roman density), similar to that found by Götz (2005) in the Goukamma MPA.

Further evidence for the effect of fishing on community structure in the exploited area was provided by comparison of stock size structure. GLM analysis showed that the fork lengths of roman (Fig. 6.4a) were significantly higher within the protected area than the exploited area. Roman lengths were quite comparable to those obtained during previous studies. Mean roman length (FL) within the protected area during the current study was 331 mm, compared to 313 mm (Smith 2005, TNP) and 302 mm (Götz 2005, Goukamma MPA), while mean length in the exploited area was 283 mm, compared to 263 mm (Smith 2005) and 279 mm (Götz 2005). Fork lengths of all species combined (Fig. 6.4b) were also significantly higher within the protected area.

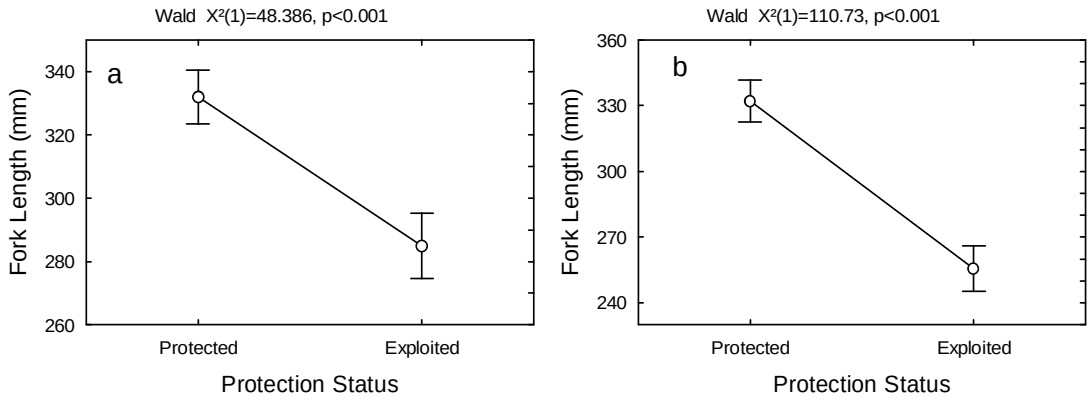

Figure 6.4: Results of GLM analyses of the effect of protection status on mean fork length (± SD) of roman (a) and all species combined (b).

Results of comparison of stock density ratios between protected and exploited areas are presented in Table 6.3. The ratios for roman and all species combined were considerably higher within the protected area than the exploited area.

Table 6.3: Stock density ratios for fork lengths from fishing stations in the protected and exploited areas, for roman and all species combined (numbers in brackets refer to sample size).

	Protected		Exploited	
Roman	0.86	(131)	0.57	(84)
All Species	0.79	(183)	0.41	(157)

T-tests also showed that mean lengths of roman ($p<0.001$) and all species combined ($p<0.001$) were significantly higher in the protected area. Length-frequency distributions further illustrated the greater sizes in the protected area, for roman (Fig. 6.5a) and for all species combined (Fig. 6.5b). Furthermore, average roman age was higher in the protected area (10.4±3.46 years) than the exploited area (8.0± 1.71 years), as was the proportion of older roman (Fig. 6.5c), suggesting that total mortality is considerably higher in the exploited area, as a result of exploitation.

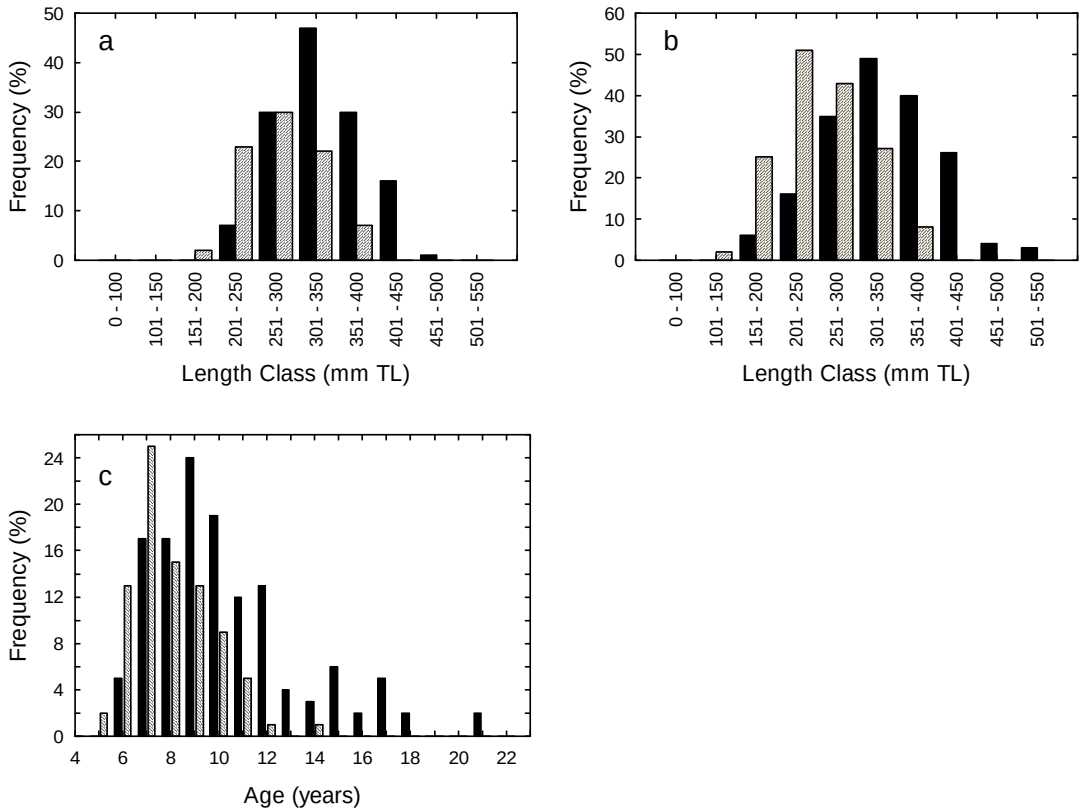

Figure 6.5: Length-frequency distributions of roman (a, n = 215) and all species combined (b, n = 340), and age-frequency distribution of roman (c) captured in the protected (solid bars) and exploited areas (grey bars).

There were no significant differences between the protected and exploited areas, for diversity (Table 6.4) or species richness (Table 6.5) for UVC or fishing data. This highlights the dubious suitability of these measures to detect differences between areas, when differences in community structure have been detected by other methods.

Table 6.4: Results of Mann-Whitney U test comparisons of diversity indices (Margalef's *d*, Pielou's *J* and Shannon's *H'*) between the protected and exploited areas, for UVC and fishing data. Values presented are p-values.

	Margalef's *d*	Pielou's *J*	Shannon's *H'*
UVC	0.147	0.346	0.258
Fishing	0.057	0.227	0.054

Table 6.5: Total number of species observed during all UVC counts and fishing stations, and mean number of species (± SD) per count and fishing station, in the protected and exploited study areas. P-values are from the Mann-Whitney U test.

	TNP	PB	p
UVC Total	27	28	
Fishing Total	17	14	
UVC Mean	10 ± 3.01	11.75 ± 3.64	0.148
Fishing Mean	2.75 ± 1.48	3.56 ± 0.89	0.183

Multivariate analysis

Comparisons between the protected and exploited areas were made using multivariate analyses, to determine the suitability of this method for analysis of long-term data, for the protected and exploited areas. Although UVC sites exhibited similar community structures within each study area, results from the cluster dendogram were not conclusive. Spatial relationships between protected and exploited sites were therefore illustrated by MDS, which separated the protected and exploited dive site communities to some extent (Fig. 6.6). As with the UVC data, the cluster dendogram did not show a clear difference in communities between protected and exploited area fishing stations. Spatial relationships were again illustrated by MDS (Fig. 6.7). Groups of protected and exploited communities were distinguishable.

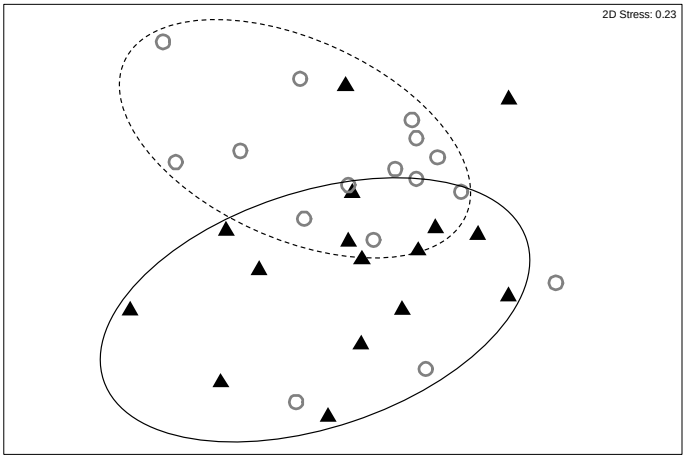

Figure 6.6: MDS plot for UVC data showing the spatial relationships between sites. Triangles (encompassed by the solid ellipse) represent sites in the protected area and circles (encompassed by the dashed ellipse) represent sites in the exploited area.

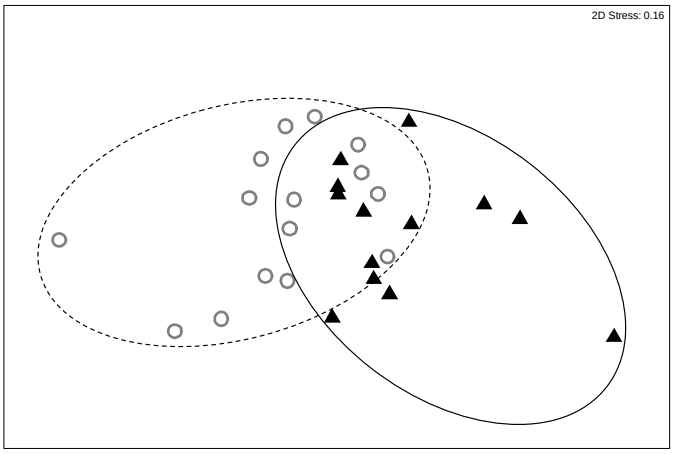

Figure 6.7: MDS plot showing the spatial relationship between the fishing stations. Triangles (encompassed by the solid ellipse) represent sites in the protected area and circles (encompassed by the dashed ellipse) represent sites in the exploited area.

There was a significant difference in the communities observed during UVC, between sites in protected and exploited areas (ANOSIM, $p = 0.01$). Results of the SIMPER analysis on UVC data showed that roman contributed 17.3% to similarity among protected area UVC counts, but only 9.9% among exploited area counts (Table 6.6). Steentjie, *Pachymetopon aeneum* (blue hottentot) and blacktail contributed most to the dissimilarity between protected and exploited groups (Table 6.7). There was also a significant difference between communities captured at sites in the protected and

exploited areas (ANOSIM, $p < 0.01$). Results of the SIMPER analysis on fishing data, showing the contribution of each species to overall similarity between and among sites, are presented in Table 6.8. The average similarity was 33.3% for the protected group and 46.1% for the exploited group. Roman contributed the most to between group similarity for both the protected and exploited sites; however, the percentage contribution was substantially higher within the protected group (84.7%) than the exploited group (54.1%). SIMPER analysis of catch data showed that roman contributed most to the between-group dissimilarity, followed by steentjie, fransmadam, *Cheimerius nufar* (santer) and *Chrysoblephus cristiceps* (dageraad) (Table 6.9).

Table 6.6: Results of SIMPER analysis of UVC data, showing average similarity and percent contribution of each species to overall similarity for protected and exploited groups. Cumulative cut-off to exclude species with low contributions was 90%.

Protected					
Species	Average Count (number per dive)	Average Similarity	Similarity SD	% Contribution	Cumulative% Contribution
Twotone fingerfin	1.83	11.11	4.53	18.71	18.71
Roman	1.58	10.25	4.62	17.25	35.96
Fransmadam	1.65	9.72	2.2	16.36	52.32
Blue hottentot	1.54	6.87	1.26	11.56	63.88
Barred fingerfin	1.37	6.74	1.43	11.34	75.22
Blacktail	1.28	6.01	1.24	10.11	85.33
Janbruin	0.67	1.97	0.64	3.32	88.65
Steentjie	0.62	1.61	0.54	2.7	91.35

Average similarity: 59.41

Exploited					
Species	Average Count (number per dive)	Average Similarity	Similarity SD	% Contribution	Cumulative% Contribution
Blacktail	2.05	10.26	3.91	17.24	17.24
Fransmadam	2.04	9.8	4.09	16.46	33.7
Barred fingerfin	1.42	7.55	3.77	12.68	46.38
Blue hottentot	1.61	6.4	1.57	10.75	57.13
Roman	1.33	5.86	1.59	9.85	66.98
Twotone fingerfin	1.46	5.09	1.25	8.55	75.53
Steentjie	1.29	3.68	0.84	6.18	81.71
Red stumpnose	0.76	2.09	0.74	3.51	85.22
Janbruin	0.78	1.97	0.76	3.3	88.53
Zebra	0.69	1.7	0.63	2.85	91.38

Average similarity: 59.51

Table 6.7: SIMPER results for percent contribution of each species to overall dissimilarity between groups, for the UVC data. Cumulative cut-off to exclude species with low contributions was 90%.

Species	Protected Average Count (number per dive)	Exploited Average Count (number per dive)	Average Dissimilarity	Dissimilarity SD	% Contribution	Cumulative % Contribution
Steentjie	0.62	1.29	3.41	1.25	7.97	7.97
Blue hottentot	1.54	1.61	3	1.1	7.02	14.99
Blacktail	1.28	2.05	2.89	1.28	6.75	21.74
Twotone fingerfin	1.83	1.46	2.6	0.96	6.07	27.81
Red stumpnose	0.06	0.76	2.24	1.16	5.24	33.05
Janbruin	0.67	0.78	2.16	1.14	5.04	38.09
Fransmadam	1.65	2.04	2.07	1.15	4.83	42.92
Zebra	0.48	0.69	2.05	1.06	4.78	47.69
Cape stumpnose	0.49	0.58	2.01	1.06	4.7	52.39
Barred fingerfin	1.37	1.42	1.88	1.07	4.4	56.79
Cape knifejaw	0.46	0.51	1.83	1.01	4.28	61.07
Strepie	0.4	0.27	1.68	0.5	3.93	64.99
Red steenbras	0.39	0.45	1.67	0.96	3.9	68.89
Dageraad	0.24	0.45	1.62	0.92	3.79	72.69
Roman	1.58	1.33	1.5	0.78	3.51	76.19
Doublesash butterflyfish	0.13	0.45	1.43	0.82	3.34	79.54
Redfinger	0.34	0.24	1.41	0.78	3.3	82.84
Santer	0	0.41	1.15	0.66	2.68	85.52
Jutjaw	0.31	0.2	1.14	0.77	2.65	88.18
Blue stingray	0.13	0.26	1	0.64	2.33	90.5

Average dissimilarity = 42.82

Table 6.8: Results of SIMPER analysis of fishing data, showing average similarity and percent contribution to overall similarity of each species for protected and exploited groups. Cumulative cut-off to exclude species with low contributions was 90%.

Protected

Species	Average Catch (fish per station)	Average Similarity	Similarity SD	% Contribution	Cumulative% Contribution
Roman	2.48	28.24	1.11	84.73	84.73
Fransmadam	0.45	2.33	0.37	6.98	91.7

Average similarity: 33.34

Exploited

Species	Average Catch (fish per station)	Average Similarity	Similarity SD	% Contribution	Cumulative% Contribution
Roman	2.07	26.84	1.53	58.13	58.13
Fransmadam	0.78	7.17	0.75	15.53	73.66
Steentjie	0.8	5.95	0.63	12.89	86.55
Santer	0.7	4.75	0.52	10.28	96.83

Average similarity: 46.17

Table 6.9: SIMPER results for percent contribution of each species to overall dissimilarity between groups, for fishing data. Cumulative cut-off to exclude species with low contributions was 90%.

Species	Protected Average Catch (fish per station)	Exploited Average Catch (fish per station)	Average Dissimilarity	Dissimilarity SD	% Contribution	Cumulative% Contribution
Roman	2.48	2.07	15.38	1.15	23.25	23.25
Steentjie	0.06	0.8	7.73	0.88	11.69	34.94
Fransmadam	0.45	0.78	7.11	1.06	10.75	45.69
Santer	0	0.7	6.89	0.82	10.42	56.1
Dageraad	0.61	0.19	5.76	0.67	8.71	64.82
Geelbek	0.47	0	3.72	0.53	5.63	70.44
Smooth-hound	0.32	0	3.53	0.34	5.34	75.79
Copper shark	0.06	0.3	3.19	0.55	4.83	80.62
Piggy	0.13	0.06	1.57	0.45	2.37	82.99
Red steenbras	0.13	0.06	1.53	0.43	2.32	85.31
Koester	0.15	0	1.2	0.37	1.82	87.13
Strepie	0	0.13	1.17	0.25	1.77	88.9
Elf	0.06	0.06	1.11	0.36	1.68	90.57

Average dissimilarity = 66.14

Roman was the seventh most abundant species in UVC counts in both areas, but provided a greater contribution to counts in the protected area (6.9%) than the exploited area (5.8%) (Fig. 6.8).

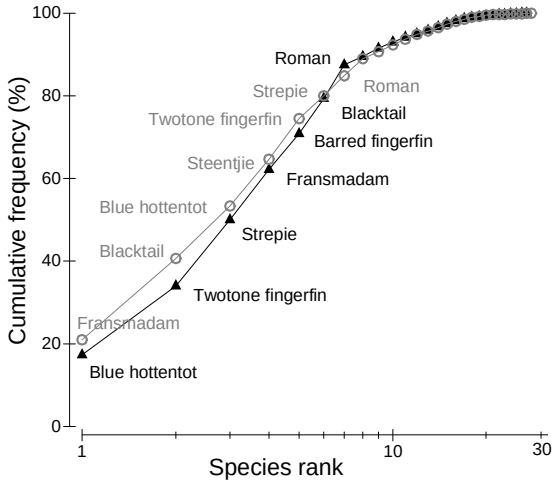

Figure 6.8: Cumulative frequency dominance plot for the seven most dominant species observed in the protected area (triangles) and exploited area (circles) UVC counts.

Roman provided the greatest catch contribution in both study areas, but the contribution in the protected area (8.4%) was considerably higher than in the exploited area (5.3%) (Fig. 6.9).

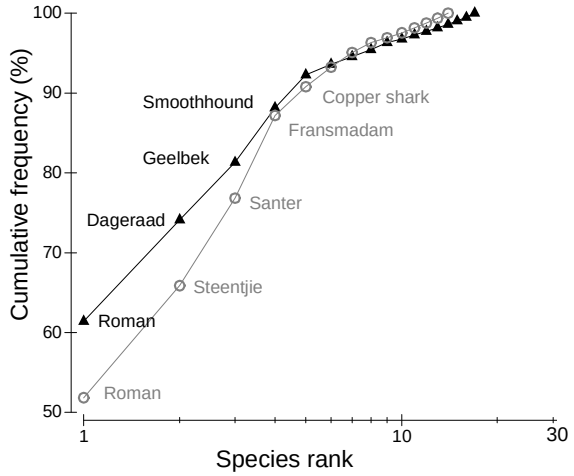

Figure 6.9: Cumulative frequency dominance plot for the four and five most dominant species captured in the protected area (triangles) and the exploited areas (circles), respectively.

6.3.3 Indicators and analyses for protected areas

The above results illustrate the ability of the methods and techniques used in the proposed sampling protocol to detect differences between the protected and exploited study areas, suggesting that the methods would be suitable for detecting the long-term effects of fishing on a reef fish community. The fact that the GLM analyses showed significantly higher counts of fransmadam, blacktail and all species combined, within the exploited areas, and that Mann-Whitney U tests showed significantly higher counts for blacktail, red stumpnose and steentjie, and higher catches of steentjie within the exploited area than the protected area, suggests that the effects of fishing are not limited to simple decreases in density and size-frequency distribution, but include larger, community-level change (Russ 1991). These are adequately reflected using the sampling methodology.

MDS analysis of community structure and ANOSIM showed two distinguishable communities, protected and exploited, for UVC and fishing data. For UVC counts, roman contributed approximately 17% and 10% to within site similarity in the protected and exploited areas, respectively (Table 6.6), but less than 4% to

dissimilarity between exploited and protected areas (Table 6.7). In contrast, roman contributed approximately 87% and 58%, respectively, to within site similarity in fishing data from the protected and exploited areas (Table 6.8), and contributed the most (approximately 23%) to dissimilarity between the protected and exploited areas (Table 6.9). Therefore, MVA was able to detect differences in community structure between areas and is, therefore, suitable for detecting temporal change in community structure within the protected area (that may be associated with natural change) and the exploited area (that may be associated with natural and fishing-associated change) over time. MVA should, therefore, provide good representation of change in community structure from annual samples within each study area, taken over the long term (Cowley and Götz 2007).

6.3.4 Indicators and analyses for exploited areas

Although MVA was able to detect differences (and therefore also change), the methods provided no insight into the status of the communities, or causal relationships between fishing pressure in the exploited area and observed differences in community structure. Therefore, for LTM in the exploited area, for which the protected area would act as a control, it would be better to assess the status of the exploited area community using indices at the species level, based on one or more indicator species, rather than the community level indices such as diversity or species richness, and MVA.

Roman was the most dominant species in the catch in both areas. GLM analysis showed significantly higher catches and lengths of roman and all species combined, in the protected area. Stock density ratios provided results consistent with the GLM analyses. It is, therefore, suggested that roman CPUE, mean length and stock density ratio be used as indicators of ichthyofaunal community health, for LTM of the exploited area, compared with the protected area. Although Mann-Whitney U test comparison and GLM analysis showed no significant difference in observed density between study areas, such a difference was shown by Smith (2005). Small sample size may be the reason that a similar result was not obtained in the current study. It is therefore suggested that observed density of roman also be used as an indicator in the warm temperate biogeographical region, for LTM of the exploited area stocks using the protected area as a control. Additional indicators, such as densities and catch of

fransmadam and blacktail, as well as densities of selected invertebrate prey species, may provide supplementary data on change in community structure as a result of fishing pressure or climate change (Pajak 2000).

6.3.5 Environmental variables

The effects of environmental variables were tested using GLM analysis, to determine which environmental variables should be measured during sampling for LTM. Regression analysis showed that there was a strong correlation between profile and rugosity ($p<0.001$, $r^2=0.962$, Fig. 6.10).

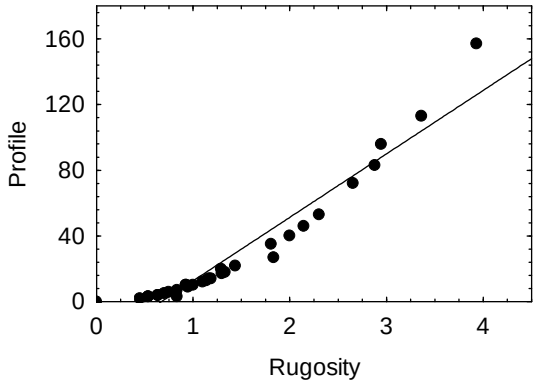

Figure 6.10: Relationship between rugosity and profile ($r^2 = 0.9259$, $p<0.01$).

This was to be expected as profile and rugosity measures, although involving different formulas, were calculated based on the same set of depth recordings. It was assumed that the actual architectural complexity was more likely to be of significance than slope (Hixon and Beets 1989, Kellison and Sedberry 1998), as most species under study are reef associated and use the reef complexity for refuge. This was based on the fact that numerous authors have tested the effects of rugosity on fish abundance or density (Risk 1972, Luckhurst and Luckhurst 1978, Carpenter *et al*. 1981, Grigg 1994, Connell and Kingsford 1998, Friedlander *et al*. 2003, Gratwicke and Speight 2005a), and a significant positive correlation between rugosity and fish density is commonly found (Gratwicke and Speight 2005b). In contrast, little work has been done on the effect of census area slope or gradient on density, and it is the actual refuges, provided by substrate unevenness, that are sought by the fish (Almany 2004). Furthermore, Bell *et al*. (1987) and Jenkins and Wheatley (1998) suggested that the presence of structure may be of greater importance as a predictor of fish abundance than the type of structure or habitat characteristics. As profile measured the overall

'slope' of the census area, while rugosity measured the complexity of the substrate (Chapter 5), rugosity was assumed to be a better predictor than profile, and in order to decrease complexity of the GLM for the diving data, profile was removed as a factor.

However, there is considerable ambiguity in the literature, in terms of definitions of terms relating to habitat complexity. For example, McCormick (1994) used vertical relief, topography, substratum topography and architecture interchangeably, Ohman and Rajasuriya (1998) used structural complexity, surface topography and rugosity interchangeably, and Ferreira *et al*. (2001) and Kuffner *et al*. (2007) used topographic complexity and rugosity interchangeably. Furthermore, many studies have used varying methods to measure habitat complexity, prohibiting comparisons between studies (Gratwicke and Speight 2005a). However, as numerous authors have found significant effects of habitat complexity on fish densities (Luckhurst and Luckhurst 1978, Grigg 1994, Friedlander and Parrish 1998, Friedlander *et al*. 2003, Almany 2004), such a measure should still be included as a predictor variable in a LTM programme, in addition to rugosity (calculated according to the method described in Chapter 5). Polunin and Roberts (1993) and Gratwicke and Speight (2005b) suggested subjective estimation of habitat complexity, as a categorical variable. Mann *et al*. (2006) classed "profile" subjectively as high or low. Smith (2005) initially proposed three profile categories, low, medium and high, but reduced these to two categories before GLM analysis, by combining medium and high categories. It is, therefore, suggested that profile be estimated subjectively during each dive as low or high, according to the criteria defined by Smith (2005). As with rugosity, profile cannot not be measured during fishing stations, or included in GLM analyses for fishing data for LTM.

Results of GLM analyses run on the main and between effects for each species are summarised in Appendix IV. These analyses identified a number of significant between effects (i.e. effects between parameters). There are two possible explanations for the occurrence of these between effects: 1) there are true interactions between variables, highlighting the need to stratify sampling and include these variables in the GLM analyses, or 2) the between effects are not real, but rather a result of the small sample sizes and the high number of variables included. Results of the GLM analyses of main effects on count, catch and length data are summarised in Table 6.10.

Table 6.10: Results of GLMs run on count, catch and length data. Values are p-values. Downward arrows represent negative correlations and upward arrows positive correlations. Factors excluded for a species during the preceding AIC best subsets analysis are indicated by X, (prot = protected, expl = exploited, ns = not significant).

	Time of day	Protection status	Depth	Temperature	Visibility	Rugosity	Substrate
UVC							
All Species	<0.001 low midday	<0.001 high expl.	<0.001 ↓	0.006 ↑	<0.001 ↑	<0.001 ↑	<0.001 ↑
Fransmadam	0.030 low midday	<0.001 high expl.	<0.001 ↓	0.113 ns	x	<0.001 ↑	<0.001 ↑
Blacktail	<0.001 high afternoon	<0.001 high expl.	<0.001 ↓	<0.001 ↑	0.001 ↑	<0.001 ↑	0.100 ns
Twotone Fingerfin	x	x	<0.001 ↓	0.011 ↓	0.016 ↓	<0.001 ↑	<0.001 ↑
Roman	0.092 ns	0.124 ns	0.013 ↑	x	x	x	x
Catch							
All Species	0.034 high midday	0.001 high prot.	x				
Roman	x	<0.001 high prot.	0.004 ↓				
Lengths							
All Species	x	<0.001 high prot.	0.047 ↑				
Roman	<0.001 high afternoon	<0.001 high prot.	x				

Time of day

For all species combined (Fig. 6.11a) and for fransmadam (Fig. 6.11b), UVC counts were significantly lower during midday dives than during morning or afternoon dives. A similar trend was found with blacktail, but for this species counts were significantly

higher during afternoon dives than morning or midday dives (Fig. 6.11c). Time of day was excluded as a factor during the AIC best subsets analysis for twotone fingerfin, and had no significant effect on counts of roman.

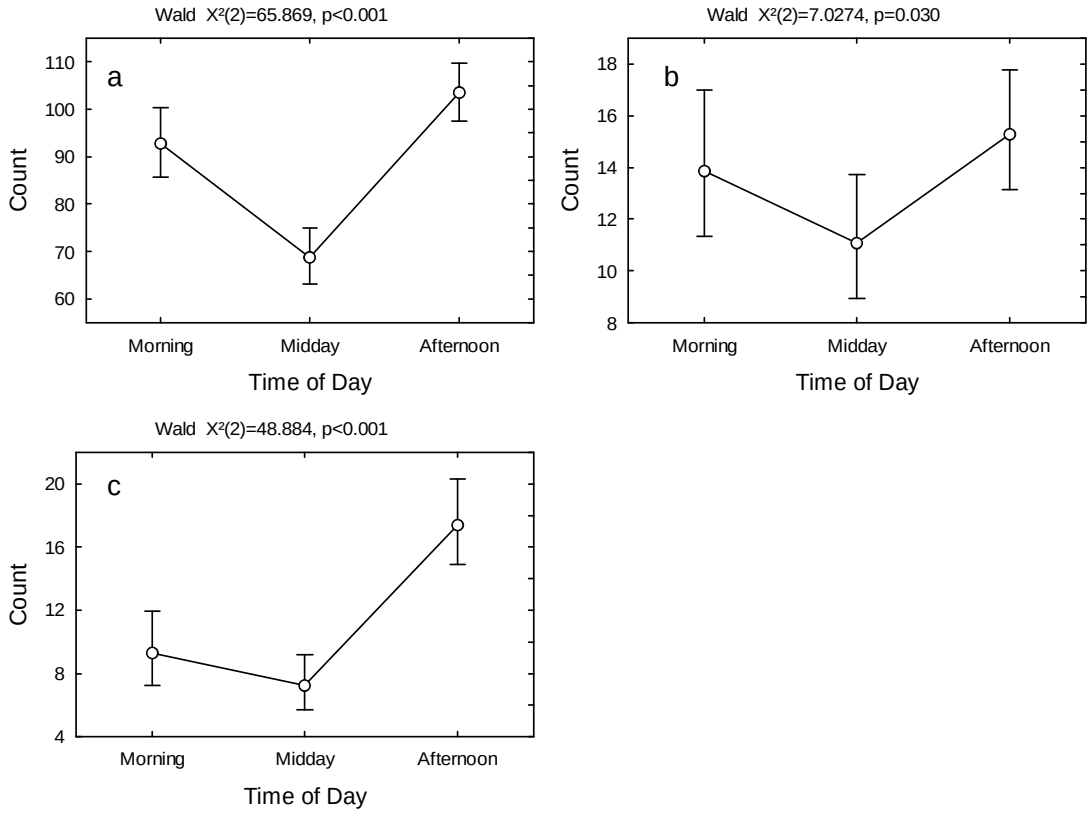

Figure 6.11: Results of GLM analysis of time of day on mean UVC count (± SD) of all species combined (a), fransmadam (b) and blacktail (c).

Time of day also had a significant effect on the catch of all species combined. However, this followed the opposite trend to that of the UVC counts, with a peak in catch at midday fishing stations (Fig. 6.12a). There was also a significant effect of time of day on mean fork length of roman captured (Fig. 6.12b), with significantly larger fish being caught in the afternoon. Time of day was excluded as a factor in the AIC analyses for catch of roman and length of all species. These results suggest that to provide greatest UVC count and controlled fishing catch, sampling dives should be concentrated during morning and afternoon sessions, and that fishing be conducted in the midday session. This study was only conducted over a two-month period. It may, therefore, be necessary to further test the effect of time of day over a longer period.

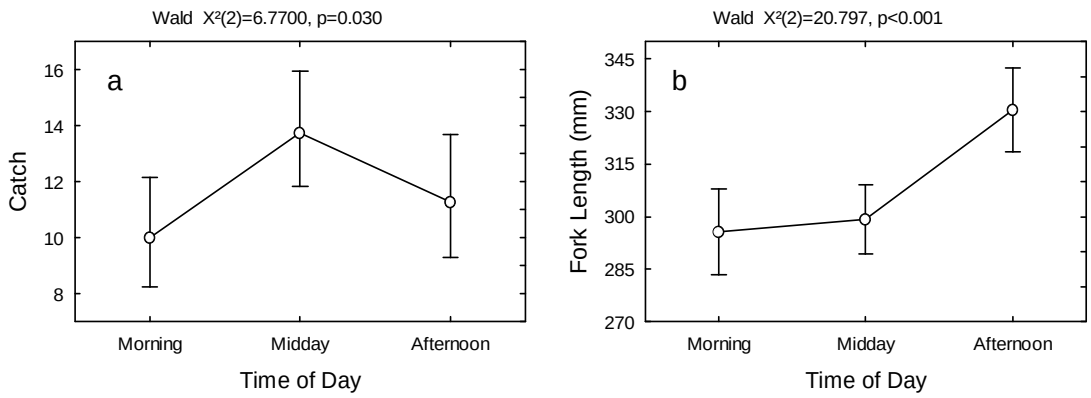

Figure 6.12: Results of GLM analysis of time of day on mean catch (± SD) of all species (a) and mean fork length (± SD) of roman (b).

Depth

GLM analyses showed that mean UVC counts of fransmadam, blacktail, twotone fingerfin and all species combined were negatively correlated with depth. Götz (2005) also found densities of blacktail and twotone fingerfin to be negatively correlated with depth, in the Goukamma MPA. Depth was the only factor that had a significant effect on roman density; however, contrary to the other species investigated, roman density was positively correlated with depth. This result is consistent with Smith (2005), who observed significantly higher densities of roman on deeper UVC counts in the TNP. Roman catch, however, was negatively correlated with depth. This is consistent with results obtained by Götz (2005) in the Goukamma MPA. As hook size selected for the larger individuals, the observed results may be a result of competitive exclusion of the smaller roman individuals by the larger, from shallower, more productive areas that may support greater densities of invertebrate prey species. However, Buxton (1987) and Heemstra and Heemstra (2004) suggested that juvenile roman are more common in shallower reefs, with adults more abundant in the deeper reefs. The discrepancy in findings suggests the need for further assessment of the effects of depth on the abundance of roman, and other species.

Sea temperature

Counts of all species combined, and of blacktail, were positively correlated with bottom temperature. Götz (2005) also found a positive correlation between blacktail density and temperature in the Goukamma MPA. Counts of twotone fingerfin were

negatively correlated with temperature. This may be a result of predator avoidance behaviour or competition in conditions favouring greater densities of most other species. Temperature had no effect on mean count of fransmadam and was excluded in the AIC analysis for roman. Temperature was not recorded during fishing stations, as the PB study area is approximately 35 km from the nearest thermoscript, and as it was likely that surface temperatures were different to bottom temperatures, it was decided that temperature measured at the surface may provide spurious results. However, as it was determined that temperature had a significant effect on the counts of certain species, it is suggested that temperature also be measured during fishing stations in a LTM programme.

Visibility

Blacktail counts were positively correlated with visibility. This is consistent with results obtained by Götz (2005) in the Goukamma MPA. This is likely due to their colouration, making them more difficult to detect, particularly at the sand/reef interface, where suspended sediment may adversely affect visibility. Counts of all species combined were also positively correlated with visibility. Counts of twotone fingerfin were negatively correlated with visibility. This may be a result of predator avoidance behaviour under increased visibility conditions. Visibility was excluded during the AIC analyses of fransmadam and roman. As with temperature, visibility could not be recorded remotely during fishing, and there was no reason to assume that visibility on the surface reflected that near the substrate. However, as the GLM analyses showed significant effects of visibility on counts of certain species, it is suggested for LTM that visibility, or turbidity, be measured during fishing stations.

Rugosity

Mean counts of all species combined, fransmadam and blacktail were positively correlated with rugosity. Smith (2005) also found counts of fransmadam and blacktail to be positively correlated with rugosity, in the TNP. Twotone fingerfin counts were also positively correlated with rugosity, which is in agreement with results from Götz (2005), in the Goukamma MPA. Rugosity was excluded in the AIC analysis of roman. However, Smith (2005) found a positive correlation between roman count and rugosity in the TNP. Rugosity should, therefore, be measured in a LTM programme. Rugosity cannot be measured during fishing stations.

Substrate

Mean count of all species combined, fransmadam and twotone fingerfin were positively correlated with substrate (i.e. % rock cover). Substrate, however, had no effect on the mean count of blacktail and was excluded during the AIC analysis for roman. It is well documented in the literature that higher densities (reef species in general) are recorded over hard substrate than nearby sand or sand/rock substrates (e.g. Anderson *et al*. 1981, Guidetti 2000, Gratwicke and Speight 2005a). Substrate, as measured in this study as the percentage of recordings representing rock, should therefore be included in the GLM analyses for LTM. However, as with rugosity and profile, substrate cannot be measured remotely.

6.3.6 Stratification

The two study areas are spatially distinct and differ in fishing intensity. Furthermore, the protected area would act as a control, against which the exploited area would be monitored, to detect changes in fishing intensity within the exploited area over the long term. It is, therefore, necessary that samples be stratified over protection status.

GLM analyses showed that depth had a significant effect on observed numbers and catch of certain species, and that depth should be included in the GLM analysis. Samples were stratified over depth, which was calculated in ArcView for each sample site, as shallow (10 – 20 m) or deep (20 – 30 m). These depths were consistent with depths measured during each of the 16 UVC and 16 fishing stations at both sites (total n = 64). It is therefore suggested that sampling should be stratified over depth, as determined in ArcView from the mapping data.

Conversely, profile (calculated for each dive using the method explained in Chapter 5) at numerous sites did not resemble the categories of high and low profile (or slope) in each grid cell, as determined using the slope function in ArcView. Therefore, in the absence of alternative measures for remotely estimating profile, it is recommended that samples should not be stratified over profile, as calculated in ArcView.

6.3.7 Power analysis

Power analyses run to calculate the UVC and controlled fishing sample sizes required to detect changes in the mean (equal to the differences between protected and

exploited means), showed that for UVC data, 54 samples were required per study area, while for CPUE fishing, 33 samples were required per study area. The sampling protocol designed in Chapter 5 proposed a sample size of 16 per study area per season, i.e. 32 samples per study area per year. Therefore, for controlled fishing, the proposed sample size of 32 would be sufficient. However, for UVC, the power analysis indicated a required sample size of 54, suggesting that the proposed sample size of 16 should be increased to 27 per study area per season. The higher sample size is required to overcome the higher variability in the UVC count data (CV = 0.66), compared with that from controlled fishing (CV = 0.46). This higher variability associated with UVC counts is likely a result of the diver encountering large schools of fish, such as strepie (which exceeded 100 individuals on two occasions during this study), on some UVC counts and not others. This is unlikely in catch data, as most of these shoaling fishes are probably too small for capture by the hook size used.

6.4 Conclusions

The protocol designed in Chapter 5 is suitable for LTM in the protected and exploited study areas. There were no logistical or financial problems with its implementation.

The sample size of 16 per study area per season is sufficient for CPUE fishing but insufficient for UVC. An increase in sample size to 27 UVC counts per study area per season would, therefore, be necessary for full-scale implementation of the protocol in this area.

GLM analyses showed that it was necessary and desirable to measure and include in the analyses, time of day, protection status, depth (as a continuous variable recorded during each sample), sea temperature, visibility, rugosity (as a continuous variable, calculated as described in Chapter 5), profile (as a categorical variable, subjectively estimated as high or low by the diver during each UVC count) and substrate (as described in Chapter 5). The effects of depth and rugosity on abundance further highlight the importance of ensuring that habitat characteristics, in study areas to be compared, are in fact comparable.

It was also determined that samples should be stratified over depth, but that stratification over profile is not recommended. As there were two distinct study areas, protected and exploited, where the protected area would act as a control against which the exploited area would be compared, it was also necessary to stratify over protection status.

The methods used were able to detect differences between protected and exploited study areas with sufficient statistical robustness. It was shown that community-level indicators were suitable for LTM within the protected area for detection of community change that may result from environmental change, but that species-level indicators (roman mean density, mean CPUE, mean length and stock density ratio) should be used for LTM in the exploited area, to detect potential changes (decreases or increases) in fishing pressure, using the protected area as a control.

Chapter 7
General Discussion

An ecosystem-based approach to fisheries management has been adopted in numerous fishing states, including South Africa, in an attempt to better manage fish stocks, and understand the complexities of ecosystems (Shannon *et al*. 2006). This has brought about the need for identification and establishment of long-term ecological monitoring sites (van Jaarsveld and Biggs 2000) to better understand the effects of fishing on fish communities. However, there is no standardised sampling protocol or methodology available for such assessments of offshore linefish stocks in South Africa. The multifaceted and multispecies nature of the linefishery makes stock assessment procedures and management difficult, and the 'missing baseline' problem makes identification of pristine stock levels difficult. The nature of the temperate environment complicates the collection of ecological data, which is exacerbated by variability in abundance and recruitment of fish stocks. Furthermore, previous studies focusing on assessment of fish stocks have incurred numerous problems. Low statistical power (as a result of insufficient sampling), pseudoreplication and spatial autocorrelation and have resulted in bias in results. The use of inconsistent or non-standardised methods has limited comparability among results from different areas and studies, while the use of a single method prevents verification of results. True variability in population abundance has been masked by temporary or localised shifts in abundance, variable mobility and sightability of different species and sampling error. This study addressed these problems by developing, implementing and evaluating a protocol for monitoring temperate reef fish communities.

Figure 7.1 illustrates the framework followed in the development of the recommended sampling protocol, and which can also be adapted for LTM protocols elsewhere.

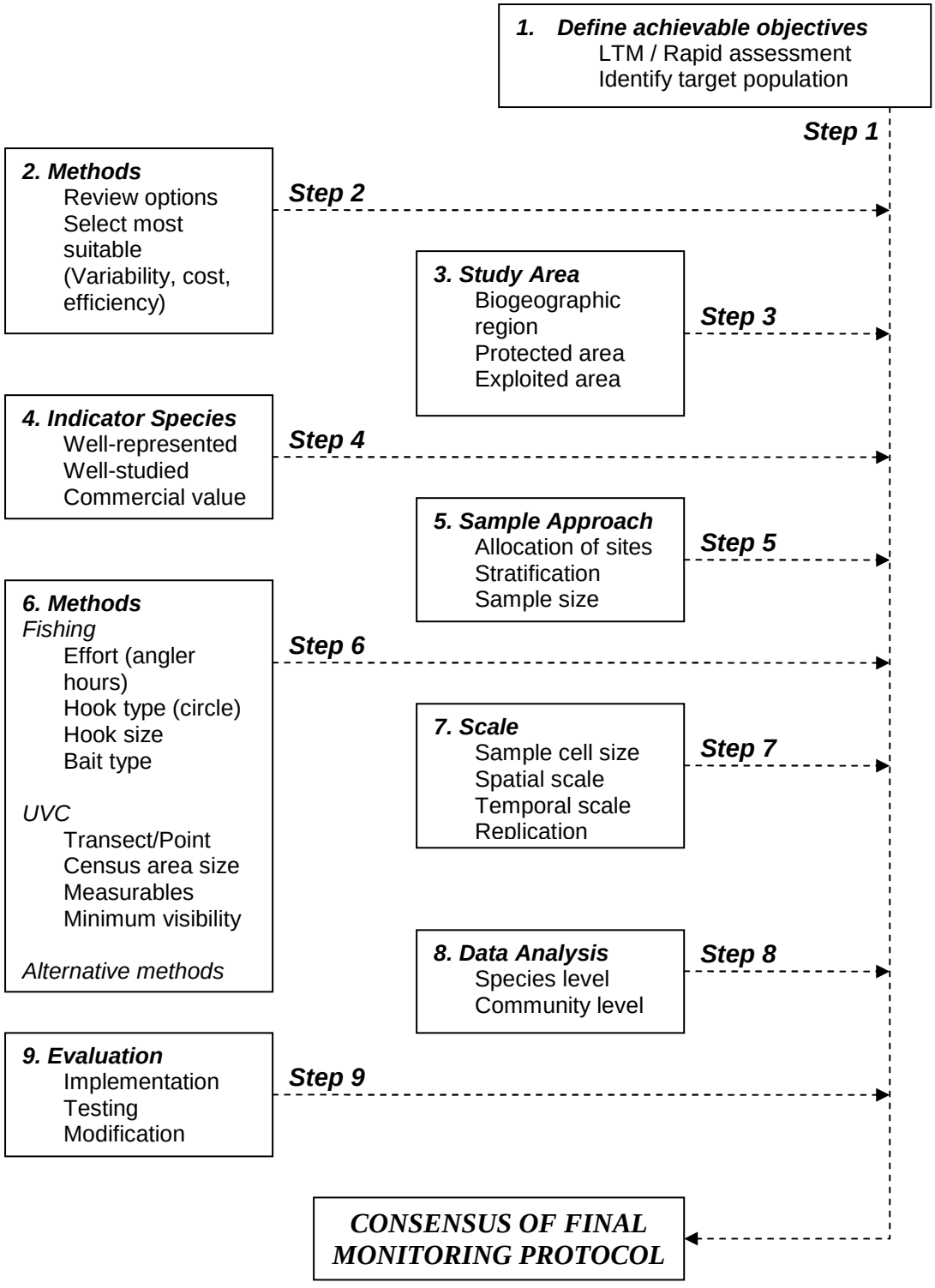

Figure 7.1: Flow diagram of the recommended steps involved in designing a sampling protocol for offshore reef fish.

The first step is to clearly define the objectives of the monitoring programme (or assessment) and identify the target population(s) or community(ies). This may include

LTM or a rapid assessment, and may or may not include comparisons across MPA boundaries. The objectives must be realistically achievable. The second step is to review available methods and broadly identify the sampling methods most suitable for monitoring the target communities and meeting the objectives of the programme. UVC and controlled fishing were identified as suitable methods in this study, and appear to be most suitable for monitoring studies in the warm temperate biogeographical region. Assessment of alternative or additional methods may be necessary or desirable in other areas.

The third step is to identify an area suitable for the objectives of the LTM programme. LTM programmes may wish to target the centre of a biogeographical region for representivity of species (Turpie *et al*. 2000), or a transition zone between biogeographical regions to detect shifts in species' distributions. Where monitoring aims to separate natural from anthropogenic change, and detect the effects of fishing, it is necessary to sample in protected and exploited areas. The fourth step is to identify suitable indicator species. Suitable species should be well-represented in the chosen study area(s), well-studied and of commercial or recreational value (Keough and Quinn 1991).

The fifth step involves the determination of the sampling approach. This includes sample site allocation (revisiting or random reallocation) and determination of required sample size (power analysis). Furthermore, GLM analysis of results from preliminary sampling may be necessary to determine whether the habitat is such that stratification is necessary. This study recommends random reallocation of sites (to allow for more representative sampling) and that a site should not be sampled by different methods in a single season or by the same method in consecutive seasons.

The sixth step includes refining of the chosen methods. Two angler hours was deemed most suitable for controlled fishing stations in this study area, but this may need to be determined for each study area. Similarly, the optimal hook size and bait type may need to be determined for each area. Due to convincing scientific evidence the use of circle hooks is recommended. However, the suitability of circle hooks may need to be determined for the selected indicator species (Cooke and Suski 2004). Optimal hook size may also need to be determined for each species. Hooks should be barbless and

have no offset point (Prince *et al*. 2002). Strip transects were found to be superior to point counts in this study, particularly in the temperate conditions of the general study region. This may, however, need to be determined for other areas, particularly where predominant visibility is greater than in the warm temperate biogeographical region. It is also necessary to determine a suitable census area size, as this has been shown to affect estimates of abundance, species richness and variability (e.g. Cheal and Thompson 1997).

Step seven includes determining the scale at which sampling is to be conducted. This includes the spatial and temporal extents of sampling, and the frequency with which sampling is to be conducted. Furthermore, it is necessary to determine whether the scale of the proposed sampling and the available resources allow replication of treatments.

Step eight should include determining what data analyses will be used to interpret the data collected. It is recommended that for monitoring of an exploited area, aimed at detecting changes relative to a protected area, the programme should focus on a suite of species-level indicators, including mean density, mean CPUE, mean lengths and length-frequency distributions. Monitoring of reef fish stocks in a protected area to detect change in the community over time, which may be associated with climate change or environmental change, should focus on community-level analyses, such as MDS and ANOSIM, and community-level indicators, such as diversity, species richness and species relative abundance. It is recommended, for UVC samples, that time of day and profile be recorded as categorical factors, and that visibility, sea temperature, rugosity and substrate be recorded as continuous factors for GLM analyses of the effects of environmental factors on abundance. Furthermore, it is recommended that samples should be stratified over protection status and depth (as categorical factors), but it is not recommended to stratify over profile. For CPUE samples, time of day and sea temperature should be recorded as with UVC, and if possible, visibility or turbidity should also be measured. However, it is not possible to remotely measure or estimate profile, rugosity or substrate, and these cannot, therefore, be recorded for inclusion in the GLM analyses of CPUE data. CPUE samples should also be stratified over depth and protection status.

The final step (step nine) is to assemble the components determined in the previous steps into a proposed protocol and implement the protocol in the chosen study areas. This preliminary sampling will allow flaws in the sampling design to be detected, provide data for power analysis to determine required sample size, and determine whether stratification is necessary. The sampling protocol proposed in this study was tested in the field and found to be suitable to meet its three objectives: (i) LTM of a protected site to detect natural change (associated with environmental and climate change), (ii) LTM of an exploited site to detect natural and fishing-associated change, and (iii) comparison of protected and exploited sites (to distinguish natural from fishing-associated change).

The protocol designed in this study is highly relevant in South Africa, at a time when LTM is becoming the focus of much ecological, and particularly marine and coastal, research (Shannon *et al*. 2006). Although the protocol was designed to provide a tool to be used in LTM programmes aimed at assessing the effects of fishing on fish stocks and for detection of natural change, it has numerous additional applications (Fig. 7.2).

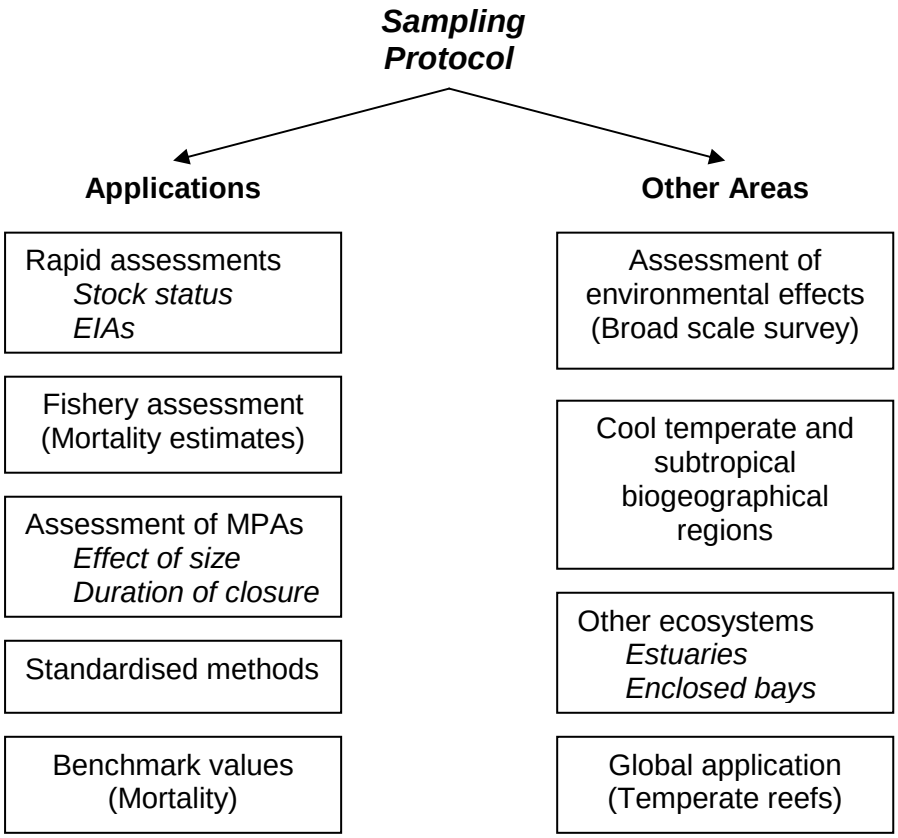

Figure 7.2: Applications of the protocol and the framework developed in this study.

The sampling protocol is suitable for rapid assessments of the status of fish stocks, and can be used to determine whether legislative protection is required, or for environmental impact assessments (EIAs). Cowley *et al*. (2002) discussed the importance of elucidating the role of the different MPAs in South Africa in the management of linefish. The protocol is suitable for assessing the effectiveness of new or well-established MPAs, through 'before-after' or 'inside-outside' comparisons, respectively. The standardised methodology will allow for comparison of results from different areas and studies, particularly of study areas from a network of LTM nodes or LTER sites. The use of a standardised protocol can provide a series of results and bench mark values (such as CPUE of an indicator species) from different areas, to assess the regional status of individual species. The use of the standardised methods on a long-term basis can provide age-frequency distributions and mortality estimates for indicator species at each site, allowing for comparison between areas, and assessment of the effectiveness of management measures. The standardised protocol can also be used in different sized MPAs to evaluate their effectiveness.

Measurement of a suite of environmental variables, using consistent techniques, from a wide range of areas will provide strong evidence of the relationships (or lack thereof) between environmental and biological variables. Such an analysis may, itself, be used for a broad-scale assessment of relationships between fish stocks and environmental variables.

Although developed and tested in the warm temperate biogeographical region of South Africa, the framework may have application in other biogeographical regions. Adaptation of necessary components (e.g. indicator species, hook size and minimum visibility for UVC) will also allow the use of the protocol in the cool temperate and subtropical biogeographical regions. Similarly, adaptations may allow for the protocol to be used for monitoring programmes or assessments in other ecosystem types, such as estuaries or enclosed bays. The framework may also have application in temperate and other reef areas, in other parts of the world.

Long-term biotic change is likely to manifest itself in two ways: (i) reduction in abundance or biomass across the range of a biogeographical region, detectable more

easily in the centre of the region, or (ii) contraction or expansion of a species' distributional range, detectable more easily at the transition between biogeographical regions. The need to monitor these changes has been realised in South Africa, by the establishment of the South African Environmental Observation Network (SAEON). The Elwandle (coastal) Node of SAEON has been mandated to undertake LTM in South Africa's coastal zone, through the establishment of a network of LTM sites. This protocol is suitable for use in such monitoring. However, studies on reef ecosystems have commonly focused narrowly on either the ichthyofaunal, coral or invertebrate communities (Hodgson 1999). Therefore, it is recommended that, for a holistic view of the reef ecosystem and a fully effective monitoring programme, monitoring extend beyond ichthyofaunal assessments, to include marine mammals and invertebrates, in the intertidal and subtidal zones.

Furthermore, underwater video cameras have been successfully implemented in a number of studies (*inter alia* Potts *et al*. 1987, Langton and Uzmann 1989, Parker *et al*. 1994, Adams *et al*. 1995, Willis *et al*. 2000, Parsons *et al*. 2004), and may provide a useful complementary method for the assessment of reef fish assemblages, or for verification of results obtained from UVC or CPUE fishing. However, such methods would require further in-field assessment, and future studies should aim to develop the optimal methodology for this technique.

Fishing pressure has had serious detrimental effects on linefish species in South Africa, and changes in the management regime are necessary to efficiently manage and protect the stocks. LTM has been identified as a means for determining the effects of fishing on fish stocks and to assess the effectiveness of current management, as well as changes therein.

It is important to understand that LTER and monitoring, even with the optimal sampling protocol in place, cannot decrease fishing pressure, or act as a tool for the recovery of fish stocks, and that management measures, such as minimum size and daily bag limits, should remain in place, and where necessary, complementary management and conservation measures should be developed and implemented. Furthermore, it is important that (in addition to LTER and monitoring) effective mechanisms are implemented and actions initiated for the establishment of integrated

coastal management plans, in which resource use areas and regulations are clearly defined and through which resources can be managed (McKenna and Allen 2005).

The sampling protocol designed in this study is suitable for rapid and long-term assessments of reef fish communities, and assessments of management measures, in South Africa and elsewhere, and can assist with biological assessments of coastal and near-shore marine biodiversity, which are essential for the protection and preservation of biodiversity and the persistence of the resources on which many South Africans depend.

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APPENDIX I

Table A1: Species observed during point counts in the TNP and PB, for preliminary diving for comparison of point and transect counts.

Family	Species	Common Name	TNP	PB
HAEMULIDAE	*Pomadasys olivaceus*	Piggy		2
SPARIDAE	*Boopsoidea inornata*	Fransmadam	828	395
	Chrysoblephus cristiceps	Dageraad		6
	Chrysoblephus gibbiceps	Red stumpnose	1	26
	Chrysoblephus laticeps	Roman	341	86
	Cymatoceps nasutus	Black musselcracker		1
	Cheimerius nufar	Santer		7
	Diplodus cervinus hottentotus	Zebra	15	26
	Diplodus sargus capensis	Blacktail	31	61
	Gymnocrotaphus curvidens	Janbruin	19	17
	Pachymetopon aeneum	Blue hottentot	947	69
	Pagellus bellottii natalensis	Red tjor-tjor	1	35
	Pachymetopon grande	Bronze bream	7	12
	Petrus rupestris	Red steenbras	16	21
	Rhabdosargus globiceps	White stumpnose	27	
	Rhabdosargus holubi	Cape stumpnose	27	58
	Sparodon durbanensis	White musselcracker	1	6
	Spondyliosoma emarginatum	Steentjie	169	192
OPLEGNATHIDAE	*Oplegnathus conwayi*	Cape knifejaw	21	19
CHEILODACTYLIDAE	*Chirodactylus brachydactylus*	Twotone fingerfin	211	28

Table A2: Species observed during transect counts in the TNP and PB.

Family	Species	Common Name	TNP	PB
SPARIDAE	*Boopsoidea inornata*	Fransmadam	1367	487
CHEILODACTYLIDAE	*Chirodactylus brachydactylus*	Twotone fingerfin	934	192
	Chrysoblephus cristiceps	Dageraad		9
	Chrysoblephus gibbiceps	Red stumpnose	4	37
	Chrysoblephus laticeps	Roman	670	140
	Cymatoceps nasutus	Black musselcracker		1
	Cheimerius nufar	Santer		14
	Diplodus cervinus hottentotus	Zebra	22	33
	Diplodus sargus capensis	Blacktail	72	162
	Gymnocrotaphus curvidens	Janbruin	68	45
LUTJANIDAE	*Lutjanus argentimaculatus*	River snapper	2	
	Lithognathus mormyrus	Sand steenbras	28	1
OPLEGNATHIDAE	*Oplegnathus conwayi*	Cape knifejaw	60	39
	Pachymetopon aeneum	Blue hottentot	2222	258
	Pagellus bellottii natalensis	Red tjor-tjor	71	12
	Pachymetopon grande	Bronze bream	8	12
	Pomadasys olivaceus	Piggy		1
	Petrus rupestris	Red steenbras	26	26
	Rhabdosargus globiceps	White stumpnose	18	
	Rhabdosargus holubi	Cape stumpnose	75	53
	Sparodon durbanensis	White musselcracker		16
	Spondyliosoma emarginatum	Steentjie	355	370
CARANGIDAE	*Trachurus trachurus*	Maasbanker	2	
SCIAENIDAE	*Umbrina canariensis*	Baardman		1

APPENDIX II

Table A3: Species captured during preliminary sampling in the TNP and PB, to determine optimal fishing station effort.

Family	Species	Common Name	TNP	PB
ARIIDAE	*Galeichthys feliceps*	White seacatfish	2	
SERRANIDAE	*Acanthistius sebastoides*	Koester	1	
POMATOMIDAE	*Pomatomus saltatrix*	Elf	16	
HAEMULIDAE	*Pomadasys olivaceus*	Piggy	5	1
SPARIDAE	*Boopsoidea inornata*	Fransmadam	79	12
	Chrysoblephus cristiceps	Dageraad	1	
	Chrysoblephus gibbiceps	Red stumpnose	3	
	Chrysoblephus laticeps	Roman	377	79
	Cymatoceps nasutus	Black musselcracker		3
	Cheimerius nufar	Santer	1	2
	Diplodus sargus capensis	Blacktail	4	5
	Pachymetopon aeneum	Blue hottentot	7	14
	Pagellus bellottii natalensis	Red tjor-tjor	26	
	Pachymetopon grande	Bronze bream		3
	Petrus rupestris	Red steenbras	9	9
	Polysteganus undulosis	Seventy-four		1
	Rhabdosargus globiceps	White stumpnose	2	
	Spondyliosoma emarginatum	Steentjie	41	72
SCIAENIDAE	*Atractoscion aequidens*	Geelbek	4	
	Argyrosomus inodorus	Silver kob	2	
CARANGIDAE	*Trachurus trachurus*	Maasbanker	3	

APPENDIX III

GOVERNMENT GAZETTE, 11 JANUARY 2002 No. 22991

GOVERNMENT NOTICES

DEPARTMENT OF LABOUR

OCCUPATIONAL HEALTH AND SAFETY ACT, 1993 (ACT NO. 85 OF 1993)

DIVING REGULATIONS, 2001

ANNEXURE D
MINIMUM PERSONNEL REQUIREMENTS

SCUBA AIR

1 × Diver
1 × Line Attendant
1 × Standby Diver
1 × Diving Supervisor

APPENDIX IV

Results of the GLM analyses run for main effects and between effects (separately) on count, catch and length data are presented in Tables A4 to A12.

Table A4: Results of the GLM analysis for UVC counts of all species combined. No initial parameters were discarded during the preceding AIC best subsets analysis. Significance level is denoted by * ($p<0.05$), ** ($p<0.01$) or ns (not significant) (df = degrees of freedom).

Main Effects	df	Wald (X^2)	p	
Intercept	1	119.472	<0.001	**
Time of day	2	65.869	<0.001	**
Protection status	1	41.699	<0.001	**
Temperature	1	7.525	0.006	**
Visibility	1	27.362	<0.001	**
Depth	1	74.628	<0.001	**
Rugosity	1	386.995	<0.001	**
% Rock	1	126.444	<0.001	**
Between Effects				
Time of day*Protection status	2	62.689	<0.001	**
Protection status*Temperature	1	14.891	<0.001	**
Protection status*Visibility	1	25.236	<0.001	**
Protection status*Depth	1	13.825	<0.001	**
Protection status*Rugosity	1	0.250	0.617	ns
Protection status*% Rock	1	44.902	<0.001	**

Table A5: Results of the GLM analysis for UVC counts of fransmadam. Visibility was discarded during the preceding AIC best subsets analysis. Significance level is denoted by * ($p<0.05$), ** ($p<0.01$) or ns (not significant).

Main Effects	df	Wald (X^2)	p	
Intercept	1	0.433	0.511	ns
Time of day	2	7.027	0.030	*
Protection status	1	22.331	<0.001	**
Temperature	1	2.515	0.113	ns
Depth	1	43.068	<0.001	**
Rugosity	1	75.802	<0.001	**
% Rock	1	26.204	<0.001	**
Between Effects				
Time of day*Protection status	2	34.943	<0.001	**
Protection status*Temperature	1	1.510	0.219	ns
Protection status*Depth	1	2.422	0.120	ns
Protection status*Rugosity	1	8.142	0.004	**
Protection status*% Rock	1	38.263	<0.001	**

Table A6: Results of the GLM analysis for UVC counts of blacktail. No initial parameters were discarded during the preceding AIC best subsets analysis. Significance level is denoted by * ($p<0.05$), ** ($p<0.01$) or ns (not significant).

Main Effects	df	Wald (X^2)	p	
Intercept	1	33.391	<0.001	**
Time of day	2	48.884	<0.001	**
Protection status	1	42.569	<0.001	**
Temperature	1	15.217	<0.001	**
Visibility	1	10.305	0.001	**
Depth	1	21.124	<0.001	**
Rugosity	1	20.864	<0.001	**
% Rock	1	2.701	0.100	ns
Between Effects				
Time of day*Protection status	2	1.305	0.521	ns
Protection status*Temperature	1	26.203	<0.001	**
Protection status*Visibility	1	20.826	<0.001	**
Protection status*Depth	1	1.620	0.203	ns
Protection status*Rugosity	1	4.738	0.030	*
Protection status*% Rock	1	6.807	0.009	**

Table A7: Results of the GLM analysis for UVC counts of twotone fingerfin. Time of day and protection status were discarded in the preceding AIC best subsets analysis. There were, therefore, no between effects with protection status. Significance level is denoted by * ($p<0.05$), ** ($p<0.01$) or ns (not significant).

Main Effects	df	Wald (X^2)	p	
Intercept	1	30.247	<0.001	**
Temperature	1	6.481	0.011	*
Visibility	1	5.801	0.016	*
Depth	1	72.446	<0.001	**
Rugosity	1	99.235	<0.001	**
% Rock	1	28.160	<0.001	**

Table A8: Results of the GLM analysis for UVC counts of roman. Temperature, visibility, rugosity and percent rock cover were excluded from the model in the preceding AIC best subsets model. Significance level is denoted by * ($p<0.05$), ** ($p<0.01$) or ns (not significant).

Main Effects	df	Wald (X^2)	p	
Intercept	1	13.722	<0.001	**
Time of day	2	4.772	0.092	ns
Protection status	1	2.365	0.124	ns
Depth	1	6.189	0.013	*
Between Effects				
Time of day*Protection status	2	13.117	0.001	**
Protection status*Depth	1	1.059	0.303	ns

Table A9: Results of the GLM analysis of catch data for all species combined. Depth was discarded as a factor during the preceding AIC best subsets analysis. Significance level is denoted by * ($p<0.05$), ** ($p<0.01$) or ns (not significant).

Main Effects	df	Wald (X^2)	p	
Intercept	1	2098.926	<0.001	**
Time of day	2	6.770	0.034	*
Protection status	1	10.820	0.001	**
Between Effects				
Time of day*Protection status	2	11.061	0.004	**

Table A10: Results of the GLM analysis of catch data for roman. Time of day was discarded as a factor during the preceding AIC best subsets analysis. Significance level is denoted by * ($p<0.05$), ** ($p<0.01$) or ns (not significant).

Main Effects	df	Wald (X^2)	p	
Intercept	1	144.465	<0.001	**
Protection status	1	13.190	<0.001	**
Depth	1	8.403	0.004	**
Between Effects				
Protection status*Depth	1	1.044	0.307	ns

Table A11: Results of the GLM analysis of length data for all species combined (excluding sharks and migratory species). Time of day was discarded during the preceding AIC best subsets analysis. Significance level is denoted by * ($p<0.05$), ** ($p<0.01$) or ns (not significant).

Main Effects	df	Wald (X^2)	p	
Intercept	1	421.6256	<0.001	**
Protection status	1	110.7326	<0.001	**
Depth	1	3.9461	0.047	*
Between Effects				
Protection status*Depth	1	2.0050	0.157	ns

Table A12: Results of the GLM analysis of length data for roman. Depth was discarded as a factor during the preceding AIC best subsets analysis. Significance level is denoted by * ($p<0.05$), ** ($p<0.01$) or ns (not significant).

Main Effects	df	Wald (X^2)	p	
Intercept	1	8171.446	<0.001	**
Time of Day	2	20.797	<0.001	**
Protection status	1	48.386	<0.001	**
Between Effects				
Time of Day*Protection status	2	12.770	0.002	**