On the underwater visual census of Western Indian Ocean coral reef fishes

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REECE WARTENBERG

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Abstract

This study conducted the first high-resolution investigation of the ichthyofaunal assemblages on a high-latitude coral reef in the Western Indian Ocean (WIO). Two-Mile reef, in South Africa, is a large, accessible patch-reef, and was selected as a candidate study area. Although the effect of season in structuring coral reef fish communities is most-often overlooked, the relationship between these fish communities and their habitat structure has been investigated. In South Africa, however, neither of these potential community-level drivers has been explored. As coral reefs worldwide are faced with high levels of usage pressure, non-destructive underwater visual census (UVC) techniques were identified as the most appropriate survey methods. This study had two primary aims that were; (1) to identify the most suitable technique for the UVC of coral reef fishes, and to test variations of the selected technique for appropriateness to implementation in long-term monitoring programs, and (2) to determine if possible changes to ichthyofaunal community structure could be related to trends in season and/or habitat characteristics.

A review of the literature indicated that the most appropriate UVC method for surveying epibenthic coral reef fishes is underwater transecting. To compare the traditional slate-based transects to variations that implement digital image technology, slate transects were compared to a first-attempt digital photographic transect technique, and digital videographic transects. Videographic transects produced the most favourable species richness, abundance, and standard deviations of the three techniques. Diversity was not significantly different between transect techniques. The minimum required sample size was lowest for videographic transects (17 replicates), intermediate for photographic transects (27 replicates) and highest for slate transects (37 replicates). Videographic and photographic transects required greater analysis time to generate counts, but required lower observer training time. While videographic transects produced the lowest proportion of species considered unidentifiable, all three transect techniques showed similar functionality to surveying epibenthic coral reef fishes. Videographic transects were therefore identified as the most appropriate UVC technique for this study.

Videographic transects at shallow (6 – 14 m), intermediate (14 – 22 m) and deep (22 – 30 m) depths in mid-winter and mid-summer, sampled a total of 41 families consisting of 209 species and 18172 individuals, dominated by pomacentrids in abundance and labrids in richness. The fish assemblages on Two-Mile Reef were found to be similar in composition to lower-latitude WIO reefs. Overall ichthyofaunal abundance and richness was significantly higher in summer than in winter, and was higher at shallow sites than at intermediate and deep sites. A multivariate approach confirmed differences between seasons at shallow depths but not between seasons at intermediate and deep depths. The fish assemblages on Two-Mile Reef can therefore be described as being comprised of four relatively distinct communities: a shallow, winter community; a shallow, summer community; a year-round intermediate community; and a year-round deep community. The distributions of discriminating species indicated that high abundances of the algal-feeding pomacentrids are observed only at shallow and intermediate sites while high abundances of the zooplanktivorous serranid subfamily, the Anthiinae, are observed predominantly at deep sites. Assessment of all measured supplementary variables indicated that of all factor combinations, observed patterns could be ascribed most strongly to depth. Quantification of reef characteristics indicated that as depth increases, habitat complexity decreases, benthic communities shift from dense coral domination to sparse sponge domination, and algal biomass and cover decreases.

The ability of the videographic transect technique to detect changes in community structure with season and depth indicates that season and depth should be accounted for in future high-latitude ichtyofaunal surveys, and that the videographic transect technique is suitable for implementation in long-term monitoring programs on coral reefs. The similarity in fish assemblages between Two-Mile Reef and lower latitude regions suggests that the protocol for surveying epibenthic coral reef fishes, resulting from this study, is relevant throughout the continental WIO.

“Nature provides an exception to every rule.”

Margaret Fuller, July 1843, *The Dial*.

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List of Acronyms and Abbreviations

ACEP	African Coelacanth Ecosystem Program
DIFS	Department of Ichthyology and Fisheries Science
EAM	epilithic algal matrix
EKZNW	Ezemvelo KwaZulu-Natal Wildlife
GIS	Geographical Information Systems
GPS	Global Positioning System
kts	Knots
MDS	Multi-Dimensional Scaling
MLRA	South African Marine Living Resources Act
MPA	Marine Protected Area
NRF	National Research Foundation
ORI	Oceanographic Research Institute
PCA	Principal Component Analysis
ROV	Remotely Operated Vehicles
RVC	Rapid Visual Counts
RVT	Rapid Visual Technique
RW	Reece Wartenberg
SADCO	South African Data Centre for Oceanography
SAEON	South African Environmental Observation Network
SANBI	South African National Biodiversity Institute
UCT	University of Cape Town
UVC	Underwater Visual Census
VOS	Voluntary Observing Ships
WIO	Western Indian Ocean

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Chapter 1:

General Introduction: The survey of reef fishes

1.1 The state of coral-reef ecosystems

Ecosystems worldwide are under advanced stages of anthropogenic pressure (Jackson *et al.* 2001; Friedlander and DeMartini 2002; DeMartini *et al.* 2008). Marine systems in particular, which have a consistent history of degradation, are pressured from numerous factors that include overfishing, pollution, disease, climate change and ocean acidification (Pandolfi *et al.* 2005; Hoegh-Guldberg *et al.* 2007). Degradation has now reached a point where pristine examples of systems such as coral reefs are all but extinct (Harvey *et al.* 2001b). Recent reports on the condition of these biodiversity hot-spots warn of their precarious position (Bellwood *et al.* 2004; Wilkinson 2008). The implementation of management measures aimed at conserving and maintaining coral-reef biodiversity, and the sustainable development of the activities which depend on coral-reef systems, is therefore urgently required (Pelletier *et al.* 2011). It is critical to describe the ecological structure and function of coral-reef systems such that they can be monitored and the effects of disturbances effectively managed (Connell 1978; Harvey *et al.* 2001a).

Fishes contribute significantly to the structure and functioning of coral-reef systems (Hoegh-Guldberg *et al.* 2007) and have previously been used to describe reef health status (Jameson *et al.* 1998; Jameson *et al.* 2001). Reef fishes are uniquely diverse (Connell 1978; Ray 1988;Choat and Bellwood 1991; Caley 1995) with even small areas of coral reef containing hundreds of species that can vary in size, shape and colour (Bell and Galzin 1984; Parrish *et al.* 1986). The description and monitoring of this diverse and variable group are complex tasks, and have previously been conducted using both fisheries-dependent and fisheries-independent survey methods (Samoilys and Gribble 1997; Bennett 2008).

Fisheries-dependent data collection simply requires data from the fishery to be adequately collated and provided to researchers (Penney *et al.* 1999; Bennett 2008). The disadvantage of fisheries data are that data are based on destructive collection methods and research is restricted to localities where fisheries exist. Fisheries-independent data avoid these limitations as research can be conducted in desired focus localities using methods of choice (Samoilys and Gribble 1997). The limitation of fisheries-independent approaches is that techniques are significantly more labour-intensive. Regardless of dependence on fisheries though, destructive methods such as fishing (Francis 1995), electric shock and explosives (Harrington and Losey 1990), or the use of ichthyocides (Willis 2001) are inappropriate for the survey of sensitive coral-reef systems (Grigg 1994). Non-destructive fisheries-independent approaches offer an alternative (Die 1997).

Non-destructive survey methods are preferred for surveying reef fishes because they do not disturb the habitat, are minimally disruptive to the marine organisms, are less selective when compared to most other sampling methods, and can be replicated (Brock 1954; Harmelin-Vivien *et al.* 1985; Bortone and Kimmel 1991; Bortone *et al.* 2000; Pelletier *et al.* 2011). Non-destructive survey methods include acoustic surveys (Bodholt and Solli 1995; Tsimenides *et al.* 1995), mark-recapture experiments (Attwood and Bennett 2002; Cowley *et al.* 2002; Zeller *et al.* 2003), mark-resight experiments (Zeller and Russ 1998, 2000; Chapman and Kramer 2000), and underwater visual censuses (UVCs) (Brock 1954; Samoilys and Carlos 2000; Tessier *et al.* 2005; Langlois *et al.* 2010). Of these, UVCs are the most appropriate for community-level surveys of highly diverse coral-reef fishes as they can provide rapid, cost-effective assessments of abundance, richness, diversity and biomass (Sale and Douglas 1981; Jennings and Polunin 1995; Watson and Quinn 1997; Samoilys 1997; Kulbicki 1998). UVCs are, however, not without their limitations.

1.2 A review of underwater visual censuses

UVCs entail the *in situ* identification of species and enumeration of individuals in the underwater environment, and have been one of the most widely used methods for surveying reef fishes worldwide (Kulbicki *et al.* 2010). A variety of approaches for conducting UVCs exist (Langlois *et al.* 2010). The literature pertaining to various aspects of UVCs is extensive (Bortone and Kimmel 1991), but riddled with ambiguous and contradictory definitions (Bennett 2008). Reviews of UVCs are either outdated (e.g. Harmelin-Vivien *et al.* 1985), tied up in the grey literature (e.g. Cappo and Brown 1996; Samoilys 1997), or are specific to certain circumstances such as the survey of artificial reefs (e.g. Bortone and Kimmel 1991). In addition, many publications categorised as reviews are merely comparisons of a limited range of UVC techniques (e.g. Bortone *et al.* 1986, 1991; Francour *et al.* 1999). Bortone *et* *al.* (2000) note that scientists wishing to use UVCs are faced with assimilating a plethora of literature, a task that is both time consuming and may not be possible in developing countries with limited access to the marine science literature.

1.2.1 UVC survey techniques

Since conceptualisation in the 1950s, a variety of UVC survey techniques have been developed to estimate ichthyofaunal abundance, richness and community structure (Harmelin-Vivien *et al.* 1985; McCormick and Choat 1987; Halford and Thompson 1994; Langlois *et al.* 2010). Over time these techniques have been adjusted to better suit specific research questions. This has, in some cases, lead to vague explanations of techniques and mixed definitions (Bortone and Kimmel 1991).

Roving Diver

The roving diver technique is a plotless UVC technique that affords a high degree of survey flexibility. This technique involves divers randomly meandering through the study locality identifying the species that are encountered (Thresher and Gunn 1986). Usually no restrictions are placed on factors such as the area surveyed or a preferred observer swimming speed. Survey time is limited only by maximum dive time allowing for relatively large areas to be easily surveyed. The method is therefore most useful for producing faunal lists. Typically more species are observed than any other survey method as divers may pause to inspect detailed habitat features for as long as necessary (Schmitt *et al.* 2002). Thus the technique is particularly useful for the survey of rare or cryptic species (Kimmel 1985; Bortone *et al.* 1986, 1989) and is particularly advantageous in situations where study sites are small enough to be holistically surveyed, e.g. small patch reefs such as artificial reefs (DeMartini and Roberts 1982; Schmitt and Sullivan 1996); in this situation it has been referred to as the patch count technique (Molles 1978). Pattengill-Semmens (2001) showed that the roving diver technique can be applied in volunteer-based studies. Attempts have also been made to incorporate an abundance component by qualitatively estimating relative abundances *post hoc* – in this case the technique is known as the estimated relative abundance technique (Smith 1975; Gilligan 1980). The relative abundance technique was improved upon by Floros (2010a), who counted all fishes observed using the roving diver technique and quantified abundance per unit time. The abundance per unit time variation of the technique was incorrectly referred to as a timed point count technique, the results of which cannot be compared to studies which quantify abundance per unit area (Floros 2010a). For most research situations, the inability to determine census area when using the roving diver technique, a lack of relative abundance information, and the resulting high variability in data render the roving diver technique inappropriate for all but baseline assemblage surveys.

Random Counts

Also known as species-time random counts (Bortone and Kimmel 1991), rapid visual counts (RVC) (Kimmel 1985), or the rapid visual technique (RVT) (Jones and Thompson 1978; DeMartini and Roberts 1982), the random counts technique was first developed by Schmidt and Thompson (1977). The technique consists of methods similar to those of the roving diver technique and therefore has similar applications. The difference between the two is that as species are observed they are scored to the appropriate time interval, the underlying assumption being that more-abundant species will be observed earlier (Bortone and Kimmel 1991). A variation of the technique has been implemented that entails divers listing fishes sighted in ranking order of initial encounter for each species (Sanderson and Solonsky 1986; Thresher and Gunn 1986; Kulbicki 1998; Mapstone and Ayling 1998). Faunal lists produced are comparable to those produced by the roving diver technique with the addition of some relative abundance information. When comparing the technique to transects, Sanderson and Solonsky (1986) found the technique to be more cost effective but statistically less rigorous. The disadvantage is that abundances are reduced to scores of the probability of an encounter, further complicated by a lack of spatial data (Bortone and Kimmel 1991; Kulbicki 1998; Mapstone and Ayling 1998). Comparisons are therefore restricted to within study comparisons because density per area cannot be calculated.

Stationary Point Counts

Also referred to as instantaneous area counts (Thresher and Gunn 1986), or the quadrat technique in smaller-scale applications (Hastings 1979), stationary point counts were first developed by Bohnsack and Bannerot (1986). The technique requires divers to remain stationary at the centre of a visually estimated cylinder of a known radius counting all species visible as quickly as possible (Ward-Paige *et al.* 2010). Once conspicuous individuals have been counted in the survey area, some authors have allowed observers to search the cylinder area more thoroughly in an attempt to gain data on shy and/or cryptic species (Lechanteur and Griffiths 2002). Point counts are effective in that they can be conducted quickly, allowing for increased replication. As the area surveyed can be calculated, ichthyofaunal densities can be extrapolated (Colvocoresses and Acosta 2007). Because divers are stationary during surveys, behavioural disturbances are minimised (Colvocoresses and Acosta 2007), although Bennett (2008) noted some attraction bias by inquisitive species towards stationary divers. To maintain accuracy, the area surveyed per replicate point counts must remain low, and is often restricted by visibility. Greenwood (1996) and Samoilys and Carlos (2000) noted that there is potential error in the visual estimation of the circular boundary of a point count, and that since area is proportional to the square of the radius, any error in distance estimation will be squared. Some researchers have mitigated this problem by demarcating cylinder boundaries prior to counts (Bortone and Kimmel 1991; Polunin and Roberts 1993). This, however, causes an increase in diver disturbance that is difficult to quantify (Fulton *et al.* 2001). It is also inevitable that variability will be introduced as some fishes entering/leaving the survey area will not be detected because some part of the survey area is always behind the observer (Bennett 2008). Point counts are therefore more appropriate to smaller, heterogenous habitats and artificial reefs (Bortone *et al.* 1991). Floros (2010a) applied this technique successfully when surveying only a selected list of species. As observers must remain stationary, the stationary point count technique is more useful to restrictive survey modes such as manned submersibles.

Strip Transects

Strip transects, or tape transects (Dickens *et al.* 2011), have been used for the survey of ichthyofauna since the 1950s (Brock 1954). Strip transects require observers to swim a predetermined distance, at a constant speed, identifying and counting all fishes occurring within a set transect width. A variation of strip transects are band or belt transects where two parallel lines are laid along the reef substrate prior to conducting each replicate (Davis and Anderson *et al.* 1989; Ratikin and Kramer 1996; Barrett and Buxton 2002) thereby mitigating the need for visual transect width estimation. The suitability of belt transects is questionable as they require increased dive time for the establishment of transect boundaries and are associated with increased pre-survey diver disturbance which is difficult to quantify (Fulton *et al.* 2001). Recently Pelletier *et al.* (2011) compared what they termed I- and S-type transects using videographic media, where I-type transects represented traditional strip transects and S-type transects were a transect variant allowing observers to alter elevation and move the camera from side to side, resulting in the survey of an increased number of fishes.

I-type transects were found to be inferior for estimating fish abundances when compared to S-type transects. As the S-type transect has not been thoroughly evaluated, research and development of the technique, and its possible applications, is recommended. It must be noted that Alevizon and Brooks (1975) used pseudo-S-type transects in their early study. Strip transects are advantageous in that the technique is easy to implement and constitutes the most well-defined survey protocol as transect length, width, and height must be standardised prior to study commencement. The comparatively large survey area per replicate afforded by strip transects is suitable for assessing large expanses of reef with diverse ichthyofaunal assemblages (Sale 1991; Edgar 2004; Colvocoresses and Acosta 2007) and species with non-random distributions (Kulbicki and Sarramégna 1999). Strip transects have become the most widespread UVC technique (Kingsford and Battershill 1998) but, because of the popularity of the technique, its application has varied widely among users (Bortone and Kimmel 1991). For example, the length of transects conducted has varied from as short as 5 m (Anderson *et al.* 1989) to as long as 500 m (Thresher and Gunn 1986). Because the area surveyed can be easily calculated, inter-study comparisons of metrics such as density are possible. Gledhill *et al.* (1996) and Francour *et al.* (1999, as cited by Tessier *et al.* 2005) suggested that, due to the forward-moving nature of transects, the method is likely to reduce the risk of duplicate counts of species such as schooling species, a criticism of the point count technique. Disadvantages of the transect technique include the behavioural disturbances associated with a moving diver and the inability of divers to search specific refuge habitats as swimming speed must remain constant (Watson and Quinn 1997; Schmitt *et al.* 2002). Consequently, the strip transect technique is well-suited to the survey of more conspicuous, epibenthic reef fishes. Samoilys and Carlos (2000) and Edgar (2004) conclude that despite suffering from limitations likely to result in underestimates, strip transects are generally recognised as the most practical method for obtaining relative density estimates of fish communities.

Distance estimates

Distance estimates, distance sampling, or instantaneous variable distance counts have been implemented using survey areas equivalent to point counts (Thresher and Gunn 1986) and strip transects (Fowler 1987). Observers conducting either a point count- or strip transect-type survey must identify individuals and estimate the distance between the individual and the observer. The technique is used to estimate density or abundance of species under the assumption that a detected individual will be detected with certainty. A detection function, representing the probability of detections as a function of distance from the observer, can then be modelled. Appropriate survey design allows the resulting object density estimates to be extrapolated to the full survey region, which yields estimates of object abundance (Buckland *et al.* 2001). Distance estimates, originally developed for use by ornithologists, were implemented as a method of UVC primarily to mitigate the error associated with the visual estimation of survey area (Thresher and Gunn 1986; Harvey *et al.* 2004). Harvey *et al.* (2004) showed that, although the requirement for survey area estimation is mitigated, the error associated with the estimation of distance to enumerated individuals was significant for both novice and experienced divers. Harvey *et al.* (2004) concluded that the technique can only be successfully implemented where stereo-video equipment is available. Criticisms of the technique include the error associated with the estimation of distance, and high observer task loading in particularly diverse areas. The technique is also time consuming and is therefore best suited to selected species rather than whole communities.

Interval Counts

The interval count, or timed count, technique can be applied to point count (Bortone *et al.* 1996; Floros 2010a) or strip transect survey areas (Bortone and Kimmel 1991). The difference with the interval counts modification is that, instead of counting all individuals observed within the survey boundaries, all individuals moving through the survey area up to a predetermined time limit are counted. The first application of the interval count technique was demonstrated by Alevizon and Brooks (1975) who used video cameras to film a point count area to maximum film exposure time. The advantage of the technique is that the process of determining when a count is complete is no longer subjective. Densities are calculated assuming that inter-site, temporal and interspecific differences in rates of movement are negligible (Thresher and Gunn 1986). Thresher and Gunn (1986) state that although the first two assumptions are reasonable, the third rarely holds. The technique is thus not suitable for inter-species comparisons of density but can be effective in comparing single species densities between areas.

Spot Mapping

Spot mapping, or home-range mapping (Edgar 2004), another technique initially developed by ornithologists, has been used for surveying fishes by Thresher and Gunn (1986). The labour-intensive technique requires detailed mapping of territories and home ranges of selected species at randomly chosen points within the area of interest. Density is estimated using mean home-range size and the spacing between home ranges. For a full explanation of the technique refer to Thresher and Gunn (1986) who concluded that the method was inappropriate for use in highly mobile organisms unless a tagging component was incorporated. Since its implementation by Thresher and Gunn (1986), the spot mapping method has not been used as a feasible UVC technique.

Point Diversity

The point diversity technique is another labour intensive UVC technique for the survey of ichthyofauna. Effectively a combination of a pseudo-transect and point counts, the technique was designed by Slobodkin and Fishelson (1974) for the survey of ichthyofaunal communities associated with specific reef features. Observers swim transects at regular intervals on a study reef, where transect length is limited only by the length of the study area. When a desired reef feature is encountered, a thorough point count of that feature is conducted before observers continue the same transect. Information on the ichthyofaunal density of the reef feature is obtained and community structure comparisons between features can be made. Slobodkin and Fishelson (1974) used the technique to effectively map and investigate the community structure of fishes at „cleaning station‟ areas where the wrasse *Labroides dimidiatus* occurred. It is anticipated that the technique could be used equally effectively to investigate ichthyofaunal community structures between different reef structures such as pinnacles, caves and crevasses, particularly if the localities of these features are not previously known. Information of this kind will be useful in understanding determinants of diversity (Bortone *et al.* 1986).

1.2.2 UVC survey modes

UVCs can be conducted using a number of survey modes. Presented here is an explanation UVC modes, ranging from those methods that require basic/minimal equipment, training and funding, to more complex and expensive methods. These include snorkelling, open-circuit SCUBA, closed circuit SCUBA, remote operated vehicles, and manned submersibles.

Snorkelling

Snorkelling is a cost-effective, simple method for conducting UVCs in relatively shallow waters. First used by Bardach (1959), observers require only a mask, snorkel and swim-fins to conduct the UVC technique of choice. Frias-Torres (2006) and Serafy *et al.* (2003) successfully used snorkelling to conduct 30 m long transects in depths <5 m. At slightly deeper depths the technique is referred to as „freediving‟ as observers must carry out in-water apnea to reach the sample depths. Lechanteur and Griffiths (2002) successfully used freediving, supplemented by SCUBA, to conduct point count surveys of temperate reef fishes up to depths of 10 m. A benefit of the technique is that observer noise, and hence disturbance, is kept to a minimum as no breathing apparatus is required. Freediving UVCs can only be conducted by experienced freedivers and are restricted to comparatively shallow sample depths and to UVC techniques that require short survey times per replicate. Other authors that have used snorkelling or freediving include Jones and Chase (1975), Stephens *et al.* (1984) and Larson and DeMartini (1984).

Open circuit SCUBA

Open circuit self-contained underwater breathing apparatus diving, or SCUBA diving, has been used since the inception of UVCs (Brock 1954). SCUBA allows divers to operate to a maximum depth of 30 m (South Africa, Department of Labour 2010). Although the technique is relatively simple to implement, training in the use of SCUBA equipment is necessary. The disadvantage of SCUBA is that, due to the build-up of residual nitrogen in the blood of observers, maximum dive time is short, decreases with increasing depth, and requires a sufficient off-gassing period for observers between dives. A criticism of SCUBA is that bubbles exhaled by divers may affect fish behaviour (Chapman *et al.* 1974; Cole *et al.* 2007). To date, open circuit SCUBA is the most commonly used mode of conducting UVCs because in comparison to other modes equipment is cheap and readily available, many sample reefs occur shallower than maximum operating depths, and training time is comparatively short.

Closed circuit SCUBA

Although an early version of the technology was first used in biological observations by Hanlon *et al.* (1982), closed circuit SCUBA diving, or rebreather diving, has been developed more recently (Pyle 2000; Lobel 2001; Bahuet *et al.* 2007). Two systems currently exist: semi-closed circuit and closed circuit rebreathers. Both methods recycle predetermined proportions of exhaled air, which significantly reduce the noise and bubbles associated with open circuit SCUBA (Radford *et al.* 2005). The primary advantages of rebreather diving are that, at depths equivalent to those dived using open circuit SCUBA, significantly longer dives can be conducted safely and significantly shorter off-gassing periods are required between dives. In addition, closed circuit divers can safely reach depths of over 200 m, but the associated in-water decompression and surface off-gassing times increase substantially (Parrish and Pyle 2002; Baheut *et al.* 2007). The disadvantage of the technique is that equipment is more costly and the training is extensive. Because the technology is still relatively new, the use of closed circuit SCUBA has not been widely implemented. In the near future, rebreather diving may entirely replace open-circuit SCUBA for conducting UVCs.

Manned submersibles

Manned submersibles were the first underwater survey mode for conducting surveys at depths greater than those safe for SCUBA diving (Rowe *et al.* 1975; Shipp *et al.*1974; Parker and Ross 1986; Ralston *et al*. 1986; Thresher and Colin 1986). Usually two to four observers will descend to sample depth within the confines of a submersible. Submersibles are usually constructed with viewing windows and often utilise photographic and videographic cameras for the observation of aquatic organisms. This relatively cumbersome mode of conducting UVCs is usually implemented when depths of over 200 m must be safely sampled for extended periods of time. As pressure within the submersible is maintained at 1 ATM, maximum dive time is limited only by logistical constraints. Limitations of the technique are that training of submersible pilots is lengthy, and the submersible and its associated equipment are extremely expensive. Submersibles are generally large and cumbersome and must therefore be launched from relatively large vessels (Csepp 2005). Krieger (1993) used a submersible to successfully survey fishes at depths ranging from 188 to 290 m. Costello *et al.* (2005) used modern technology and found the technique to be successful for fish identification and habitat description purposes. However, Moffit *et al.* (1989) state that manned submersibles are, in many respects, more difficult to implement than other modes of UVC and are thus impractical for use at shallower depths. Moffit *et al.* (1989) also found that because manned submersibles are generally not manoeuvrable, they are better suited to relatively open areas that require surveying.

Remotely Operated Vehicles (ROVs)

The use of ROVs for the assessment of fish assemblages was first described by Thompson *et al*. (1982). ROVs, mounted with videographic equipment, are controlled by surface-based operators who use live video feed, with an overlay of necessary information, to control the vehicle at the sample depth. The advantages of using ROVs, depending on make and model, are that they can be operated to depths far greater than open- or closed- circuit SCUBA, they are not restricted in terms of dive time, and the risk to researchers conducting underwater surveys is mitigated (Pacunski *et al.* 2008). The disadvantages of ROVs are that equipment is costly and that vehicles can only be operated by experienced, extensively trained pilots. Similar to manned-submersibles, ROVs must be deployed from a large vessel, although the operation and deployment of newly developed, smaller ROVs from small boats has been explored (Csepp 2005). In an early study conducted by Greene and Alevizon (1989), using early ROV technology, the technique was found to result in low accuracy data and was relatively inefficient when used in a controlled aquarium environment where actual fish assemblage structure was known. Vrana and Schwartz (1989) used an advanced instrument sled attached to their ROV to improve ROV performance. Adams *et al.* (1995) used an ROV system equipped with advanced cameras, lenses, lights, and fibre-optic cables to conduct 200 to 600 m long transects more successfully. Adams *et al.* (1995) found that their ROV, compared to swept area trawls, produced higher estimates of abundance with low coefficients of variation. They also considered the ROV to be superior to swept area trawls because observers are capable of directly assessing the underwater environment, enabling observed patterns to be related to habitat. Although O"Connell and Carlile (1994) have shown that, due to logistical and technical problems, successful deployments of ROVs can be as low as 60%, it is possible that with further ROV development they may become the mode of choice for conducting UVCs at any depth. Costello *et al.* (2005) conclude that the use of ROVs offers an ideal method by which to link biological observations with precise knowledge of the habitat under study.

1.2.3 UVC survey media

All UVC techniques, and the modes with which they are conducted, require media based on either the *in situ* recording of data by observers, or the *post hoc* examination of visual records (Edgar 2004). Often the term UVC is used to describe surveys specifically using eyesight to directly identify and enumerate fishes (e.g. Samoilys 1997; Colton and Swearer 2010). Despite the name underwater „visual" census, not all UVCs are conducted by direct visual assessment of fish assemblages. The media available for conducting UVCs now range from simple to advanced technology and include: slates, audiotapes, photographic stills, and videographic media.

Slate

Until the mid 1990s plastic slates of various types were used for almost all transcribing of underwater data (Helfman 1983; English *et al.* 1994). Visually observed data are typically written on slates by an observer using a pencil. The advantages of slate media are that the technique is inexpensive and fully adaptable to any UVC mode. Disadvantages to using slates are that valuable observation time is lost as the diver must look at the slate to record data (Brock 1982; Bortone and Kimmel 1991) and there is no reference of each replicate, only the raw data. A further disadvantage is that the technique is entirely dependent on the ability of the observer to make correct identifications rapidly and *in situ*, which can be a difficult task in highly diverse areas. Identification error resulting from rushed identifications could have particularly severe consequences in areas where few ichthyofaunal surveys have been previously conducted as the community represented is almost entirely dependent on the knowledge of observers, previously gained either from experience or from the literature. Despite these disadvantages, slate media are still regularly used for conducting UVCs (e.g. Colton and Swearer 2010; Floros 2010a; Dickens *et al.* 2011; Pelletier *et al.* 2011; Williams *et al.* 2011).

Audiotape

Audiotape was introduced to mitigate diver distraction while recording data (Alevizon *et al.* 1985). Observers, equipped with underwater audio recording equipment, log verbal records of necessary observations *in situ* (Bortone *et al.* 1991). Greene and Alevizon (1989), who compared audiotape media to slate and videographic media, found the audiotape methods to be more cost effective, efficient, and accurate when estimating ichthyofaunal abundance. Bortone *et al.* (1991), who used larger sample sizes, disagreed with Greene and Alevizon (1989) and suggested that audiotape in conjunction with videographic media was a more favourable method. As with slate media, however, descriptions of ichthyofaunal communities using only audio media are dependent on the ability of the observer to rapidly and accurately identify fishes. As audio records are stored, some reference record of each replicate is available.

Traditional slate and audiotape methods are steadily being replaced by image-based media because of five primary advantages (Pelletier *et al.* 2011). First, image-based media do not require the presence of fish identification experts in the field as media may be analysed at any time (Pelletier *et al.* 2011). Second, digital-based media can reduce the time required underwater allowing time for an increased number of observations. Third, images allow scope for the analysis of detailed environmental information such as reef architecture or benthic community structure. Environmental information that can be obtained from image media includes, but is not limited to, habitat availability, benthic community structure, and reef architecture (Simpson 1977; Roberts *et al.* 2000; Costello *et al.* 2005). Fourth, images may be reviewed multiple times by multiple observers, thus limiting and allowing for quantification of observer bias (Preuss *et al.* 2009). Lastly, image media may be analysed for other purposes such as identifying the sex-ratios of sexually dimorphic species (e.g. Platten *et al.* 2002), or assessing the degree of coral-bleaching (e.g. Gates *et al.* 1992). The disadvantage of digital media is that time must be spent transcribing data, which can be greater than the time taken to make the initial recording (Bortone and Kimmel 1991). There are two types of image-based media: stills photography and videography.

Stills photography

The use of stills photography in UVCs provides a permanent database of information stored as still images. Stills photography requires that observers are equipped with photographic equipment capable of descending to the appropriate sample depth. Lundälv (1971) first used stills photography in UVCs to record population information and geographic conditions. He noted however, that the technique performed poorly under low visibility conditions. Weinberg (1981) used stills photography to create a photographic record of sections of reef, but the technique proved only partially successful as supplemental drawings were required. It was concluded that, at that stage, stills photography was impractical because of the equipment and facilities required for a technique that was not producing the required results. Langton and Uzman (1989) conducted photographic surveys of fishes using a submersible in conjunction with a roving diver approach. Stills photography was successfully employed by Langton and Uzman (1989) as the aim of their study was merely to catalogue the species present. Langton and Uzman therefore recommended stills photography as an aid to fish identifications, but not as a tool for obtaining relative abundance information. In the mid 1980s technology improved and Bortone *et al.* (1986) were the first to use photographs with the statistically more rigorous transect technique. Using strobe-illuminated photographs to survey fish assemblages, Bortone *et al.* (1986) found the photographic transect technique to be relatively successful, but noted that one disadvantage to stills images was that the lack of movement against a background hindered the identification of fishes. As stills photography was proven to be largely unsuccessful in the past, no further studies have investigated its applicability to more advanced survey techniques such as transects. With the advancement of new digital technology, and associated cost decreases, it is possible that stills photography will become a useful media for conducting UVCs.

Videography

Videographic equipment has been used in many areas of marine research since the 1960s (Barnes 1963; Myrberg 1973, as cited by Potts *et al.* 1987; Smith and Tyler 1973; Alevizon and Brooks 1975). Initially underwater television (Myberg 1973; Smith and Tyler 1977) and underwater movie (Ebeling *et al.* 1980) equipment were cumbersome to operate and required advanced, specific alterations to standard land-based video equipment (e.g. Potts *et al.* 1987). In the last two decades, with the advent of digital devices, video recording has become an appropriate, simple tool for surveying fish assemblages (Tessier *et al.* 2005; Langlois *et al.* 2010; Pelletier *et al.* 2011). The benefit of videographic media over photographic media is that identifications are made using moving images allowing for factors such as swimming behaviour and variable orientations towards the camera to facilitate identifications. There are three primary applications of videographic media in UVCs; single-video, stereo-video and remote-video.

Single-video

Single-video refers to the use of a single, standard video camera in an appropriate underwater housing. Standard video media can be used with any UVC techniques and/or mode and requires only that observers operate a video camera at the sample depth. Bortone *et al.* (1986) successfully used single videographic media in variations of transects, termed linear cinetransect, and point counts, termed cineturrets, or circular cinetransects. Two recent studies implementing single-video media were conducted by Harvey *et al.* (2010) and Pelletier *et al.* (2011). Harvey *et al.* (2010) compared single digital video to high definition digital video and concluded that high definition is the preferred format. Pelletier *et al.* (2011), citing that high definition video is becoming the standard, compared only the high definition format to slate techniques. Pelletier *et al.* (2011) found that although fewer species and individuals were detected using videographic media, results were comparable.

Stereo-video

In an attempt to reduce some of the biases associated with traditional UVC methods, stereo-video systems have recently been introduced as a UVC media (Boland and Lewbell 1986; Harvey and Shortis 1996; Harvey *et al.* 2001a, b). The stereo-video system requires the calibration of two single-video cameras, which are mounted together on a portable frame, such that length and distance can be measured *post hoc* using specialised software. Length and distance measurements are usually applied to the determination of survey areas for density calculations and/ or fish lengths for biomass calculations. In the case of non-mobile survey techniques, such as point counts, the stereo-video system frame will rest on the seafloor (e.g. Harvey and Shortis 1996; Petrell *et al.* 1997; Watson *et al.* 2005) or, in the case of transects, is carried by divers (e.g. Watson *et al.*2005). The primary advantage of stereo-video is that precise and accurate estimates of sampling area and fish size are arrived at in a more objective, unbiased manner. Stereo-video can be similarly implemented to quantify diver effects and observer bias in distance and length estimation (Harvey *et al.* 2004). Stereo video is also less restricted by range and subject orientation than standard video and has therefore been successfully applied in aquaculture for the estimation of fish density and biomass in enclosures (Naiberg *et al.* 1993). The primary disadvantage of this media is that there is more than a two-fold increase in necessary equipment, access to advanced stereo-video software is required, and divers must operate significantly more equipment at the sample depth (Harvey *et al.* 2001a, b, Harvey *et al.* 2002a, b; Watson *et al.* 2005; Cappo *et al.* 2005). Despite the disadvantages, where funding and access to stereo-video technology is available, it is recommended that stereo-video systems be selected over standard video media. One shortcoming of all UVC media is that, where advanced diving equipment, such as closed-circuit SCUBA and ROVs are unavailable, UVCs are restricted to shallow depths and a limited number of deployments per diver per day. These depth and replicate limitations can be minimised using remote-video media.

Remote-video

Remote-video media allow for UVCs to be conducted without *in situ* observers. Typically observers will deploy a video system, attached to an appropriately designed frame and linked to the surface, to the sample depth. Although the applications of remote video media to conducting transects on sleds suspended below a survey ship have been investigated (Costello *et al*. 2005), remote-video media are used more commonly in variations of the point count technique. Remote-video media can be applied to both single-video (Francour *et al.* 1999, as cited by Tessier *et al.* 2005) and stereo-video (Watson *et al.* 2005; Watson *et al.* 2010) applications. Two examples of the remote-video point count technique are baited and non-baited video stations – the comparative advantages and limitations of which have been examined (Watson *et al.* 2005; Willis *et al*. 2000; Willis and Babcock 2000; Harvey *et al.* 2004; Cappo *et al.* 2005, 2007; Stobart *et al.* 2007). The primary advantage of remote video is that sample depth is not limited by safe diving for observers (Bortone and Kimmel 1991) but rather by logistics such as sea surface conditions, the topographic characteristics of the seafloor, and the specific study objectives. Priede *et al.* (1994) successfully used baited remote cameras to sample depths up to 4100 m, but supplemented relative abundance estimates with trawl catch data. As no *in situ* observers are present, the primary limitation of the remote-video is that the technique is inflexible and is significantly influenced by visibility (Willis and Babcock 2000; Cappo *et al.* 2005). As the camera is immobile, cryptic species are excluded and it is impossible to ascertain if individuals are entering the survey area repeatedly (Ellis and DeMartini 1995; Willis and Babcock 2000). The technique is therefore better suited to investigating the relative abundance of specific species rather than overall fish assemblage structures.

1.2.4 Techniques for analysing UVC media

Variations of simply counting all fishes observed in image-based survey media have been developed to minimise some of the limitations of UVCs. These include; *Standard-counts*, *MaxN* and *Multiple MaxN* techniques. Unfortunately there has been no attempt at comparing the appropriateness of each analysis technique to specific applications.

Standard-counts

Standard-count analysis refers to the traditional, most simple technique for analysing media: simply identifying and counting all individuals observed on images (e.g. Potts *et al.* 1987). Although all traditional slate surveys effectively implement the *Standard-count* technique, *Standard-counts* were first used for analysing image-based media with the implementation of the first video surveys (e.g. Alevizon and Brooks 1975; Potts *et al.* 1987). The primary advantage of the *Standard-count* technique is that it provides representation of the entire community surveyed. The primary limitation is that there is a possibility of duplicate counts if the same individual is counted multiple times. Duplicate counts are of particular concern in stationary, attraction-type techniques such as baited underwater video surveys because fishes will aggregate at the bait site and re-enter the survey area numerous times. To reduce this error the *MaxN* technique was developed.

MaxN

MaxN analysis, also referred to as n_{peak} (Priede *et al.* 1994), *MAXNO* (Ellis and DeMartini 1995) and *MAX* (Willis and Babcock 2000), was developed specifically to reduce the occurrence of double counts using stationary video techniques (Colton and Swearer 2010). *MaxN* requires that only the single frame in a replicate, i.e., a video clip, containing the greatest number of individuals for a species is taken as the count for that species (Priede *et al.* 1994; Willis and Babcock 2000). *MaxN* counts are conducted on a per-species basis until all species in a replicate are accounted for (Priede *et al.* 1994). *MaxN* is appropriate to situations where all species exhibit a common grouping behaviour, such as aggregating around a bait site. The technique is also useful when conducting UVC video point counts in areas where many species exhibit schooling behaviour as it reduces the associated over-representation of those species (Gledhill *et al.*1996, Francour *et al.*1999, as cited by Tessier *et al.* 2005). Applying the *MaxN* approach in inappropriate situations, such as circumstances where species do not aggregate, can result in „perverse" estimates of abundance (Priede and Merrett 1996). *MaxN* may have limited use in conjunction with mobile UVC methods, such as transects, which are characterised by a forward movement, in a single direction, at a constant speed, an action that should reduce the occurrence of duplicate counts and the associated error regardless of the media analysis technique employed (Watson *et al.* 2005, Bennett 2008).Videographic transects will reduce duplicate counts as it is possible to track individuals/schools throughout a video clip such that the large majority of double counts are avoided. Watson *et al.* (2005) noted that while the use of *MaxN* as a relative density measure for remote techniques avoids repeated counts, it can be assumed that transect techniques also avoid duplicate counts. Regardless of the UVC technique implemented, *MaxN* analysis is appropriate to investigations comparing the abundance of single species, or species grouped by similar behaviour (Stobart *et al.* 2007).

Multiple MaxN

Bortone *et al.* (1986) provide the earliest example of implementing a slight variant, *Multiple MaxN* analysis. Bortone *et al.* (1986), who referred to their approach as a *framework* approach, conducted 100 m long photographic transects consisting of ten stations. Four photographs were captured per station and, for each species observed, abundance was estimated as the mean of the number of individuals counted in each photo, for each station, over the entire transect. More recently, *Multiple MaxN* has been used as an extension to *MaxN* analysis in an attempt to gain more representative information from each replicate. The analysis requires that each clip of footage be divided into equal segments, essentially pseudo-replicates. The original *MaxN* approach is then applied to each segment and the mean value of species counts over all segments is accepted as the count of each species for that replicate (Colton and Swearer 2010; McKinley *et al.* 2011).

1.2.5 Shortcomings of UVCs

No sampling protocol, even if rigorously statistically conceptualised, is perfect. UVCs are no different as all methods used to investigate various aspects of ichthyofaunal assemblages have shortcomings (Edgar 2004). Besides experimental design flaws, which are reviewed by Bennett (2008), the shortcomings of UVCs are generally centralised around error in the form of bias and variability. Error, bias and variability are terms often used interchangeably in the literature resulting in blurred definitions and unclear explanations (e.g. Colvocoresses and Acosta 2007).

Biases usually result from observer error and/or the nature of the UVC technique and are associated with: the presence of the observer affecting fish behavioural changes (Sale and Sharp 1983; Thresher and Gunn 1986; Lincoln Smith 1988; Cole 1994; Kulbicki 1998; Watson *et al*. 2007), for example, the attraction or repulsion of some species by divers (Minte-Vera *et al.* 2008; Cowley and Naesje 2004); observer experience in conducting a UVC (Sale and Sharp 1983; Mumby *et al.* 1995; Thresher and Gunn 1986; Lincoln Smith 1988; St John *et al.* 1990); observer swimming speed (Sale and Sharp 1983; Thresher and Gunn 1986; Lincoln Smith 1988 and 1989); the difference between true values and estimates based on observations that often manifest as underestimates of highly abundant species (Stewart-Oaten *et al.* 1995; Willis 2001); length estimation of fishes by observers (Harvey *et al.* 2001a, b, 2002a); subjective decision making and underwater behaviour of observers

(Edgar 2004), such as site or pass choice and observer swimming patterns (Cheal and Thompson 1997; Harvey *et al.* 2001a); double counts, i.e. individual fish are counted twice (Samoilys and Carlos 2000); failure of observers to notice individuals (Brock 1982; Sale and Sharp 1983; Watson et al. 1995; Watson and Quinn 1997), which is of particular relevance when cryptic, small or nocturnal species are concerned (Brock 1982; Kulbicki 1998; Ackerman and Bellwood 2000; Willis 2001; Minte-Vera *et al.* 2008); incorrect identifications (Lincoln Smith 1988; Legg and Nagy 2006), which are of particular concern in highly diverse systems such as coral reefs; the estimates of range for determining survey boundaries such as transect length or width (Harvey *et al.* 2004); and species and/or size selectivity of the UVC technique being implemented (Thompson and Mapstone 1997; Willis 2001; Edgar 2004).

Sources of variability usually result from natural phenomena and include: temporary, localised or small scale shifts in the distribution and abundance of individuals (Sale and Douglas 1984; Thompson and Mapstone 1997; Ault and Johnson 1998; Tessier *et al.* 2005), such as the movement of reef fishes linked to tidal or diurnal cycles (Watson *et al.* 1995); permanent shifts in the distribution, abundance and sizes of specimens (Sanderson and Solonsky 1986; Thompson and Mapstone 1997; Mapstone and Ayling 1998); natural changes, such changes in abundance associated with, for example, environmental, climatic or oceanographic change (Samoilys and Carlos 2000; Thompson and Mapstone 2002; Tessier *et al.* 2005); intra-species variability in mobility or detecability (Edgar 2004; Thompson and Mapstone 2002), such as changes in detectability resulting in variable species coloration depending on particular fish behaviours; fishes moving across boundaries of sampling units during the time taken to survey each unit, regardless of whether they move away from the sampling site, or respond positively or negatively to divers (Watson *et al.* 1995; Kulbicki 1998); and the underwater visibility and structural complexity of the habitat (Edgar 2004).

Several studies have assessed the shortcomings of UVCs relating to bias and variability (Harmelin-Vivien *et al.* 1985; Jennings and Polunin 1995; Watson *et al.* 1995; Cheal and Thompson 1997; Kulbicki 1998; Kulbicki and Sarramegna 1999; Samoilys and Carlos 2000; McNeill *et al.* 2008; Langlois *et al.* 2010; Ward-Paige 2010; Pelletier *et al.* 2011). Thresher and Gunn (1986) and Edgar (2004), in assessing the effectiveness of UVCs, found that although UVCs can be affected by natural variability, such error is generally systematic, random, and is probably not as large as is widely perceived. Sources of bias, however, do not possess these qualities. The effect of bias is, nevertheless, reducible through a careful, standardised study design that is aimed at reducing statistical noise (Winer *et al.* 1991; Willis *et al.* 2000). In this situation bias and variability can be statistically separated and quantified. Yet some sources of bias cannot be removed in the study design process. An example is the species and size selectivity of the UVC technique being implemented. Although Ackerman and Bellwood (2000) state that unavoidable bias is an acceptable limitation of UVCs, it is important that these limitations are considered when drawing inferences.

1.3 Study objectives and research plan

As anthropogenic pressures, whether direct, such as fishing and pollution, or indirect, such as climate change, continue to mount on marine environments, baseline information and the development of long-term monitoring in appropriate and representative areas is a matter of urgency. The south Western Indian Ocean (WIO) is a biodiversity hotspot largely because it contains the world‟s southernmost coral reefs in an area considered only marginal for coral growth (Schleyer 1999; Schleyer and Celliers 2003a, b) and therefore a unique biogeographic subregion. Although corals in the south WIO have been well-documented, information pertaining to the fish assemblages of the high-latitude WIO coral reefs is scarce (Floros 2010a). An area where easily accessible high-latitude coral reefs occur is the iSimangaliso Wetland Park in South Africa.

Current trends in climate change have caused warming of South African coastal waters resulting in a shift in coral community structure from hard to soft corals – probably the cause of an evident increase in coral accretion (Jordan and Samways 2001; Celliers and Schleyer 2008). Schleyer and Celliers (2005) have warned that sea surface temperatures are already approaching the local coral bleaching threshold after Celliers and Schleyer (2002) previously reported the occurrence of localised, small-scale bleaching on a cluster of South African coral reefs known as the Central Reef Complex.

The potential for future bleaching events on the South African coral reefs indicates that continued research and the development of long-term monitoring are critical. Schleyer and Celliers (2003b) suggest that changes evident on the marginal coral reefs of South Africa are likely to precede changes on typical, non-marginal coral reefs in the greater WIO. Baseline research in this unique, biodiversity hot-spot is therefore essential. This study conducted extensive ichthyofaunal surveys of the fish assemblages of Two-Mile Reef, a South African coral patch reef, in an attempt to ascertain which factors drive ichthyofaunal community structure and dynamics.

The aim of this thesis was to provide baseline spatial and temporal descriptions of high latitude WIO coral-reef fish communities using a non-destructive, standardised UVC technique suitable for implementation in future long-term monitoring work in the greater WIO. Specific objectives included:

1) Provide a critical review of the intricacies of conducting UVCs and the shortcomings associated with UVCs.
2) Review the available physical and biological information pertaining high-latitude WIO coral reefs in a South African context using a broad-scale ecological approach.
3) Provide the information required for developing a standardised protocol for the survey of coral-reef fish assemblages based on a meta-analysis of the literature.
4) Develop a standardised protocol specifically geared towards surveying the epibenthic reef fishes of the South African coral reefs.
5) Develop a method for conducting transects using digital photographic media.
6) Determine the most appropriate UVC transect technique for surveying the epibenthic reef fishes of high-latitude South African coral reefs by comparing slate, photographic, and videographic transects.
7) Determine the most appropriate media analysis technique for analysing footage generated by digital photographic and videographic transects.
8) Provide a first attempt at validating reef fish counts generated from reviewable digital imagery.
9) Describe the ichthyofaunal communities of a large, high-latitude WIO patch reef using the most appropriate techniques determined by achieving objectives 1–8.
10) Georeference possible spatial and temporal variation in univariate fish assemblage metrics using geographical information systems.
11) Determine if season, a factor largely ignored on low-latitude reefs, has a significant effect in structuring high-latitude coral-reef fish assemblages.
12) Determine if depth and its associated changes in habitat have a significant effect in structuring high-latitude coral-reef fish assemblages.
13) Describe any change in fish community structure, evident from objectives 10–12, in an ichthyofaunal functional group context.

14) Contexualise this research such that the implications of the conclusions in this study are understood in relation to their relevance in the field of UVCs and their relevance to surveying high-latitude coral-reef fishes
15) Provide suggestions for the implementation of a long-term monitoring program appropriate to the iSimangaliso Wetland Park marine protected area, KwaZulu-Natal, South Africa.

This thesis, therefore, not only describes the fish communities of a high-latitude coral reef, but first determines the best approaches to achieving this objective.

In Chapter 1, a brief summary of the pressure facing marine ecosystems, a review of underwater visual census survey methods, and the key objectives that this thesis aims to address, are provided.

In Chapter 2, a broad-scale ecological review of the literature pertaining to the high-latitude coral reefs of the south Western Indian Ocean is contextualised from a South African perspective. The motivation for selecting the representative study reef investigated in this thesis, Two-Mile Reef at Sodwana Bay, is presented. Biological information relevant to Two-Mile Reef is discussed.

In Chapter 3, an initial standardised protocol for surveying epibenthic coral-reef fishes, based on the best available information in the literature, is presented. Standardised approaches to selecting an appropriate UVC technique, experimental design, the establishment of appropriate survey parameters, the quantification of supplementary variables, observer-related standardisation, and transect deployment standardisation are presented. This protocol is refined and then implemented in Chapters 4 and 5 respectively.

In Chapter 4, a stratified, random sampling design was used to compare the appropriateness of slate, photographic and videographic transecting to surveying high-latitude epibenthic reef fishes on Two-Mile Reef. In addition to the design and development of a method for stills-photography transecting, a preliminary method for validating counts generated from reviewable digital imagery is presented.

In Chapter 5, using a stratified, random sampling design in conjunction with the conclusions of the previous four chapters, a detailed ichthyofaunal survey of Two-Mile Reef was conducted. Sampling was stratified by season and by depth such that possible changes in ichthyofaunal community structure, and the drivers of those changes, could be determined. In addition, the seasonal- and depth-related dynamics of ichthyofaunal functional groups was examined.

In Chapter 6, the implications of the research conducted for this thesis is contextualised so that conclusions are understood in relation to their relevance in the field of UVCs and their relevance to surveying high latitude coral-reef fishes. Lastly, the recommended approaches to implementing a long-term monitoring program in the iSimangaliso Wetland Park, which contains numerous high-latitude coral reefs, are outlined.

Chapter 2:

Study Area: Physical and biological information relevant to the south Western Indian Ocean

2.1 Introduction

Almost all the accessible, high-latitude coral reefs of the Western Indian Ocean (WIO) fall into an area known as the iSimangaliso Wetland Park situated on the KwaZulu-Natal Coast, South Africa (Fig. 2.1) (Schleyer 1999). Formerly the Greater St Lucia Wetland Park (GSWP) until 2007, the iSimangaliso Wetland Park is a World Heritage Site (South Africa, Department of Environmental Affairs and Tourism 2000) and is considered to be one of South Africa‟s most valuable national parks (iSimangaliso Wetland Park Authority 2009).

The north-eastern park boundary lies on the Mozambique border. The south-western park boundary is situated 160 km north-east of Durban. The marine sector of the iSimangaliso Wetland Park, first protected under the Natal Nature Conservation Ordinance of 1974, is 155 km long (26° 51‟26‟‟S, 32° 06‟25‟‟E to 28° 29‟07‟‟S, 32° 56‟46‟‟E) and 5 km wide, and comprises two adjacent marine protected areas (MPAs) - the Maputaland MPA (Notice GN 404/86, 1986) and the St Lucia MPA (Notice P 35/79, 1979). The Maputaland MPA, situated in the north-east, has an area of 39 740 ha while the St Lucia MPA, situated in the south-west, is 44 280 ha (iSimangaliso Wetland Park Authority 2009). Physically the coastline within the park is sandy, relatively linear and receives little sediment input from non-marine and terrestrial sources due to the low numbers of rivers in the area (Cooper 1991; Riegl and Branch 1995; Schleyer 2000). The shoreline consists of numerous zeta-bays which have resulted from littoral drift dominating towards the north-east (Cooper 1991). The continental shelf along this section of the South African coast is narrow, 2–4 kms in width, and consists of a shelf break between depths of 45 m and 70 m (Roberts *et al.* 2006).

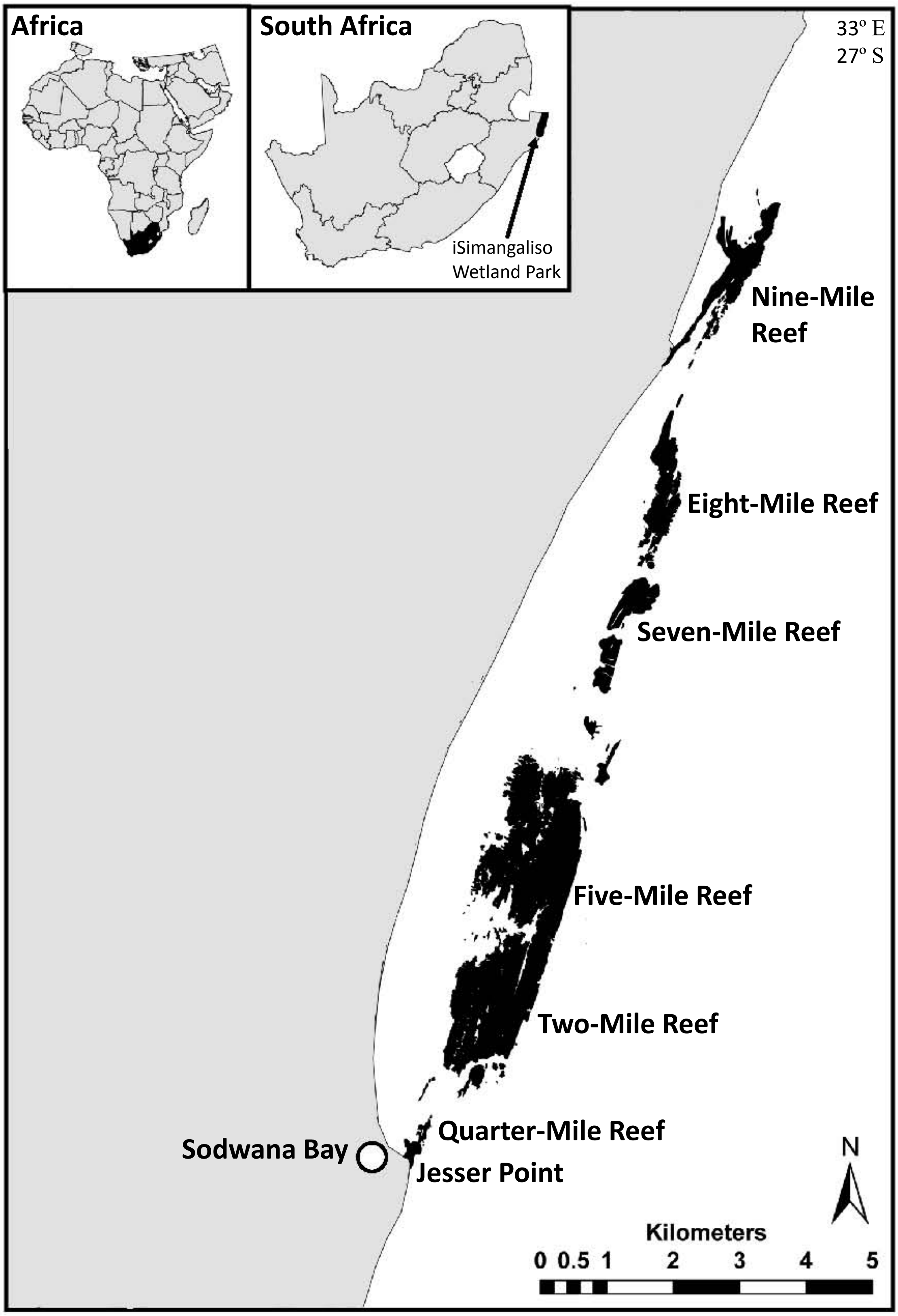


Figure 2.1: The Central Reef Complex of the iSimangaliso Wetland Park, KwaZulu-Natal, South Africa. Bathymetric data were obtained from the Marine Geoscience Unit of the Council for Geoscience, South Africa. Spatial reference system: Transverse Mercator (Central Meridian +27.00)

2.2 Climatology

The iSimangaliso Wetland Park falls between the tropical and subtropical climatic zones of Southern Africa resulting in a terrestrial region consisting of diverse woodland/ savanna biogeography (iSimangaliso Wetland Park Authority 2009).

Climatologically, the woodland/ savanna biogeographical zone is characterised by warm, moist summers and mild dry winters (iSimangaliso Wetland Park Authority 2009). Mean annual air temperature in the area exceeds 21°C with the coastal areas receiving relatively high precipitation of approximately 1300 mm per annum. Of the total rainfall, 60% falls during the Austral summer from December to March. The prevailing winds of the coastal region run parallel to the coast in either a south-westerly or north-easterly direction (iSimangaliso Wetland Park Authority 2009). During this study the prevailing wind force was from the south-west, attaining mean monthly velocities from 9.6 kts in August to 11.0 kts in November, and the north-east, attaining mean monthly velocities from 5.7 kts in February to 8.9 kts in July (Fig. 2.2). For the purposes of this study, wind data for the period June 2010 to June 2011 were extracted from the voluntary observing ships (VOS) database of the South African Data Centre for Oceanography (SADCO) for the area 27ºS–30ºS and 31ºE–34ºE.

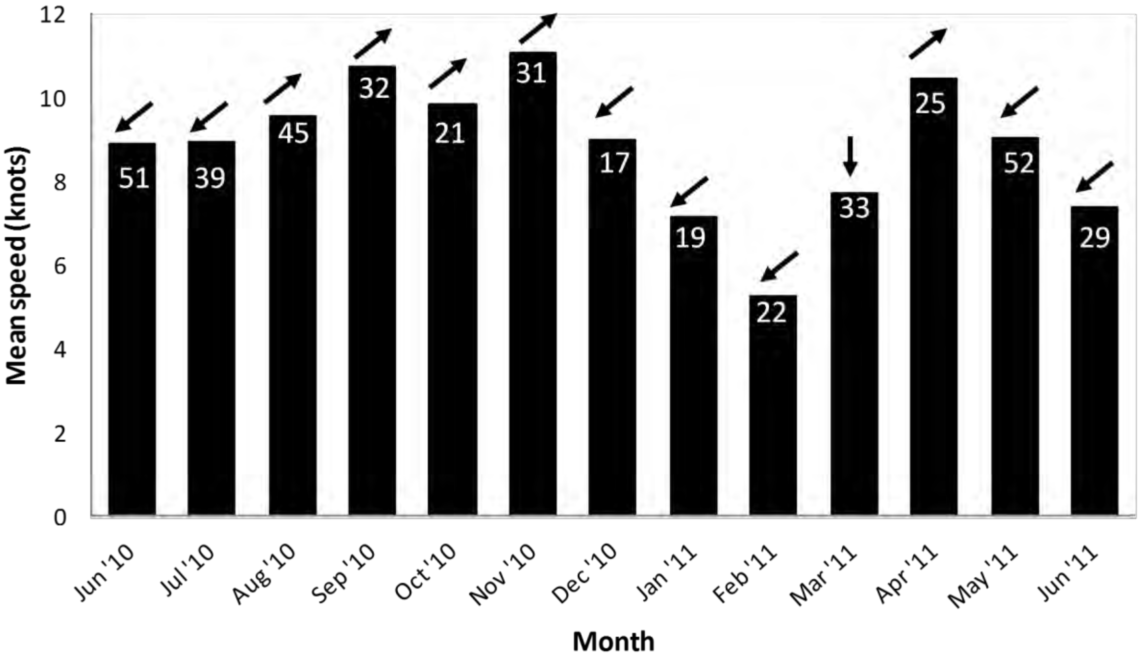


Figure 2.2: Mean monthly wind speed for the area 27ºS–30ºS and 31ºE–34ºE. Arrows depict modal wind direction for that month. Numbers within bars depict the number of observations for that month. Data are courtesy of South African Data Centre for Oceanography.

2.3 Oceanography

The most influential, large-scale oceanographic feature in the marine sector of the iSimangaliso Wetland Park is the Agulhas Current (Schumann 1988; Jury *et al.* 1993; Roberts *et al.* 2006). The Agulhas Current flows parallel to the coast from the north-east, originating off the Mozambique coast from a confluence of warm tropical waters passing into the Mozambique Channel. Attaining an offshore velocity of up to 3 $m.s^{-1}$, the current markedly affects the waters on the continental shelf with a reported shelf velocity of up to 1.4 $m.s^{-1}$(Ramsay 1994) and mean surface velocity of 0.27 $m.s^{-1}$ (Morris 2009). Although the Agulhas Current flows in a south-westerly direction, the shelf currents can be variable as a result of anti-cyclonic wind-driven circulation (Lutjeharms 2006). Acoustic Doppler Current Profiler data, from the continental shelf at the iSimangaliso Wetland Park, showed that southerly currents predominate while northerly flowing counter-currents can occur during periods of strong southerly winds (Morris 2009).

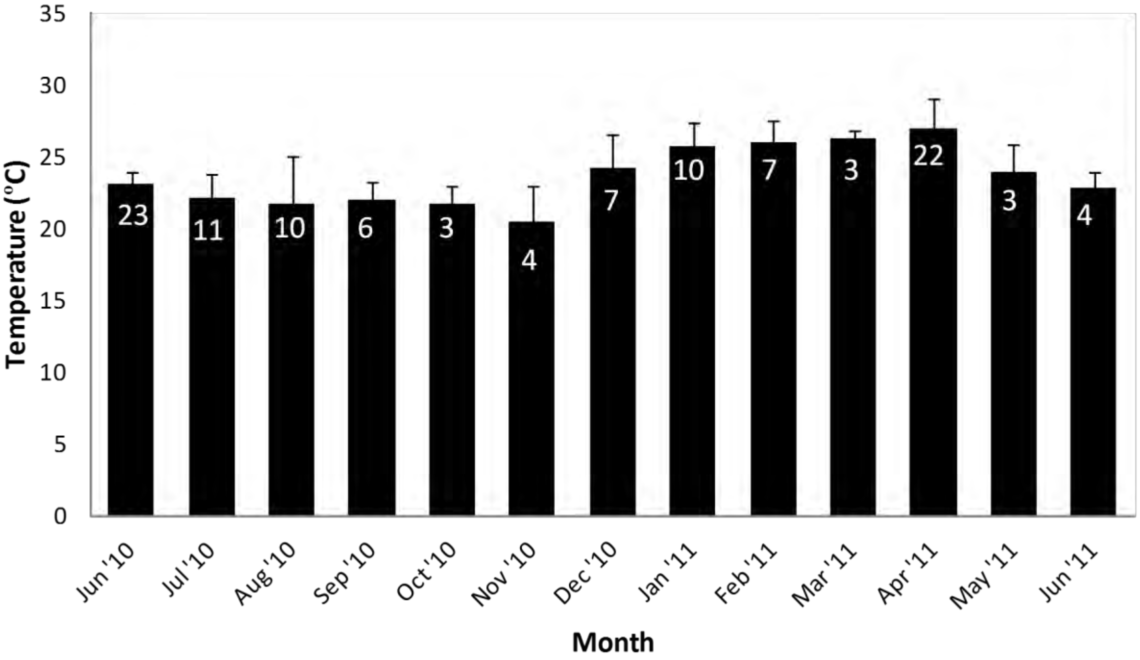


Figure 2.3: Mean monthly sea surface temperatures for the area 27ºS–30ºS and 31ºE–34ºE. Numbers within bars depict the number of observations for that month. Data are courtesy of South African Data Centre for Oceanography.

Mean seasonal sea surface temperature (SST) largely determined by the Agulhas Current has been reported to range from 22˚C in winter to 27˚C in summer (Smith *et al.* 1996). During this study mean monthly sea surface temperature was lowest during November, 20.4ºC, and highest during April, 27.0ºC (Fig. 2.3). For the purposes of this study temperature data for the period June 2010 to June 2011 were extracted from the voluntary observing ships (VOS) database of the South African Data Centre for Oceanography (SADCO) for the area 27ºS–30ºS and 31ºE–34ºE.

Salinity in the region is reported to range from 35.0 to 35.5‰ (Schumann 1988) and the tidal range in the area averages approximately 2 m during spring tides and approximately 1 m during neap tides (SA Navy 2010).

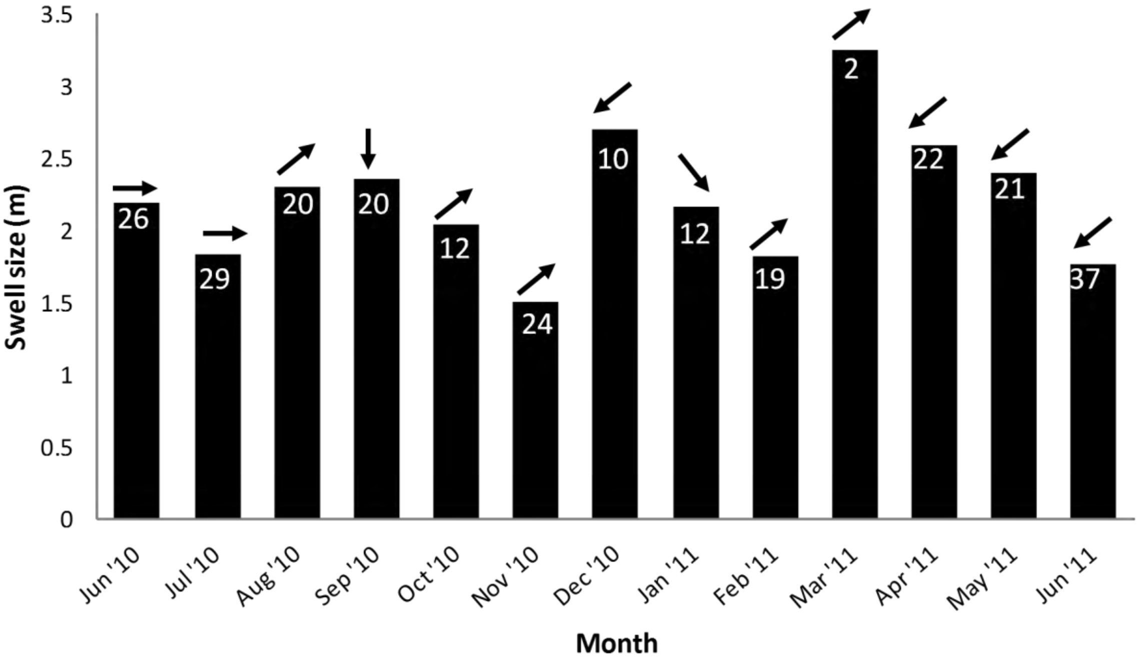


Figure 2.4: Mean monthly swell height for the area 27ºS–30ºS and 31ºE–34ºE. Arrows depict modal swell direction for that month. Numbers within bars depict the number of observations for that month. Data are courtesy of South African Data Centre for Oceanography.

Persistent high-energy waves and large-amplitude swells approach the coastline predominantly from the south-east as a result of strong southerly winds (Schleyer 2000). During this study modal swell direction was variable between months but occurred most frequently out of the south-east, attaining a maximum mean monthly swell height of 2.7 m during March, and the north-east, attaining a maximum mean monthly swell height of 3.3 m during December (Fig. 2.4). For the purposes of this study, swell data for the period June 2010 to June 2011 were extracted from the VOS database of SADCO for the area 27ºS–30ºS and 31ºE - 34ºE.

2.4 Subtidal geology

Two unique late-Pleistocene geological structures, sand dunes and river canyons, facilitate the establishment, and continued growth, of the iSimangaliso Wetland Park high-latitude coral reefs.

During the late-Pleistocene the glacial maximum low sea level was at least 100 m below the present level (Roberts *et al.* 2006). Prehistoric sand dunes, terrestrially exposed during glacial maximum lows, developed into large sandstone structures which now provide the sub-tidal substratum for the establishment of corals on the continental shelf (Ramsay 1994; Riegl and Branch 1995). Although the beachrock and aeolianite substrate enables coral establishment, continued growth is made possible by the presence of submarine canyons in the area which are a product of paleo-river outlets.

Seven submarine canyons intersect the continental shelf in the vicinity of the iSimangaliso Wetland Park. These canyons prevent the siltation of the shallow-water corals by capturing, and directing seaward, any sediment transported by the Agulhas Current (Ramsay 1994; Ramsay 1996). Shallower corals are therefore maintained free of sediment and subsequent siltation, enabling unhindered establishment and growth. These uniquely high-latitude coral reefs are associated with highly diverse biota from a wide range of taxa.

2.5 Marine biology

The iSimangaliso Wetland Park is South Africa‟s biggest marine protected area. Ocean-based research was initiated in the 1950s through preliminary collection and exploration (Heydorn 1972; Ballard 1973). Even at that early stage, Heydorn (1972, pg 20) referred to the coral reefs, of the then Tongaland, as an „endangered heritage‟. In 1980 the first directed, broad-scale investigations were conducted that included specific sections devoted to corals and marine fishes (Bruton and Cooper 1980). Awareness of to the great biodiversity of the region grew steadily and recreational SCUBA diving became increasingly popular. By 1987, 20 070 SCUBA dives were conducted from the primary launch site at Sodwana Bay (Ezemvelo KZN wildlife, as cited by Schleyer and Celliers 2003b). Diving intensity now fluctuates between 60 000 and 110 000 dives from the Sodwana Bay launch site anually (Chater *et al.* 1995; Floros 2010a). Despite its recreational popularity, it was not until the late 1980s and early

1990s that directed biological research commenced. Initial efforts in the 1990s were largely devoted to the study of the unique corals of the iSimangaliso Wetland Park (Ramsay and Mason 1990a, b; Riegl and Branch 1995; Riegl *et al.* 1995; Riegl 1996; Riegl and Riegl 1996; Benayahu and Schleyer 1998; Kruger *et al.* 1998).

2.5.1 Coral reefs

The coral reefs of the iSimangaliso Wetland Park constitute the southernmost coral reefs on the east African coast (Ramsay and Mason 1990b). These reefs occur in shallower, more photic waters at depths of 8 to 35 m and occur up to 155 km south from the Mozambique border (Schleyer 1999). Coral growth is determined by the variables: salinity, the nutrients NO_3 and PO_4, light penetration, temperature, and aragonite saturation. Because the South African reefs occur at a relatively high latitude, growth conditions, particularly in terms of the latter three variables, are considered only marginal (Schleyer 1999; Schleyer and Celliers 2003a, b). Colonisable substrate is patchy resulting in numerous small patch reefs which range from up to two kilometres in length and one kilometre in width and which do not form massive, accretive carbonate structures, but rather cover the sandstone substrate with approximately 30–40 cm coral cover (Ramsay and Mason 1990a; Schleyer 1999). Topography is largely governed by the shape of the prehistoric sand-dune substrate and any erosion that has occurred. Besides a few small, infrequent patches of reef, the total coral reef area within the iSimangaliso Wetland Park falls into one of three major reef complexes; the Northern Complex, the Central Complex, and the Southern Complex (Schleyer 1999). Despite their marginal nature, however, corals on these reefs provide up to 70% of living benthic cover. Soft corals, Alcyonacea (11 genera) comprising relatively few species, cover the majority of the available substrate, while the highly diverse hard corals, Scleractinia (46 genera), cover the remaining substrate (Jordan and Samways 2001; Schleyer and Celliers 2003b). Coral communities are predominantly Indo-Pacific and represent 133 known species (Pereira 2003; Celliers and Schleyer 2008). For a detailed community structure analysis of the South African coral reefs refer to Celliers and Schleyer (2008).

2.5.2 Ichthyofauna

The ichthyofauna of the iSimangaliso Wetland Park has been surveyed over three periods roughly 15 years apart, by Smith (1980), Chater *et al.* (1993, 1995) and Floros (2010a).

Between 1976 and 1979, Smith (1980) conducted three investigations into the ichthyofauna of the area. Collections during these expeditions were focussed on Two-Mile Reef at Sodwana Bay (Fig. 2.1), but ranged from Kosi Bay mouth in the north to Leadsman Shoal Reef, north of Leven Point, in the South, and were made from the intertidal zone, using snorkelling gear, to depths of 30 m using SCUBA. A large array of habitats was sampled, including estuary mouths, the surf zone, sand substrate, rock substrate and coral-reef substrate. Recreational angler catches were also included in the collection process. Fishes were found to be abundant in every niche except the midwater zone. Smith (1980) concluded that the clear, nutrient deficient waters resulted in small numbers of numerous species that are primarily dependent on the coral reefs for food and shelter. In total, the resulting paper lists 1198 species from 573 genera, and 150 families were listed that included many non-reef associated and pelagic species. Preliminary information on the biology of some of these species was conducted by van der Elst (1982). The important, exploratory surveys of Smith (1980) and van der Elst (1982), while providing check-lists of species occurring in the area, lacked the sampling design or replication required for the estimation of community dynamics such as richness and abundance.

Chater *et al.* (1993) used SCUBA surveys to visually sample the reef-associated fishes throughout the extent of the iSimangaliso Wetland Park. Sampling was unstructured and dives were conducted haphazardly without standardisation. Identifications were not recorded *in situ* but were recorded at post-dive debriefings. Their check-list included 73 families containing 25 elasmobranch species and 374 teleost species. The best represented family was the Labridae (wrasses) while other well represented families included the Serranidae (rock cods and goldies), Lutjanidae (snappers), Chaetodontidae (butterflyfishes), Carangidae (kingfishes), Pomacentridae (damselfishes) and Acanthuridae (surgeonfishes). Of the species recorded, 75% had Indo-Pacific distribution ranges, 21% were known only from the Indian Ocean, and 4% (7 species) were endemic to South Africa. Circum-African and circum-global species comprised species of sharks, billfishes, and tunas. Chater *et al.* (1993) made an attempt to quantify relative abundances using the categories rare, present, common, or shoaling. Chater *et al.* (1993) stated that they observed just over half of the species they expected to find based on the work of Smith (1980). The reason given for this difference was the diverse habitats sampled by Smith (1980) who did not restrict his research to offshore coral reef areas. Chater *et al.* (1993) further stated that 70 of the species found in their study were not recorded by Smith (1980). Chater *et al.* (1993) concluded that their list was comparable to lists from other tropical Indian Ocean areas such as the Seychelles Islands, although they noted that overall fish assemblage structures differed.

Chater *et al.* (1995) susbsequently conducted a more specific study, with a focus on replicated SCUBA surveys on Two-Mile Reef, in the Central Complex, and Leadsman Shoal Reef, in the Southern Complex, in an attempt to provide the first quantitative estimates of fish abundance. The work of Chater *et al.* (1995) was carried out on a selected list of 13 families of fishes, which were identified as being typical, representative species. Their study must be interpreted with some level of caution, however, as there are errors in their survey design which would have caused pseudoreplication and temporal autocorrelation. Chater *et al.* (1995) constructed permanent transect start and end points at the Leadsman Shoal and Two-Mile Reef. Each study location therefore consisted of only one true replicate. Furthermore the same transects were resampled in subsequent trips leading to temporal autocorrelation. Prior to each SCUBA transect, a transect line was fixed between permanent poles marking the transect start and end points. This could have disturbed the fish community, particularly in the case of shy or inquisitive species. Four to six divers would then, simultaneously, conduct each transect. Large numbers of divers are almost certain to have disturbed the ichthyofaunal community. The transects of Chater *et al.* (1995) also lacked adequate standardisation. Variable swimming speed, and hence transect time, could have resulted in increased data variability.

Survey design criticisms aside, Chater *et al.* (1995) noted that counts of muraenids, large serranids, sparids, lethrinids, carangids, sphyraenids and scombrids were low. They found that the wrasse *Thalassoma hebraicum* was the most abundant species on both reefs at a mean abundance of 40.1 fish.1000 m^{-2} on Leadsman Shoal and 24.2 fish.1000 m^{-2} on Two-Mile Reef. Other regularly recorded species included *Chaetodon madagaskariensis*, *C. unimaculatus*, *Forcipiger flavissimus, Bodianus diana, B. bilunulatus*, *Lutjanus bohar, L. russelli, Amphiprion allardi,* and *Acanthurus leucosternon*. Differences in mean abundance between Two-Mile Reef and Leadsman Shoal Reef were observed; however, there were no apparent reasons for these differences. The authors recommended detailed investigation into the habitat complexity of each reef and species specific requirements in terms of habitat. Chater *et al.* (1995) finally concluded that the diving activity on Two-Mile Reef (105 267 dives in 1992), at that stage had no effect on fish abundances when compared with Leadsman

Shoal. Some 15 years later, Floros (2010a) undertook a more holistic, truly replicated investigation of the ichthyofaunal community dynamics in the region.

Floros (2010a) used a stratified, random sampling design to conduct replicated 60 minute roving diver surveys. Floros (2010a) conducted a mean of 11 replicates per study site, but restricted replicates to the dominant benthic community type at a depth range of between 10 and 16 m. A total of 284 species belonging to 50 families were recorded on South African and nearby Mozambican coral reefs. Floros (2010a) concluded that the species richness observed in South Africa was comparable to other reefs in the Western Indian Ocean. Six families contributed more than 50% towards fish community composition. These were the Labridae, Acanthuridae, Chaetodontidae, Lutjanidae, Pomacentridae and Serranidae. Floros (2010a) constructed a list of indicator species for which biomass was quantified for comparison between reefs.

Ichthyofaunal research in the iSimangaliso Wetland Park has therefore been staggered and non-continuous. The work of Floros (2010a), although initiated in 2007, provides the only non-seasonal, baseline information on the fish assemblages of the region associated with only a narrow depth range and a single benthic community.

2.6 Reef protection status and usage

The adjacent MPAs of the iSimangaliso Wetland Park are managed by dual authorities: Ezemvelo KwaZulu-Natal Wildlife (EKZNW) and the iSimangaliso Wetland Park authority (Lemm and Attwood 2003). Legislatively, management conducted by these authorities must adhere to the South African Marine Living Resources Act (MLRA, Act 1998).

Various reefs in the marine sector of the iSimangaliso Wetland Park have been zoned into either restricted-use or sanctuary zones. Under the MLRA, restricted-use zones permit unrestricted shore-based angling and restricted boat-based angling for a selected list of pelagic fishes (MLRA Section 3.1 (G) Regulation R1429). Restricted zones also allow recreational SCUBA diving, provided a code of conduct is adhered to by all divers and dive concession holders. Sanctuary zones prohibit all human activities and are considered strict no-take zones.

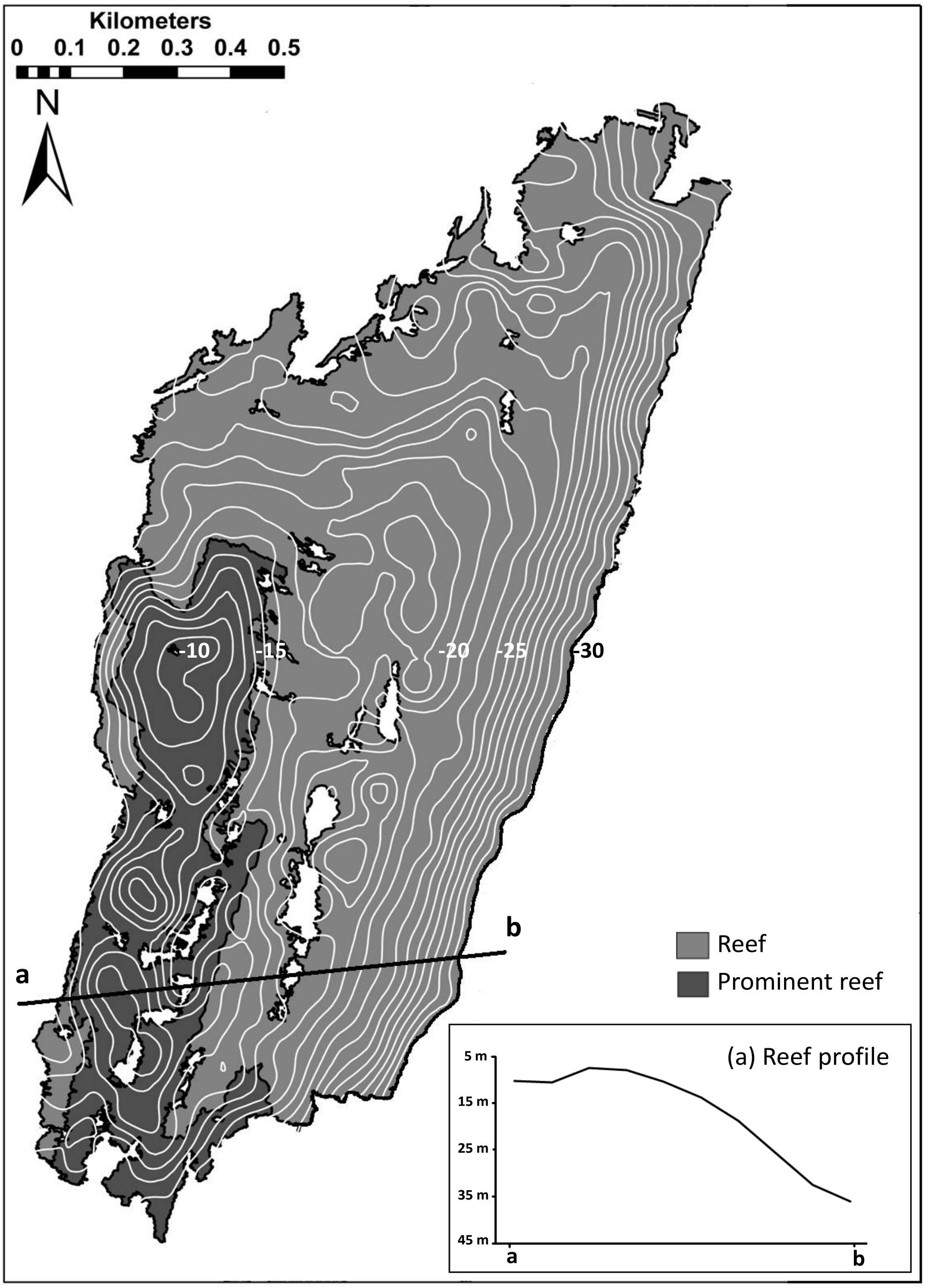


Figure 2.5: Two-Mile Reef in the Central Reef Complex. Inset (a) is a cross-section of the reef along line a–b. Depth contour and geological data were obtained from the Marine Geoscience Unit of the Council for Geoscience, South Africa. Spatial reference system: Transverse Mercator (Central Meridian +27.00).

The majority of reefs in the northern complex and all reefs in the southern complex have been zoned as sanctuary areas (Table 2.1). The remaining areas, including all reefs in the central complex, form a restricted zone. Two-Mile Reef (Fig. 2.5), in the central reef complex, is an exception to this in that only recreational SCUBA diving is permitted on the reef while restricted fishing may occur only at depths >30 m (Table 2.1).

Of restricted-use reefs, the Central Reef Complex is the most accessible via access points at Sodwana Bay. All other reefs in restricted-use zones are only accessible via private access points or lengthy boat-based journeys and therefore receive comparatively low usage pressure. The easily accessible Central Reef Complex receives the greatest anthropogenic usage in the form of recreational fishing, spearfishing and SCUBA diving (Schleyer 1999).

Table 2.1: Marine Protected Area zonation of the major coral reefs in the iSimangaliso Wetland Park marine sector (Table compiled from information supplied by Schleyer (1999) and KwaZulu-Natal Wildlife Nature Conservation Services marine zone management plan (2003)). Terminology is in accordance with the South African Marine Living Resources Act 1998.

Reef Complex	Reef	Permitted activities
Northern	North of Saxon Rocks	None
	South of Saxon Rocks	Recreational angling/spearfishing, SCUBA diving
Central	9-Mile Reef	Recreational angling/spearfishing, SCUBA diving
	7-Mile Reef	Recreational angling/spearfishing, SCUBA diving
	5-Mile Reef	Recreational angling/spearfishing, SCUBA diving
	2-Mile Reef	SCUBA diving, recreational angling/spearfishing at depth >30 m
Southern	Red Sands Reef	None
	Leadsman Shoal	None

2.7 Study site

Biological information on the Central Reef Complex is of immediate importance to inform management through monitoring anthropogenic usage. The Central Reef Complex was therefore selected as the focus area for this thesis. The complex lies adjacent to Lake Sibaya in the north and extends south to Jesser Point, an aeolianite shelf at Sodwana Bay (Anderson *et al.* 2005) (Fig. 2.1). The central complex consists of five major reefs either semi-detached from, or adjacent to, each other. From the north these reefs are Nine-Mile Reef, Seven-Mile

Reef, Five-Mile Reef, Two-Mile Reef and Quarter-Mile Reef – all named after their distance from the boat launch site at Jesser Point. The smallest discrete reef in the complex is Seven-Mile Reef, while the largest reef, semi-detached from Quarter-Mile and Five-Mile Reefs, is Two-Mile Reef. Although semi-detached from one another, each reef within the Central Complex is a discrete reef structure possessing unique physical features (Schleyer 2000, Floros 2010a). As ichthyofaunal assemblage structure across depths and habitats was a study objective, a single representative reef had to be selected for investigation. To be representative of the Central Reef Complex, and to ensure consistent coral recruitment, the selected reef could not be small and isolated (Ault and Johnson 1998). The largest discrete patch reef within the Central Reef Complex, Two-Mile Reef, was therefore selected as the study site.

Table 2.2: Summary of Ramsay and Mason's (1990a) zonation of Two-Mile Reef in terms of physiographic and biological criteria.

Zone	Physical Criteria	Coral Biological Criteria
1	Leeward slope: Landward margin of reef with occasional spur and groove formations; depth 15–21 m	Poor coral growth, the genus *Pocillopora* being the most abundant hard coral.
2	Leeward reef: Flat, low-relief zone on the landward side; depth12–14 m	Mixed coral community of low cover (10%) in most of the zone
3	Reef crest: The shallow, high energy reef top of high-relief with gullies, pinnacles and overhangs; depth 9–13 m	Abundant coral growth, particularly soft corals of the genera *Lobophytum* and *Sinularia*
4	Gully zone: offshore of zone 3 and 20–50m wide. Depth 13 m	None
5	Fore reef: Flat, low relief fore reef with regular narrow, sediment-filled gullies. Depth 13–20 m.	Mixed coral community in which the genus *Sarcophyton* is conspicuous.
6	Fore reef: 80–100 m wide with longshore, sediment-filled gullies.	None
7	Fore-reef zone of moderate relief (1.5 m) but with no true inter-reef gullies; depth 21–28m	Mixed coral community which includes the genera *Lobophyllia* and *Leptogorgia*
8	Flat, deep reef-front grading into sediment; depth 29–34 m	Numerous sponges and gorgonians with few corals

Two-Mile Reef (27° 30.5" S; 32° 42.2" E) is the largest discrete patch reef in the Central Reef Complex. Spanning 2.1 km in length and 0.9 km in width, the reef lies approximately 1 km offshore and runs parallel to the coast. Depth on the reef ranges between 6 m and 34 m and consists of a wide range of habitats such as shallow pinnacles (6–10 m), extensive deep subtidal reef flats (14–18 m) and a gently sloping seaward edge (24–27 m) (Ramsay and Mason 1990a; Schleyer and Celliers 2005; Celliers and Schleyer 2008; Floros 2010a) (Table 2.2, Fig. 2.5). The reef has a short, gentle leeward-reef slope, a flat reef top, atypical of a reef crest, and a large, gradual fore-reef slope (Table 2.2, Fig. 2.5). The euphotic zone along this section of coast has been reported to occur at a depth of approximately 30 m (Ramsay and Mason 1990a. Full coral cover on Two-Mile Reef extends to approximately 25 m after which gorgonian and sponge communities exist, interspersed with only scattered coral colonies (Ramsay and Mason 1990a) (Table 2.2, Fig. 2.2).

Chapter 3:

Designing a suitable, standardised Underwater Visual Census protocol for surveying epibenthic coral reef fishes in the Western Indian Ocean

3.1 Introduction

Coral-reef fish assemblages are abundant, rich, diverse, and variable in comparison to other vertebrates (Hughes 1994). It is widely accepted that the most appropriate way to survey coral-reef fishes is by fisheries-independent, non-destructive, *in situ* underwater visual census (UVC) (Samoilys and Carlos 2000; Langlois *et al.* 2010; Pelletier *et al.* 2011). Under ideal circumstances, if the study area is known and there are clear objectives, it should be straight forward to select an appropriate, well-defined, rigorously-tested underwater UVC sampling protocol. In reality, no standardised protocols have been recommended.

Despite six decades of conducting UVCs, no guidelines and/or protocols exist that outline the most appropriate approaches to implementing general UVCs, or which outline the most appropriate techniques to specific research circumstances. The choice of which technique to employ, and the design of the survey protocol to implement, has therefore been researcher-specific (Bortone and Kimmel 1991). It is known that different techniques applied to the same community can yield quite different results (Branden *et al.* 1986; Davis and Anderson 1989; Connell and Kingsford 1998; Kulbicki and Sarramégna 1999; Willis *et al.* 2000) as variations in techniques, such as different survey dimensions and observer behaviour, can greatly affect counts (Sale and Sharp 1983; Lincoln Smith 1988). Reduced intra- and inter-study comparability is therefore a common criticism of UVCs (Watson *et al.* 2005). While flexibility of implementation is a strength of UVCs, directly comparing data obtained by different and/ or variable methods is difficult, if not impossible (Bortone and Kimmel 1991; Colvocoresses and Acosta 2007) because biases associated with different UVC methods remain poorly understood (Edgar 2004). An urgent need for the standardisation of techniques and approaches to conducting UVCs, both between and within studies, has been identified (Samoilys and Carlos 2000; Bennett 2008; Currie 2005). One approach would be to implement regulatory measures at two levels. These include (1) the standardisation of approaches to UVCs in general resulting in greater inter-study comparability, and (2) the development and testing of specific protocols to decide what techniques and methods are best suited to specific research questions, resulting in greater intra-study comparability.

This chapter aims to directly address the issue of inter- and intra-study standardisation. While focus is placed on developing a protocol specific to this study"s aims, the protocol is general and applicable across all UVCs. The objectives of the present chapter, therefore, were to (1) provide an example of strict within-study standardisation, based on meta-analysis of the literature, for implementation as a specific standardised protocol, (2) provide sufficient background information such that future between-study standardisation and the development of future protocols is facilitated, and (3) provide clarity on a number of definitions and terms pertaining to UVCs which were not addressed in Chapter 1. The standardised protocol outlined here forms the general materials and methods for this thesis, but some additional, chapter-specific materials and methods are outlined where necessary.

3.2 Survey technique

Selecting appropriate survey methods and an appropriate study site are not mutually independent decisions. Most often, suitable survey techniques must be based on the characteristics of the study site and the ichthyofaunal community under investigation. It is therefore necessary to identify suitable methods *a priori,* so that sampling is standardised for the duration of a short-term study, or a long-term monitoring program (Sutherland 1996; ICES 2006).

The most appropriate UVC technique for efficiently surveying epibenthic reef fishes is the strip transect technique because a relatively large area can be easily surveyed, transects are easily implemented and replicated, transect parameters can be clearly defined, and the limitations of the technique are well understood. The measures of standardisation described below are therefore in the context of surveying epibenthic reef fishes using the strip transecting.

3.3 Standardisation

Failure to standardise is failure to hold study conditions constant and should, at all costs, be avoided (Winer *et al.* 1991) in an effort to limit the error resulting from bias and variability (Harvey *et al.* 2002b). Despite the obvious reasons for requiring UVC standardisation, standardisation amongst studies is uncommon. Bortone and Kimmel (1991) have assigned guidelines for a „bare minimum‟ level of standardisation where (1) each technique must be described clearly and referenced so that it can be duplicated to aid future research and facilitate evaluation of appropriateness, (2) the assumptions associated with data collection should be clearly stated, and (3) the perceived beneficial and detrimental features of the techniques used, and recommendations for their modification, should be identified.

3.3.1 Survey design

Survey research is most often associated with a wide range of potential sources of error (Winer *et al.* 1991) all of which have the potential to discredit entire research programs (Chater *et al.* 1995). UVC surveys are no different as surveys are conducted in an aquatic medium often many kilometres from land, requiring extensive logistical planning. In UVC survey design, researchers must be cognisant of sampling bias, environmental variability and biotic and abiotic interactions (Ludwig and Reynolds 1988). As biotic and abiotic factors are complex, and often interrelated, it is impossible for a researcher to control all the attributes that may influence the study outcome (Cox 1958). Prior to conducting a study, it is necessary to decide which factors are to be accounted for in the study design process, i.e. stratification, which factors can be „neutralised‟ by varying at random, i.e. replication and randomisation, and which factors will be measured as supplementary variables (Kempthorne 1952).

Stratification

Stratification refers to the spatial and/or temporal subdivision of sampling according to physical and/or biological characteristics (Vos *et al.* 2000). In UVC surveys the stratification of reefs into sampling units/strata with similar characteristics assists in removing spatial and temporal irregularities, as the variability within each stratum is likely to be lower than the study area as a whole (Green 1979; Bortone and Kimmel 1991). Stratification also provides better information on which causal factors may be structuring reef communities, and allows some scope for the extrapolation of conclusions to similar strata in other areas (Green 1979; Lohr 1999).

Coral reefs usually comprise a variety of natural materials including living material (hard corals, soft corals, sponges, invertebrates and algae) and non-living materials (rocks, sand and other geological formations). All these features contribute to reef substrate heterogeneity and physical reef relief (Bortone and Kimmel 1991) and can drive structural changes to reef fish communities (Choat and Bellwood 1991). Stratification must therefore be implemented at both spatial and temporal levels. Within a coral-reef perspective, spatial characteristics are stratified by reefs, reef zones, depth and substrate while temporal characteristics are stratified by season and time of day.

Reefs

Floros (2010a), conducting a more generalised study on a number of reefs in Mozambique and South Africa, showed that different ichthyofaunal communities can exist on different discrete reefs. To avoid these differences a single, large and representative patch reef in the area, Two-Mile Reef, was selected for investigation.

Reef zones

Fish community structures differ between different reef zones such as the fore- and leeward-reef slopes, and the reef crest (Choat and Bellwood 1991). Specifically for Two-Mile Reef, the fore- and leeward-reef zones comprise different benthic communities (Ramsay and Mason 1990a), a factor which could also affect fish community structures (Bell and Galzin 1984; Carpenter *et al.* 1981; Chabanet *et al.* 1997; Feary *et al.* 2007). To avoid confounding in this regard, sampling was conducted on the reef crest and leeward-reef slope only.

Depth

Depth has been shown to be a major factor influencing the fish community structures (Thresher 1983; Friedlander *et al.* 2003). On Two-Mile Reef, increasing depth has previously been associated with a change in benthic community structure (Celliers and Schleyer 2008) and decreasing algal cover (Anderson *et al.* 2005). Stratification must therefore be within a narrow depth range, or between contrasting depth ranges to address specific hypotheses.

Substrate

Substrate is a factor which has been associated with changes to fish community structures (Guidetti 2000; Gratwicke and Speight 2005). As this study investigated reef-associated species, only purely reef substrate was sampled.

Season

The effect of season has been ignored in studies investigating low-latitude coral reef investigations, possibly because it is widely accepted that the negligible seasonal fluctuations in physical and biological variables at equatorial latitudes are unlikely to drive change (Freeman 2005). One of the aims of the present study, however, was to determine if seasonal changes to fish assemblages occurred on high-latitude reefs. Seasons should therefore be discrete and contrasting. This study was therefore conducted in mid-winter and mid-summer.

Time of day

Colton and Alevizon (1981) and Götz (2006) showed that the time of day at which surveys are conducted can influence the fish community surveyed. The South African coast is exposed and variable, and typically has rougher sea conditions in the afternoon. The time of day that surveys were conducted was therefore restricted to a narrow, mid-morning to mid-day time window (09:00–13:00). It was envisaged that this time window was late enough to ensure sufficient ambient light, early enough to avoid the rough afternoon sea conditions, and narrow enough to reduce variability in fish behaviour associated with crepuscular activity (Davis and Anderson 1989).

Replication

Bortone and Kimmel (1991) note that due to the complex nature of environmental parameters, their inherent natural variability and the large number of potential interactions, the data obtained from surveys that have not been carefully designed and adequately replicated should be interpreted with caution. There are two primary benefits to replication. It provides both an estimate of error and increases the statistical power with which effects can be detected (Bros and Cowell 1987; Thompson and Mapstone 2002; Cohen 1973). In general the frequency of independent replications is proportional to the precision with which dependent variable effects can be estimated (Winer *et al.* 1991). There is, however, a point where the cost of conducting more replicates outweighs the value of minor improvements to precision (Samoilys and Carlos 2000). It is important to first determine the minimum sample size necessary for achieving a desired level of statistical power such that resources are not wasted on over-sampling (Samoilys and Carlos 2000).

Where studies have not previously been conducted, statistical methods such as power analysis cannot be employed to determine the minimum recommended replication. Bortone and Kimmel (1991) note that, when conducting UVC studies, a large number of samples measuring the variables of interest must be collected, such that variability can be statistically accounted for. This is an inefficient approach. It is perhaps more appropriate to examine the literature of studies conducted on similarly diverse systems such that a „best guess‟ of the appropriate sample size for a study can be determined. From the results obtained using a „best guess‟ sample size, the actual sample size required to achieve a desired power level can be determined.

Several studies have determined the appropriate number of samples that should be conducted in conducting coral-reef fish surveys (Table 3.1).

Table 3.1: Summary of studies that determined the minimum number of replicates appropriate for surveying coral-reef fishes. Studies are ordered by the number of replicates recommended. RSA = Republic of South Africa.

Authors	Locality	Survey technique	Recommended replicates
Sale and Douglas (1981)	E. Australia	Roving Diver	4
Bortone *et al*. (1986); Jones and Thompson (1978); Kimmel (1985)	Various	Various	<8
Floros (2010b)	Mozambique & RSA	Roving diver	10
Harmelin-Vivien *et al.* (1985)	Various	Various	12
Samoilys and Carlos (2000)	E. Australia & Fiji	Strip-transects & Point counts	10–15
Floros (2010c)	Mozambique & RSA	Point counts	18
Bortone *et al*. (1989)	Puerto-Rico	Various	16–32

Andrew and Mapstone (1987) state that the number of samples required is often specific to each situation and will probably vary between situations. If variation in the factor of interest is low, as few as three to ten samples may be required, but if variation is high, as many as 50 may be necessary (Bortone and Kimmel 1991). Despite variation in the literature, Samoilys and Carlos (2000) recommended 10–15 replicates when conducting transects and point counts on coral-reef fishes. Based on these findings, and the large variability in minimum sample sizes recommended in the literature, it was decided that the present study would implement the maximum number of replicates logistically feasible, 15 replicates per stratum, in the first phase of the study.

Randomisation and sample site allocation

Fisher (1951) was the first to suggest that randomisation could be used as a method of experimental control. Adequate randomisation is not only important for the avoidance of experimental design flaws such as pseudo-replication and autocorrelation (Bennett 2008), but is important for neutralising those factors which cannot be stratified. A truly random sample must have two characteristics. The sample must be drawn in such a way that all elements in the population have an equal and constant chance of being drawn on all draws, and all possible samples have an equal, i.e. fixed and determinable, chance of being drawn (Winer *et al.* 1991). In UVC this applies to the selection of sites to be sampled from a list of all appropriate sites.

Sampling was therefore conducted using a stratified-random sampling design (Colvocoresses and Acosta 2007) without replacement (Bennett 2008). Bathymetric and geological data which had been digitally georeferenced, using Geographical Information Systems (GIS), were obtained from the Marine Geoscience Unit of the Council for Geoscience (Durban, South Africa). Using the appropriate stratification prerequisites in ArcGIS 10 (© Environmental Systems Research Institute), a grid overlay with cells 50 m × 50 m was used to identify all possible sampling sites meeting the necessary criteria, where cell size was based on transect length. Sampling sites were chosen by randomly drawing geographic coordinate pairs. Sites were drawn without replacement within and between all levels of stratification such that pseudoreplication, temporal autocorrelation, and the possible effects of divers repeatedly sampling individual sites, were avoided (Thompson and Mapstone 2002; Fowler 1987; Kulbicki 1998). To avoid spatial autocorrelation, no adjacent sites were sampled in any phase of the study (Bennett 2008). Additional co-ordinate pairs were generated as a contingency against incomplete samples or particular sites not falling within the required depth range. Post-site selection, coordinates were entered into a handheld Global Positioning System (GPS) (etrex, © Garmin) and their depths ground-truthed using a boat-based echo-sounder.

Supplementary variables

To understand and ultimately manage a dynamic and variable environmental system, it is important to understand the specific controlling or influencing factors (Patton *et al.*1985). Abiotic and biotic parameters were quanitified as supplementary variables, *in situ,* at the completion of a dive to minimise pre-transect diver disturbance to fishes. To prevent longer-term changes in supplementary variables confounding the primary comparisons of interest (Underwood 1993), each sampling event was conducted within a 6-day period; this was the shortest time period possible based on the number of available divers and the required replication.

Topographic complexity, substrate type, and current, were determined by each diver conducting transects, and the mean value for each transect calculated. Visibility, water temperature and turbidity were measured, and the time of day and tide recorded.

Topographic complexity

Clua *et al.* (2006) stated that reef topographic characteristics, referred to here as topographic complexity, are usually interrelated and contribute towards general habitat heterogeneity of reef. The topographic characteristics of a reef have been referred to using a number of terms, including habitat complexity (Grigg 1994; Caley and St John 1996; Beukers and Jones 1998), shelter availability (Connell and Kingsford 1998; Friedlander and Parrish 1998) rugosity (Floros 2010a), and profile (Bennett 2008). These terms are used interchangeably between studies and are often quantified using unrelated techniques. For example, compare „rugosity" as defined by Luckhurst and Luckhurst (1978), Chandler *et al.* (1985), McClanahan (1994), Bennett (2008), and Floros (2010a). As no current metrics of topographic complexity can provide all the information a researcher requires regarding the structural architecture of a survey reef (Clua *et al.* 2006), three pertinent metrics of topographic complexity were selected for use in this study. Two of these metrics, profile and rugosity, were selected because they are simple and can be efficiently to quantified in comparison to techniques such as the chain-link method (e.g. Almany 2004), and each provides slightly different information on reef structure. The third metric, termed habitat complexity, is a qualitative metric for which a specific set of definitions was developed in this study.

To calculate profile and rugosity, the depth of each transect was recorded at the start of that transect and at each 10 m interval. Profile was calculated as the sum of squares of the differences between consecutive depth readings (McCormick 1994). Differences between consecutive depth readings were calculated as x_1-x_0, x_2-$x_{1, \ldots,}$ x_{max} - x_{max-1} . This method was able to distinguish between a level sea floor and one that deviates from level, thus providing information on the angle of the average slope. The limitation of profile is that the metric does not enable a distinction between a sloping seafloor with even surface and an equally sloping seafloor with uneven surface. Therefore rugosity was used as a second measure of topographic complexity.

Rugosity was calculated as the absolute value of the standard deviation of the differences between consecutive depth readings (Bennett 2008). Differences between consecutive depths were calculated as indicated for profile. Using this metric, an even seafloor of any slope will provide a lower rugosity value than an uneven, level seafloor. The magnitude of unevenness is reflected in the calculated value. Neither rugosity nor profile, however, provides information on the texture or detail of the reef substrate and/or benthic community.

An area which produces high values of profile and rugosity can in reality provide homogenous habitat. For example, a reef with a steep reef slope and the presence of large, rounded boulders will produce high profile and rugosity values, but will provide less surface area and habitat than a reef with equally high profile and rugosity, but comprised of jagged boulders and detailed coral heads. The qualitative metric „habitat complexity‟ was therefore developed as an easily implementable description of the structural detail of the sites surveyed. Although a coarse version of the metric has been implemented by Polunin and Roberts (1993), the habitat complexity metric is essentially a visual index of the intricacy and detail of a particular site, independent of profile or rugosity. Habitat complexity for this study was categorised qualitatively as either highly complex, complex, semi-complex, homogenous, or highly homogenous. These categories are summarised in Table 3.2. Photographic examples are provided in Appendix 1.

Table 3.2: Categories for the assessment of habitat complexity and substrate type at each study site

Level	Profile: Habitat complexity	Substrate type
1	Highly complex: Habitat highly intricate; many nooks, crannies, caves and crevasses. Deep in structural detail. Structurally complex corals abundant.	Reef: Entirely rock/ coral reef.
2	Complex: Many nooks, crannies, caves and crevasses. Moderately deep in structural detail. Structurally complex corals abundant, but interspersed with less complex patches.	Reef-Sand: Predominantly of rock/ coral reef interspersed with sandy areas.
3	Semi-complex: A detailed area interspersed with large homogenous substrate or barren areas. Nooks, crannies, caves and crevasses shallow. Habitat refuge for fishes still abundant.	Reef/Sand: Approximately 50% rock/ coral and 50% sand.
4	Homogenous: Area flat, barren, and almost void of complex coral structures. Rocks/corals smooth and rounded, regardless of rugosity, providing only low levels of habitat refuge for fishes.	Sand-Reef: Sand interspersed with rock/ coral reef areas.
5	Highly homogenous: Area flat, barren, and void of complex coral structures. If area has a high rugosity, rocks/corals smooth and rounded providing almost no habitat refuge for fishes. Generally interspersed with large dead/ sandy patches.	Sand: Entirely sand.

Substrate

Substrate type was assessed qualitatively. Visual assessment of the substrate present at each site was made by assigning the observed substrate to one of five categories; Reef, Reef-Sand, Reef/Sand, Sand-Reef or Sand (Table 3.2). Photographic examples are provided in Appendix 1.

Current

Current speed was estimated by each diver. Divers noted the time taken for the current to transport the diver a distance of 1 m. The mean of both stop-watch based estimates was accepted as the current speed for that sample and extrapolated to $m.s^{-1}$. Current direction at the sampling depth was determined visually by each diver.

Visibility

Horizontal visibility was measured by divers at the sampling depth. Each diver pair would swim away from each other along the extended transect line and note the distance at which the divers could no longer visually distinguish one another.

Water temperature

Water temperature at the sampling depth was measured with Gekko Dive Computers (Suunto ©).

Turbidity

A water sample for each transect was collected in an 80 ml plastic jar at the sample depth. Water samples were frozen as soon as possible for later laboratory analysis. Samples were defrosted and turbidity measured using an H198703 portable turbidity meter (Hanna Instruments ©).

Time of day

Although the time of day for conducting surveys was restricted to a narrow time window, the time at which each transect was conducted was recorded to confirm that any observed differences were not a product of changes in time. The start time of each transect was recorded by divers once the transect start point had been established.

3.3.2 Survey technique and fish assemblage parameters

Survey technique parameters

Transect length

Stewart-Oaten *et al.* (1996) states that the spatial scale of sampling should be related to the range of movement of the study species. To minimise spatial and temporal variability, particularly when relying on visual distance estimates, it is preferable that replicates are spatially reasonable (Tessier *et al.* 2005). Reef fishes are resident, with home ranges varying from <10 m^2 to >100 m^2 (Allen 1991). Large areas cannot be accurately surveyed using visual transects. Transect length was therefore set to 50 m as, after reconnaissance diving, it was envisaged that 50 m would be large enough to capture most species, but small enough to enable manageable sampling. Standardisation of transect length was achieved by divers carrying a 50 m shot line that was extended from the transect start point. Extending a shot line while conducting a transect would exclude diver disturbance effects which could have been generated if a transect line were constructed prior to conducting each replicate (Edgar 2004).

Swimming speed

Lincoln Smith (1988), Samoilys and Carlos (2000), Lacour *et al.* (2001) and Tessier *et al.* (2005) have each recommended a swimming speed appropriate for underwater transects, noting that slower swimming speeds enable more accurate and precise visual counts and that swimming speed is a factor that should be standardised across studies. In contrast, swimming speed should not be so slow that dive time is extended beyond reasonable efficiency thereby decreasing the number of replicates divers can conduct in a given day due to residual nitrogen loading (Watson *et al.* 1995). In line with the conclusions of Lincoln Smith (1988), Samoilys and Carlos (2000), Lacour *et al.* (2001); Tessier *et al.* (2005) and Watson *et al.*(1995), swimming speed was set to 6.25 m.min^{-1} translating into a total time of 8 min.transect^{-1}. Reconnaissance diving confirmed that this speed was slow enough to incorporate sedentary species, but fast enough to ensure observations of more mobile species. Swimming speed was standardised in that divers monitored the time relative to the distance covered along the transect, on dive watches.

Fish assemblage parameters

Fish assemblage parameters must be established to suit the research objectives of a study and to reduce any unnecessary data variability (Samoilys and Carlos 2000). This study established standardised fish assemblage parameters by excluding cryptic species, setting a minimum fish size, assigning species groups, reducing double counts, and by temporally stratifying counts.

Cryptic species

Brock (1982), Ackerman and Bellwood (2000), Willis (2001), and Lechanteur and Griffiths (2002) state that even when meticulous searches of small areas are implemented, cryptic fishes can be underestimated by up to an order of magnitude. As the focus of the present study was on epibenthic reef fishes, cryptic species, which are commonly benthic species (Lechanteur and Griffiths 2002), were excluded.

Minimum size

The difficulties associated with identifying and counting small fishes could dramatically increase variability. It is therefore necessary to set a minimum size limit to the fishes observed (Samoilys and Carlos 2000). To prevent excluding important species, the size limit should not exclude species which only grow to small sizes. Based on observations made during reconnaissance dives, minimum fish size for the present study was set to 4 cm – small enough to reduce variability, but large enough to include families such as the Apogonidae and the Pomacentridae, which do not attain large sizes (Smith and Heemstra 2003).

Assignment of species groups

Some species require detailed taxonomic assessment for identification to species level. As detailed taxonomic assessment is not possible when conducting UVCs, morphologically similar species were assigned to species groups (Alevizon and Brooks 1975). These groups were assigned either because species were indistinguishable during visual transect identifications, which needed to occur instantaneously, or their defining characteristics were indistinguishable during photographic and video identifications. A total of eight species groups consisting of 24 species were assigned. All species in all groups were observed in the study area in varying proportions at some point during reconnaissance dives or during the study. The same species groups were applied to all transect types. Species groups were treated as species in subsequent analyses (Alevizon and Brooks 1975; Samoilys and Carlos 2000).

Reduction of double counts

The problems associated with enumerating individuals more than once have been documented (DeMartini 1993; Colton and Swearer 2010). To reduce the occurrence of double counts, fishes passing from behind divers were excluded from the counts of all transect techniques (Mapstone and Ayling 1998).

Temporal stratification of counts

Fishes entering and leaving a survey area once counts have commenced produce a potential source of variability in UVCs (Mapstone and Ayling 1998). To reduce this effect, counts were temporally stratified to capture those species most likely to leave the sampling area first, either because of mobility or because of their tendency to move away from approaching divers (Samoilys and Carlos 2000). Fishes entering a survey area once counting had commenced were excluded (Mapstone and Ayling 1998; Samoilys and Carlos 2000).

Other fish assemblage parameters

Other standardised fish assemblage parameters were set during this study. If an individual is nearby but does not enter the survey zone then that individual is not counted. Once an individual is counted it is not recounted. Fishes swimming behind the observer are never counted, species clearly attracted to the area due to diver activity should not be counted, and the distance and angle of the observable field should be kept constant to maintain a consistent level of detectability for all fishes (Bortone and Kimmel 1991; Brock 1954; Harmelin-Vivien *et al*. 1985; Keast and Harker 1977).

3.3.3 Observer-related standardisation

Several observer-related standardisation measures were taken in this study that include establishing criteria for the selection of divers, conducting reconnaissance dives, standardising the equipment carried by divers, and conducting thorough observer training.

Observer and diver selection

To conduct research up to a depth of 30 m, South African underwater research regulations require that, at a minimum, all divers are commercial Class-IV or higher certified (South Africa, Department of Labour 2010). A total of eight Class-IV or higher certified divers, excluding non-diving dive supervisors, were used.

Certified divers were categorised as either observers or divers. The term observer refers to those divers who have a duty that has an observational nature, such as fish identification or camera operation, while the term diver refers to any person conducting underwater work in this study and can include observers.

To reduce variability in visual counts only a single observer, Reece Wartenberg (RW), was used for all slate transects. Three other observers were restricted to camera work and the remaining four divers restricted to the buddy-diver role. After selecting divers, a reconnaissance diving phase was initiated.

Reconnaissance diving

Reconnaissance dives served as the pilot phase for the study. A total of four reconnaissance dives were carried out on Two-Mile Reef before transect type, survey technique, and fish assemblage parameters were established. Reconnaissance dives were conducted at random dive sites using the flexible, parameter-free roving diver technique. Dives were to maximum depths of 11 m, 14 m, 18 m and 24 m, with total dive times of 55 min, 45 min, 38 min and 32 min respectively. During reconnaissance dives, each diver was equipped with a double-sided A4 dive slate. One diver was provided with a photographic camera and one diver was provided with a video camera. Information and footage were analysed in a group discussion after each dive. Only one reconnaissance dive took place on a given day with two to four days between dives. Reconnaissance dives afforded an opportunity to trial run transects for the purpose of fine tuning parameters, such as transect length and minimum fish size, to practice fish identifications, to gather footage for fish identification and site parameter training, and to determine which camera settings would be the most appropriate. Reconnaissance diving also afforded an opportunity to determine what equipment divers would need during transects.

Diver equipment

The equipment carried by divers was provided to divers prior to deployment and did not change throughout the study.

All divers carried slates to record site parameter information *in situ*. Dive slates consisted of a table with cells for recording depth readings, habitat complexity, substrate type, visibility, water temperature, current speed and direction, and time of day.

The single visual count observer was provided with a double-sided A4 dive slate divided into columns for „common name‟ and „number observed‟ so that abundance and richness information could be recorded.

Other equipment that was necessary for conducting transects included an 80 ml plastic jar for collecting water samples, a 50 m dive reel with a dive weight attached to the tag end enabling fixation of the transect start point, and primary and secondary pencils for recording information.

Diver training

Edgar (2004) state that, in the case of studies conducting spatial analyses, extensive diver training is less necessary as any confounding associated with diver variability can be removed using a balanced sampling design with the same combination of divers at different sites. Mapstone and Ayling (1998) and Langlois *et al.* (2010) further stress that the importance of diver calibration will be greatest in studies where temporal rather than spatial comparisons are of interest. Two temporal scales are of interest in the present study: directly between seasons and indirectly in the development of a long-term monitoring program. The diver training (Appendix 2) was conducted in terms of fish identification, transect width estimation, diver swimming speed, and the measurement and/or estimation of supplementary variables.

Once diver training was completed, the in-field survey phase of the study could commence under a standardised transect deployment protocol.

3.3.4 Standardised transect deployment

All diving operations were conducted from a rigid-inflatable boat. Upon arrival at randomly selected co-ordinates, two divers were deployed using the negative entry technique.

Once at the bottom, and maintaining neutral buoyancy 1 m above the reef to prevent diver damage, divers swam 10 m away from the drop zone, into the current, along the depth contour. This 10 m buffer zone was implemented to reduce the effect of diver disturbance on fish behaviour. As the effect of depth was of interest in the present study, transects were conducted along the depth contour. To minimise visibility disturbance and to facilitate easier swimming speed control, transects were conducted into the current. Each diver within a buddy team, referred to as Diver I and Diver II, was assigned specific duties for each transect type (Appendix 1).

The duties of Diver I for all transects was to swim a straight line transect that tracked the depth contour as closely as possible (Fig. 3.1), while maintaining swimming speed at 6.25 $m.min^{-1}$ (Appendix 1). A straight strip transect, opposed to S-type strip transects (Pelletier *et al.* 2011), was chosen as it was anticipated that straight transects would allow for decreased variability in transect width estimates by observers conducting slate transects.

Diver II swam next to Diver I while extending the transect line. The line was marked at 10 m intervals which served to alert diver II as to when to record depth and when to terminate the transect (Mapstone and Ayling 1998; Zeller and Russ 2000) (Appendix 1). The tag end of the transect line, effectively the start of the transect, was connected to a 1 kg lead weight which was placed haphazardly on the substrate at the start of the transect without damaging corals.

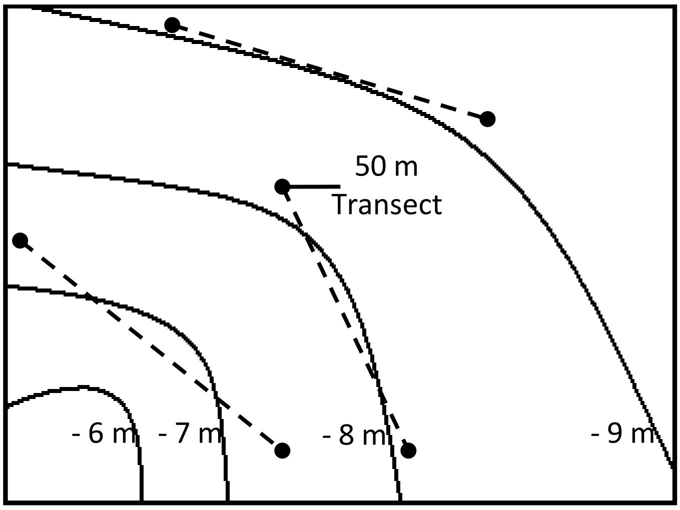


Figure 3.1: Theoretical example of three 50 m straight strip transects conducted along respective depth contours.

Depending on the transect technique being conducted, the identification and enumeration of fishes or camera operation commenced as soon as the dive weight marking the start of the transect had been placed and the stop watches of both divers had been simultaneously started. All site parameter information was recorded *in situ*, after completion of each transect, at the sample depth.

Chapter 4:

Comparing underwater visual census transect and analysis techniques applicable to surveying epibenthic coral reef fish assemblages in the Western Indian Ocean

4.1 Introduction

Coral reef ecosystems are amongst the world"s most complex and anthropogenically threatened marine systems (Connell 1978; Ray 1988; Wilkinson 2004; Wilson *et al.* 2008; Pelletier *et al.* 2011). If coral-reef biodiversity is to be conserved, they need to be surveyed. Underwater visual censuses (UVCs) of various forms have been implemented, with varying degrees of success, to estimate fish assemblage metrics such as abundance, richness, diversity and biomass (Bortone *et al.* 1991; Tessier 2005). Understanding spatial and temporal patterns in these metrics is essential for within- and between-study comparisons (Clarke and Gorley 2006; Kremen 1992). Although unconfounded comparisons can only be made with data that are sufficiently accurate, precise, and representative of the community in question (Bower *et al.* 2011), the survey technique implemented must be appropriate to the goals of a study, efficient and cost effective (Langlois *et al.* 2010). The most crucial consideration in any ecological study is the choice of sampling method (Thomas 1996; Rotherham *et al.* 2007). Of the many UVC techniques available, the strip transect technique is one of the most widely used because it is easy to implement, possesses fixed area parameters, and it is suitable to the survey of epibenthic reef fishes (Schmitt *et al.* 2002; Colvocorresses and Acosta 2007; Dickens *et al.* 2011). Traditionally all UVC techniques have been conducted by observers who record observations on dive slates, but digital technology, such as videographic media, has become readily available and affordable.

The slate transect technique has been used since the 1950s (Brock 1954) and is now one of the most common methods of surveying reef fishes (English *et al.* 1994). Provided trained observers are available, all UVC survey techniques conducted using slates are simple to implement. The use of slates has been shown to possess limitations though, which include the subjective estimation of survey area, a lack of reference images, inter-observer variability, and observer task-loading (Brock 1982; Bortone and Kimmel 1991). The use of digital recording media can mitigate these limitations in that survey area is standardised, reference images can be stored, inter-observer variability is minimised, and because digital media can be reviewed at any speed, observer task-loading is omitted (Watson *et al.* 2005; Langlois *et al.* 2010; Watson *et al.* 2010). Additional benefits to conducting surveys using digital image media are that no fish identification experts are required in the field for conducting surveys, images allow scope for the analysis of additional environmental information such as reef architecture, and images may be reviewed multiple times by multiple observers (Chapter 1). Traditional slate surveys are therefore beginning to become replaced by surveys conducted using digital image media (e.g. Naiberg *et al.* 1993; Pettrell *et al.* 1997; Watson *et al.* 2005; 2010; Pelletier *et al.* 2011).

Digital videographic media for use with traditional strip transects has been assessed (e.g. Watson *et al.* 2010; Pelletier *et al.* 2011), but its applicability to a variety of systems is not fully understood. Three variations of the strip transect technique were therefore selected for *in situ* assessment in this study; traditional slate, photographic, and videographic transects. As the identification and enumeration of fishes observed using digital photography and videography does not need to occur instantaneously, footage from these digital images of transects can be analysed using a number of techniques.

There are three different approaches to analysing ichthyofaunal abundance and richness from digital footage. These are *Standard-count*, *MaxN*, and *Multiple MaxN* techniques (Chapter 1). The *Standard-count* or *MaxN* techniques are more commonly used for the analysis of digital media and have been applied to a variety of UVC scenarios. *Multiple MaxN* analysis is essentially an extension of the original *MaxN* technique that is less commonly applied. Although previous investigations have used *Standard-count* and *MaxN* techniques for different within-study applications (e.g. Watson *et al.* 2005; Becker *et al.* 2010), no previous study has directly compared these two techniques. This study therefore compared the suitability of *Standard-count* and *MaxN* media analysis techniques to generating fish assemblage data obtained by photographic and videographic transects by reviewing footage on multiple occasions.

Despite the ability to re-review photographic and videographic footage (Watson *et al.* 2005; Pelletier *et al.* 2011), no previous studies have suggested a method for validating counts generated by digital image UVCs. Although it is not possible to validate counts generated by a transect technique against true community values, known as true validation, it is possible to validate the counts of observers against the true count generated by a digital transect technique. Observer count validation would effectively remove, or at least minimise, observer count bias and variability, one of the fundamental criticisms of UVCs.

The aim of this chapter was to compare and contrast three transect techniques (slate, photographic and videographic) and two media analysis techniques (*Standard-counts* and *MaxN*) to determine which technique-analysis combination is the most appropriate for surveying epibenthic coral reef fish communities. The five transect-analysis combinations investigated were: slate transects, photographic transects with *Standard-count* analysis, videographic transects with *Standard-count* analysis, photographic transects with *MaxN* analysis, and videographic transects with *MaxN* analysis. For each combination, overall abundance, community richness and community diversity were estimated. In addition, the sample size required to achieve sufficient statistical power, the implementation efficiency, and the ecological groups surveyed by each technique were assessed. Lastly, a preliminary method for validating counts generated from digital media is presented. Comparisons were conducted on a poorly studied, highly diverse, high-latitude coral reef in the Western Indian Ocean.

4.2 Materials and Methods

4.2.1 Study area and sampling design

The study was conducted on Two-Mile Reef in the iSimangaliso Wetland Park, South Africa. Motivation for the selection of Two-Mile reef is provided in Chapter 2, while additional details pertaining to the randomised allocation of sample sites, quantification of sample site parameters, observer and diver selection, reconnaissance dives, transect and community parameters, transect deployment protocol and diver training are provided in Chapter 3. Parker *et al.* (1994) and Clua *et al.* (2006) note that of the various physical and biological factors structuring fish communities, the most influential are substrate type and the associated benthic community. A random stratified sampling design therefore was employed by sampling predominantly non-sandy, reef substrate areas, and only one similar and comparable benthic community was sampled.

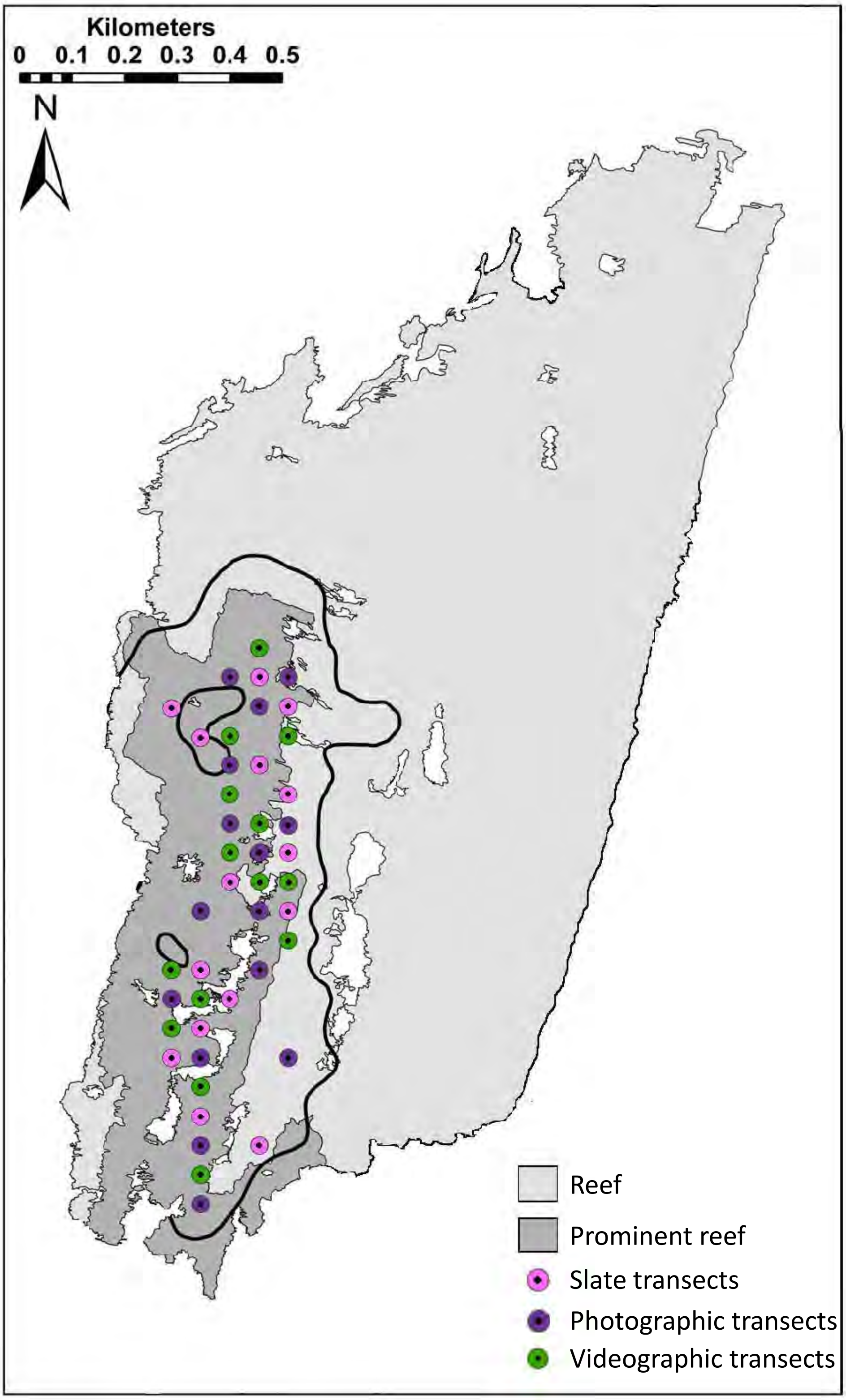


Figure 4.1: Transect start points for slate, photographic and videographic transects conducted between -10 m and -16 m on Two-Mile Reef in the Central Reef Complex in the iSimangaliso Wetland Park, South Africa. Depth contour and geological data were obtained from the Marine Geoscience Unit of the Council for Geoscience, South Africa. Spatial reference system: Transverse Mercator (Central Meridian +27.00).

Celliers and Schleyer (2008), who conducted a detailed assessment of the benthic communities of South African coral reefs including Two-Mile Reef, showed that 16 discrete coral communities exist. Of these 16 communities, the dominant community occurs on Two-Mile Reef at a depth ranging from 10 m to 16 m. This community consists of hard and soft corals forming most of the living benthic cover (Celliers and Schleyer 2008). Sampling for this work was restricted to this narrow 10–16 m depth range on the reef flat and fore-reef slope of Two-Mile Reef under the assumption that a only a single benthic invertebrate community, and it‟s associated coral-reef ichthyofauna, would be surveyed (Fig. 4.1). Using ArcGIS 10 (© Environmental Systems Research Institute), a grid overlay with 50 m × 50 m cells was georeferenced over the reef area meeting the necessary survey criteria. Sample sites were then randomly selected without replacement (Chapter 3).

The most suitable sampling design maximises the number of replicates logistically feasible, while taking into account the time and funding available such that there is a realistic trade-off between accuracy and precision (Winer *et al.* 1991). To meet these criteria, Samoilys and Carlos (2000) showed that for counts of typical reef fish, which tend to be highly variable, a minimum of 10 replicates should be conducted. This study conducted 15 replicates of each transect technique resulting in a total of 45 successful transect deployments.

Table 4.1: Sampling dates and design for the comparison of slate (S), photographic (P), and videographic (V) transects.

Date	Transects/day	Sampling structure
09/06/2010	4	P,V,S,P
10/06/2010	9	V,S,P,V,S,P,V,S,P
11/06/2010	9	V,S,P,V,S,P,V,S,P
12/06/2010	9	V,S,P,V,S,P*,V,S,P
13/06/2010	9	V,S,P,V,S,P,V,S,P
14/06/2010	6	V,S,P,V,S,P
Total	46	S & V (n = 15); P (n = 16)

* Indicates a failed transect

The dive plan for each transect deployment was standardised prior to study commencement. Planned total dive time was 21 minutes which, within the depth range, allowed for each diver to conduct three dives per day while remaining within acceptable residual nitrogen levels under South African Department of Labour regulations (South Africa, Department of Labour

2010). A team of six divers, comprising three buddy pairs, conducted a maximum of nine transects per day. Buddy pairs were permanently assigned to each transect type to ensure consistency in sampling and reduced intra-transect type variability.

Sampling occurred over six days from the 09 June 2010 to 14 June 2010 (Table 4.1). To minimise the effects of short-term shifts in environmental conditions (e.g. wind, tide, moon phase etc.), a single replicate per transect technique was conducted and repeated until all 45 transects were completed (Table 4.1).

4.2.2 Transect techniques

Different transect techniques have specific characteristics. To reduce intra-transect type variability, divers were permanently assigned to specific duties for each transect type (Chapter 3). Only the additional duties for Diver I varied between transect type. The specific parameters of slate, photographic and videographic transects are summarised in Table 4.2.

Table 4.2: Slate, photographic (Photo) and videographic (Video) transect parameters.

Characteristic	Slate	Photo	Video
Length (meters)	50	50	50
Time (minutes)	8	8	8
Width (meters)	6.00	8.66	4.79
Area (meters^2)	300	433	239.5
Camera	N/A	SEA&SEA DX1200 HD	SEA&SEA DX1200 HD
Camera lens	N/A	Wide angle	Standard
Camera settings	N/A	Mode: SEA&SEA Size: 4344 x 3258 Quality: S. Fine Metering: Multi	Mode: HD Size: 1280 x 720 30 FPS Quality: S. Fine Metering: Multi

Slate transects

Diver I was responsible for visually identifying, counting and recording all individuals present within the transect dimensions. Transect width was set to 6 m by visually estimating a 3 m distance either side of the transect line (Table 4.2) (Chapter 3). This modest transect width was selected *a priori* because it reduced observer task loading thereby increasing accuracy and precision (Zeller and Russ 2000) and is below the minimum reported visibility of the area ensuring that fish could be easily identified and enumerated (Schleyer 2000). All observations falling within the 300 m^2 transect area were noted by Diver I on an A4 Perspex dive slate. Instances where large monospecific groups (>50 individuals) were observed, sets of 20 individuals were used to form a visual image of a proportion of the group. These figures were then multiplied by the area of the group to form an estimate of the total group size (Harmelin-Vivien *et al.* 1985, Bortone *et al.* 2000). The same method was used for mixed species groups but differed in that an estimate of the proportion of each species was used to estimate species specific counts for the entire group (Bortone and Kimmel 1991; Bortone *et al.* 2000).

Photographic transects

Photographic transects were conducted using a DX 1200 HD Camera (© SEA&SEA SUNPAK Co., Ltd.), set to underwater stills mode. A TTL SEA&SEA YS-17 strobe and wide angle conversion lens were externally attached. Transect width was therefore 8.66 m – the width of the camera‟s field of view at a distance of 5 m. Total photographic transect area was therefore 433 m^2 (Table 4.2). Diver I was responsible for the camera rig while taking photographs at 1m intervals resulting in 50 photographs per transect.

Video transects

Video transects were conducted using a DX 1200 HD Camera (© SEA&SEA SUNPAK Co., Ltd.) in its SEA&SEA underwater housing set to underwater video mode. The external wide angle conversion lens was not used during video transects as it was determined, during reconnaissance dives (Chapter 3), that video quality was not satisfactory to facilitate accurate fish identifications. Transect width was therefore 4.79 m – the width of the camera‟s field of view at a distance of 5 m. Total transect area was 239.5 m^2 (Table 4.2). Diver I was responsible for the camera rig while recording a single, continuous video clip of the 50 m transect resulting in approximately eight minutes of video per transect. The distance and angle of the observable field should be kept constant to maintain a consistent level of detectability for all fishes (Keast and Harker 1977). All photographs and video footage were captured at a 1 m elevation on a plane parallel to the slope of the reef.

4.2.3 Digital media analysis

Transects were randomly selected for analysis such that the observer was unable to relate individual transects to particular localities. Fishes that could not be accurately identified because they were too far away, moved through the census area too rapidly, or were obscured due to poor lighting, camera angle, individual orientation or other factors were categorised as „unidentifiable‟ (Alevizon and Brooks 1975).

Photographs were analysed using Adobe Photoshop CS3 (© Adobe Systems Incorporated). Simple image adjustments were made to brightness, temperature, tint and saturation of photographs to facilitate easier identifications as required. High picture quality allowed for up to 300% image enlargement.

Videos were analysed using VLC media player 1.0.5 Goldeneye (© VideoLAN) software at 0.33× playback speed. Multiple pauses, rewinds, and repeated playbacks were used as and when required.

Standard-counts

Photographs

During photographic *Standard-counts*, all fishes appearing in each photo were counted unless it was clear that they had been counted in a previous photograph. If it was unclear whether or not an individual had been counted that individual was excluded from the count.

Video

During video *Standard-counts* all fishes appearing on the screen during a transect (an entire video clip) were identified and counted. In the case of species occurring at higher abundances, i.e. >10 individuals per transect, transects were analysed on a per species basis such that full transect playback occurred for each species. Species occurring at lower densities, i.e. <10 individuals per transect, were counted simultaneously thereafter. Short portions of the video were analysed multiple times where necessary with numerous pauses, rewinds and playbacks.

MaxN

The methods of *MaxN* analysis are based on methods outlined by Priede *et al.* (1994) and Willis and Babcock (2000).

Photographs

During photographic *MaxN* counts each transect was analysed on a per species basis. For each species in each transect only the photograph displaying the greatest number of individuals was used as the count for that species. For example, within a set of 50 photographs species *x* occurs in three photographs. Each of these photographs is separated sufficiently in space, i.e. distance along the transect, and time to suggest that, in each of the three photographs, new individuals are being observed. It is not the sum of the individuals counted in all three photographs, as is the case for *Standard-count* analysis, but rather the single photo with the greatest number of individuals of species *x* that is counted. The resulting value is then accepted as the total count for that species for the entire transect.

Video

During video *MaxN* counts, each video was analysed on a per species basis. For each transect only the single video frame which displayed the greatest number of individuals per species was taken as the total count for that species. This was conducted in a manner similar to that of photographic *MaxN* analysis except individual video frames were used instead of photographs.

The total time taken to complete the analysis process for each photographic and videographic transect using both *Standard-counts* and *MaxN* was noted for all analyses.

4.2.4 Ecological group affinity

Previous studies have assigned species to ecological groups to obtain information on factors such as community trophic structure (Floros 2010b), the influence of particular groups on results (Tessier *et al.* 2005) and the colonisation of particular groups on artificial reefs (Nakamura 1985). This study assigned species groups to all individuals observed in photographic and videographic transects (*Standard-counts* and *MaxN*) to determine if different groups were favoured by different transect/media analysis techniques. An individual was assigned to these groups based on the position and movement of that individual in the water column and its position relative to the reef (Nakamura 1985; Tessier *et al.* 2005). Ecological groups were similar to those outlined by Tessier *et al.* (2005) and consisted of three groups. The benthic fishes, which included individuals that tend to have direct contact with the reef structure and often occupy crevices, holes or gaps within the reef. Benthic individuals may leave the reef surface if disturbed but settle back down soon after. Epibenthic fishes, which included individuals swimming just above the reef surface. Epibenthic individuals may come into direct contact with the reef intermittently for shelter, to forage, or for other activities but only for short periods. Pelagic fishes, which included individuals which are found above the reef in mid-water or in the pelagic zone. Pelagic individuals are constantly roaming. Slate transects were excluded from this comparison as the assignment of individuals to species groups during transects would have resulted in observer task over loading. It must be noted that while analysing photographic transects the assignment of species to species groups required some subjectivity.

4.2.5 Validation of photographic and videographic counts

Observer count validation was achieved by analysing photographic and videographic digital media multiple times using the same observer, the underlying assumption being that if two stages of analysis generate sufficiently similar counts, then those counts are probably similar to the true count generated by the technique. This yields an estimate with reduced observer error.

Analyses were conducted two weeks apart with transects analysed in a new random order (Chapter 3). Percentage agreement between the first two stages of analysis for each technique was used as the proxy to determine whether or not a third analysis was required. Two aspects of percentage agreement were implemented for all transects. These were diversity agreement and species-specific abundance agreement. Percentage diversity agreement, i.e. species presence/absence in a transect, was set to 100% as a species is either present or absent on a photograph or in video footage. Species-specific abundance agreement was set to >95% for all species observed in each transect. This level of agreement was considered acceptable because of the difficulties associated with counting highly abundant or shoaling species, and because a 5% count will not greatly bias results. For example, when a particularly abundant species is counted, with say >200 individuals per transect, an error of <10 individuals is unlikely to bias community-level analyses over many replicates. Alternatively, an error of just one or two individuals is also unlikely to bias community-level analyses of species occurring at very low abundances over many replicates. In cases where species-specific abundances were between 95% and 100% agreement, the mean of the two abundance estimates was accepted as the final value. In cases where more than two stages of analyses were required, the percentage agreement rules applied to the two most recent stages of analysis. The same >95% agreement rule was also applied to the number of fishes deemed „unidentifiable‟ thereby accounting for all observed individuals in the validation process. Once all transects had been analysed, all individuals assigned to ecological groups, and the percentage agreement requirements met, additional statistical analyses were conducted.

4.2.6 Statistical analyses

Site parameters

Site parameters were grouped as either measured quantitatively or categorised qualitatively. All quantitative parameters were tested for normality using Shapiro–Wilk‟s normality test within each transect type and for homoscedacity, i.e. equality of variances, using Levene‟s test between transect types. In instances where these assumptions were violated, data were ln(x + 1) transformed. Time of day, wind direction and current direction were converted to their periodic equivalents as described by Jammalamadaka *et al.* (2006) prior to testing. The qualitative site parameters, habitat complexity and substrate type, could not be statistically analysed and were assessed visually.

Transect type comparison

Richness and Abundance

UVC has been shown to underestimate richness and abundance because of the innate difficulties in non-destructively detecting all fishes present in a population (Chapter 1). When comparing UVC techniques, therefore, the technique which provides the highest estimates of richness and abundance is considered to be the most appropriate (Underwood and Chapman 2003). As each transect technique consists of a different transect area (Table 4.2) all metrics were area standardised for a total abundance or richness per 100 m^2.

All area standardised estimates of the three transect techniques and two media analysis types were tested for normality using Shapiro–Wilk‟s normality test within each technique and for homoscedacity using Levene‟s test between techniques. Where parametric assumptions were met, a one-way ANOVA was used to test the hypothesis that there was no difference in abundance or richness between transects using each media analysis method. Where parametric assumptions were violated, data were root-root transformed. Tukey‟s HSD test was used for pairwise comparisons.

Variability

Variability provides a reasonable proxy for the quality of data produced by a transect technique, because the technique that produces the lowest variability is assumed to provide the best quality data (Underwood and Chapman 2003; Winer *et al.* 1991). Rosner (2000) provides an explanation of a wide range of measures potentially suitable for quantifying variability. McArdle *et al.* (1990) suggest that, of the available measures of variability, the most appropriate are those that are independent of mean population estimates and sample size, and should utilise the data with the highest resolution. Bennett (2008) states that the two most commonly used, appropriate measures of variability that meet these criteria are the standard deviation of the natural logarithms of successive population estimates and the Coefficient of Variation. The standard deviation of the natural logarithms of successive population estimates $SD(\ln(x_i + 1))$ (Connell and Sousa 1983) is approximately equal to the Coefficient of Variation (Rosner 2000). The present study therefore only used the standard deviation of the natural logarithms of successive population estimates as a measure of data variability.

Diversity

The diversity captured by each transect technique, and by *Standard-counts* and *MaxN* analysis, was compared using the Shannon–Wiener diversity index, Shannon‟s *H'* (Shannon and Weaver 1949) that takes into account the relative proportions of each species. Only a single diversity index was employed for this phase of the project as the aim was not to investigate the dynamics of fish communities, but rather to conduct an intra-transect type comparison (Magurran 1988). Shannon‟s *H'* is calculated as

$$H' = -\sum p_i(logp_i)$$

where $p_i = (n_i/N)$, the proportional abundance of the ith species, n_i is the abundance of that species i, and $N = \sum n_i$ the total abundance for the sample (Shannon and Weaver 1949). All diversity estimates were tested for using Shapiro–Wilk‟s normality test within each technique, and for homoscedacity using Levene"s test between techniques. One-way ANOVA was used to test the hypothesis that there was no difference in Shannon"s *H'* between transect techniques within each analysis technique.

In addition to *H'*, a proportional list of the five most common species detected by each transect technique was compiled to enable fine scale assessment of possible differences in the most common species surveyed by each technique.

Sample size

Determining the correct number of replicates, or sample size, is an essential component of the experimental design of any sampling program (Samoilys and Carlos 2000). From the species richness data obtained for the three transect techniques it is possible to calculate the statistical power (Kapadia *et al.* 2005). Similarly, when planning for future research, one can calculate the required sampling effort required using a particular transect technique based on a predetermined level of statistical power (Utts and Heckard 2007). When comparing transect techniques, therefore, the most appropriate technique for future implementation will be the technique requiring the lowest number of replicates to achieve the predetermined power level.

The sample size, n, required to detect a 10% change in the mean estimate of species richness, $\Delta\mu$ (Kapadia *et al.* 2005), at a significance criterion of $\alpha = 0.05$ (Rosner 2000; Bausell and Li 2002) with a power of 80% , such that $\beta = 0.2$, (Fairweather 1991, Rosner 2000, Lenth 2001, Utts and Heckard 2007) was calculated as

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

where z_* is the z-statistic corresponding to a probability of $*$, and σ is the standard deviation of the estimates.

Efficiency

The efficiency of the three transect and two media analysis techniques was assessed in three ways. These were the time required to train observers, the time required for analysis of the different transect techniques, and the proportion of individual fish rendered unidentifiable by each technique. The technique displaying the greatest efficiency would be the most appropriate for future implementation.

Time taken to train observers was quantified by the product of the number of days it took to train the single visual observer (Chapter 4) and the number of visual observers that would have been required to implement this study.

Total analysis time for each transect technique was calculated as the sum of the two stages of analysis meeting the percentage agreement criteria summed over all transects for that technique. The mean time required for the analysis of slate transects, i.e. data capture, was standardised to 15 minutes per transect.

The proportion of individuals rendered unidentifiable was calculated by dividing the number of unidentifiable individuals by the total number of individuals in a transect, i.e. number unidentifiable / (number unidentifiable + number identified), for each transect and media analysis technique. As it was impossible to determine the proportion of individuals unidentifiable using the slate transect technique, the worst case scenario in this comparison of efficiency, this value was set to 1.0 for all transects to aid in visual assessment.

The open source statistical environment in R 2.11.0 (© R development core team 2010) was used for all analyses.

4.3 Results

4.3.1 Site parameters

Depth, visibility, turbidity, temperature, time of day, current speed, current direction, wind speed, wind direction and transect duration data were all normally distributed for each transect type and homoscedastic between transect types ($p>0.05$). Profile and rugosity data had to be $\ln(x + 1)$ transformed ($p>0.05$ post transformation). The results of parametric one-way ANOVA indicate that there was no significant difference in parameters between transect types ($p>0.05$; Table 4.3).

Visual assessment of habitat complexity and substrate type shows that there were no significant differences in these parameters between transect type (Fig. 4.2). No replicates were conducted on a purely sandy substrate (Fig. 4.2).

4.3.2 Transect type comparison

A total of 184 species from 35 families were identified using all three transect techniques. A total of 12 681 individuals were counted using the *Standard-counts* technique (slate, photographic and videographic transects), and a total of 3 694 individuals were counted using *MaxN* analysis (photographic and videographic transects only). Data specific to each transect technique are summarised in Table 4.4. Richness data for the *Standard-counts* and *MaxN* media analysis techniques are identical. Full lists of species observed using each technique is appended (Appendix 3).

Table 4.3: Summary of supplementary variables recorded for the comparison of Slate, Photographic (Photo), and Videographic (Video) transects. Supplementary variable means (± SD). Values are for untransformed data. Time of Day, Current Direction and Wind Direction data are for circular equivalents. n = 15 for all parameters recorded for Slate, Photo, and Video transects.

Parameter #	Parameter	Slate	Photo	Video	Statistical results
1	Depth (m)	$\bar{x}$ = 12.01, SD = 01.51	$\bar{x}$ = 11.37, SD = 01.07	$\bar{x}$ = 11.22, SD = 1.03	$F_{2,42}$ = 1.76, P = 0.19
2	Visibility (m)	$\bar{x}$ = 11.50, SD = 02.00	$\bar{x}$ = 12.25, SD = 01.21	$\bar{x}$ = 12.77, SD = 1.82	$F_{2,42}$ = 2.08, P = 0.14
3	Turbidity (NTU)	$\bar{x}$ = 0.37, SD = 00.11	$\bar{x}$ = 0.34, SD = 00.09	$\bar{x}$ = 0.38, SD = 0.13	$F_{2,42}$ = 0.53, P = 0.59
4	Temperature (˚C)	$\bar{x}$ = 22.80, SD = 00.41	$\bar{x}$ = 22.80, SD = 00.41	$\bar{x}$ = 22.80, SD = 0.41	$F_{2,42}$ <0.01, P = 1.00
5	Current Speed (s/m)	$\bar{x}$ = 12.13, SD = 08.83	$\bar{x}$ = 13.40, SD = 06.65	$\bar{x}$ = 13.15, SD = 8.11	$F_{2,42}$ = 0.11, P = 0.90
6	Wind Speed (kts)	$\bar{x}$ = 8.73, SD = 04.45	$\bar{x}$ = 8.60, SD = 04.52	$\bar{x}$ = 9.12, SD = 4.53	$F_{2,42}$ = 0.06, P = 0.95
7	Transect Duration (mins)	$\bar{x}$ = 8.83, SD = 00.96	$\bar{x}$ = 8.37, SD = 01.22	$\bar{x}$ = 8.58, SD = 1.05	$F_{2,42}$ = 0.70, P = 0.50
8	Profile (index)	$\bar{x}$ = 1.74, SD = 01.57	$\bar{x}$ = 2.33, SD = 01.72	$\bar{x}$ = 2.74, SD = 2.97	$F_{2,42}$ = 0.70, P = 0.57
9	Rugosity (index)	$\bar{x}$ = 0.57, SD = 00.26	$\bar{x}$ = 0.64, SD = 00.03	$\bar{x}$ = 0.70, SD = 0.41	$F_{2,42}$ = 0.47, P = 0.63
10	Time of Day (hh:mm)	$\bar{x}$ = 00.42, ρ = 0.99	$\bar{x}$ =0.43, ρ = 0.99	$\bar{x}$ = 0.39, ρ = 0.99	$F_{2,42}$ = 2.14, P = 0.13
11	Current Direction	$\bar{x}$ =0.52, ρ = 0.31	$\bar{x}$ =0.23, ρ = 0.37	$\bar{x}$ =0.52, ρ = 0.31	$F_{2,42}$ = 1.0, P = 0.98
12	Wind Direction	$\bar{x}$ =1.85, ρ = 1.0	$\bar{x}$ =1.85, ρ = 1.0	$\bar{x}$ =1.85, ρ = 1.0	$F_{2,42}$ = 0.02, P = 0.38

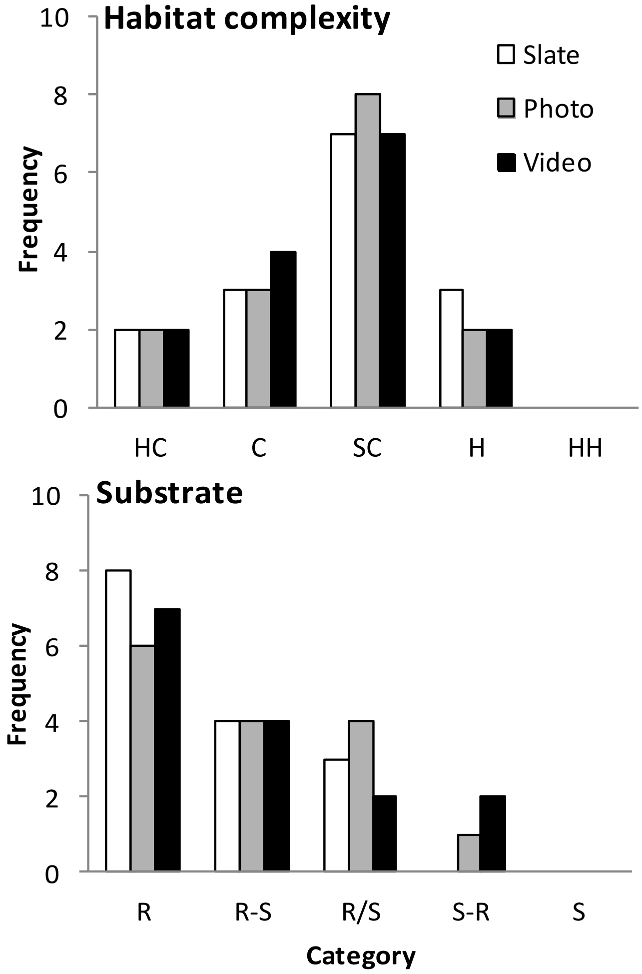


Figure 4.2: Categorical frequencies for the site parameters for habitat complexity (HC = highly complex, C = complex, SC = semi-complex, H = Homogenous, HH = highly homogenous) and substrate type (R = reef, R-S = reef-sand, R/S = reef/sand, S-R = sand-reef, S = sand) as estimated by divers.

Table 4.4: Summary data of non-standardised richness (number of species) and abundance (number of individuals) for Slate, Photographic (Photo), and Videographic (Video) transects. *Standard-counts* (*S-C*) and *MaxN* (*MN)* media analysis data are displayed for photographic and videographic transects.

	Slate	Photo (*S-C*)	Video (*S-C*)	Photo (*MN*)	Video (*MN*)
Transect area (m^2)	300	433	239.5	433	239.5
Total richness	140	118	137	118	137
Minimum richness	22	28	34	28	34
Maximum richness	51	52	56	52	56
Mean richness	35.4	38.93	42.47	38.93	42.47
Total abundance	3853	4492	4336	2027	1667
Minimum abundance	114	126	154	64	71
Maximum abundance	413	582	499	234	181
Mean abundance	256.87	299.47	288.47	135.13	110.53

Richness and Abundance

Area standardised richness and abundance data for slate, photographic (*Standard-count*) and videographic (*Standard-count*) transects were found to be normally distributed for each transect type and homoscedastic between transect types (p>0.05, Table 4.5). Videographic *Standard-count* estimates of richness were significantly higher than the the other two techniques ($F_{2,42}$ = 58.50, p<0.01; Fig. 4.3, Table 4.5). All estimates of abundance estimated using the *Standard-count* technique were significantly different from one another with videographic transects providing the highest estimate of area standardised abundance ($F_{2,42}$ = 7.92, p < 0.01; Fig. 4.3, Table 4.5). Area standardised *MaxN* richness and abundance data, which can only be determined for photographic and videographic transects by re-analysing digital media, was root-root transformed before the parametric assumptions of normality and homoscedacity were met (p>0.05,Table 4.5). Videographic transects analysed using the *MaxN* technique provided significantly higher estimates of abundance ($F_{1,28}$ = 22.40, p<0.01; Fig. 4.3, Table 4.5).

Table 4.5: Summary of area standardised richness and abundance data for Slate, Photographic (Photo) and Videographic (Video) transect techniques and analysis type (*Standard-count* and *MaxN*). Mean ($\bar{x}$) ± standard deviations (SD) are for untransformed data. Common superscripts denote statistically homogenous groups at the 0.05 significance level as determined using Tukey's HSD test within each analysis technique.

Transect technique	Analysis technique	Area standardised metric	$\bar{x} \pm SD$	Root-root transformed
Slate	*Standard-count*	Richness	11.80 ± 02.56^{a}	False
Photo	*Standard-count*	Richness	8.99 ± 01.63^{a}	False
Video	*Standard-count*	Richness	17.73 ± 02.47^{b}	False
Photo	*MaxN*	Richness	8.99 ± 01.63^{a}	False
Video	*MaxN*	Richness	17.73 ± 02.47^{b}	False
Slate	*Standard-count*	Abundance	85.62 ± 30.86^{c}	False
Photo	*Standard-count*	Abundance	69.16 ± 31.31^{d}	False
Video	*Standard-count*	Abundance	120.45 ± 44.34^{e}	False
Photo	*MaxN*	Abundance	31.32 ± 23.46^{f}	True
Video	*MaxN*	Abundance	46.15 ± 13.58^{g}	True

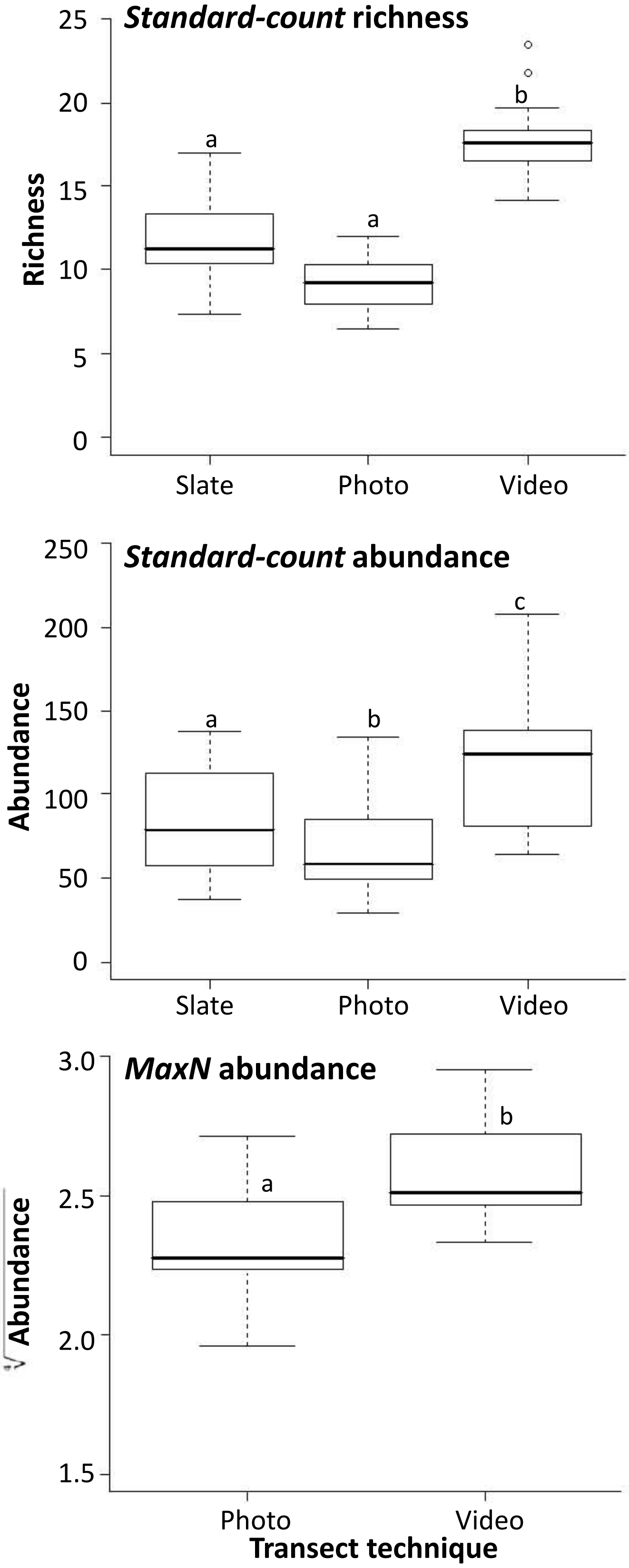


Figure 4.3: Area standardised richness and abundance for the *Standard-count* analysis techniques, and root-root transformed area standardised abundance for the *MaxN* analysis technique. Common superscripts denote statistically homogenous groups at α = 0.05 determined using Tukey's HSD test. As richness data for *Standard-count* and *MaxN* analysis techniques are identical, *MaxN* richness results are omitted.

Variability

Variability in richness and abundance was lowest for videographic transects using both the *Standard-count* and *MaxN* media analysis techniques (Fig. 4.4).

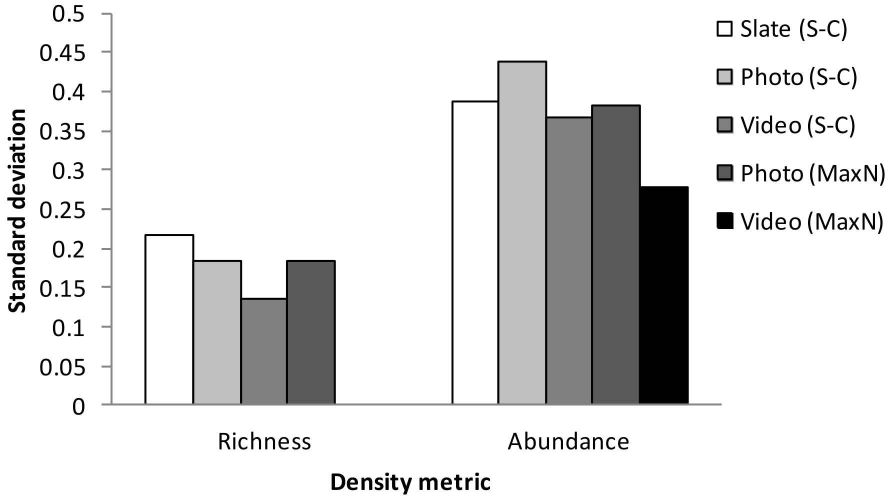


Figure 4.4: Slate, Photographic (Photo) and Videographic (Video) transect variability. S-C = *Standard-count* analysis, MaxN = *MaxN* analysis. As richness data for S-C and MaxN analysis techniques are identical, MaxN richness results are omitted.

Diversity

Shannons *H'* was estimated between 2.48 ± 0.44 and 3.13 ± 0.26 for all transect and media analysis technique combinations (Table 4.6). There was no significant difference in Shannon *H'* between transect techniques for either *Standard-count* ($F_{2,42}$ = 2.39, p = 0.10) or *MaxN* ($F_{2,42}$ = 3.58, p = 0.06) analyses.

Table 4.6: Summary statistics of Shannon's *H'* data for Slate, Photographic (Photo) and Videographic (Video) transect techniques and analysis type (*Standard-count* and *MaxN*). Common superscripts denote statistically homogenous groups at the 0.05 significance level as determined using Tukey's HSD test.

Transect technique	Analysis technique	$\bar{x} \pm SD$
Slate	*Standard-counts*	2.48 ± 0.44^a
Photo	*Standard-counts*	2.57 ± 0.23^a
Video	*Standard-counts*	2.73 ± 0.21^a
Photo	*MaxN*	2.95 ± 0.26^b
Video	*MaxN*	3.13 ± 0.26^b

For *Standard-count* and *MaxN* analysis techniques *Chromis dimidiata* and *Chromis weberi* were the most common species, respectively accounting for 45% and 31% of the total abundance for the five most commonly observed species (Table 4.7).

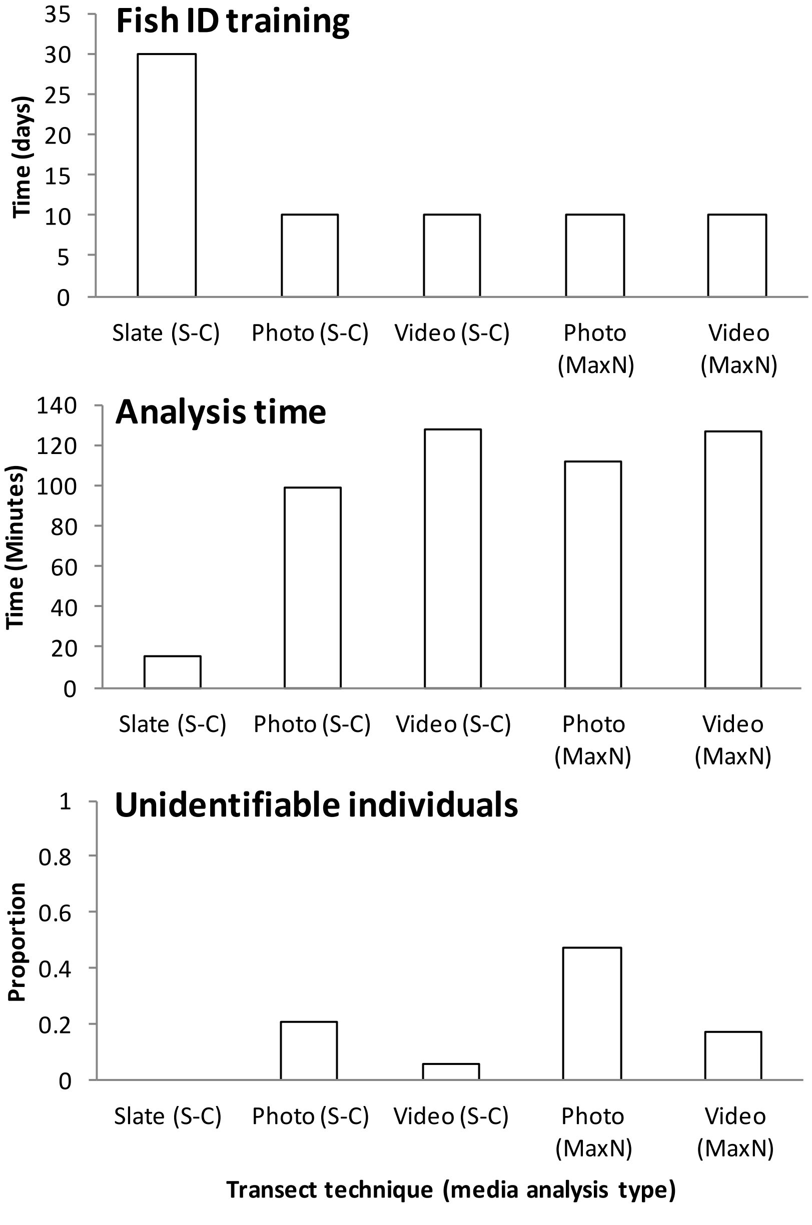


Figure 4.5: Fish identification training time for observers, mean analysis time for each transect, and the proportion of fishes that could not be identified (Unidentifiable individuals). As the proportion of unidentifiable individuals could not be determined, data are omitted. S-C = *Standard-count* and MaxN = *MaxN* analysis techniques.

Table 4.7: Summary of the five most commonly observed species. Data include the total number of individuals (n) detected for each species using *Standard-count* and *MaxN* media analysis techniques, the proportional contribution of total abundance for the five most commonly observed species (Prop. top 5), the rank of the species in overall counts (Overall) and specific slate (Slate), photographic (Photo) and videographic (Video) ranks for each analysis method.

		Standard-counts						*MaxN*				
Species	Family	n	Prop. top 5	Overall	Slate	Photo	Video	n	Prop. top 5	Overall	Photo	Video
Chromis dimidiata	Pomacentridae	3136	0.45	1	1	1	1	472	0.30	2	1	2
Chromis weberi	Pomacentridae	1542	0.22	2	8	2	2	476	0.31	1	2	1
Anthiinae	Serranidae	976	0.14	3	3	3	4	274	0.18	3	3	4
Ctenochaetus binotatus	Acanthurdae	706	0.10	4	6	5	3	132	0.08	7	8	5
Chromis nigrura	Pomacentridae	628	0.09	5	5	6	5	198	0.13	4	6	3

Sample size

Power analysis of transects using *Standard-count* analysis indicated that, in order to detect a 10% change in mean richness with a statistical power of 80%, the number of replicates required for slate, photographic and videographic transects would be 37, 27, and 17, respectively. For photographic and videographic transects, the required number of replicates using the *MaxN* analysis technique are identical to the *Standard-count* technique because species richness results are the same

Efficiency

It took 30 days to train an observer to identify fish using the slate technique and 10 days for the other techniques (Fig. 4.5). Total analysis time was 03 h 45 m, 49 h 32 m, 64 h 13 m, 56 h 12 m and 63 h 51 m for the Slate, Photographic (*Standard-counts*), Videographic (*Standard-counts*), Photographic (*MaxN*) and Videographic (*MaxN*), transects respectively (Fig. 4.5). Total and mean analysis time was longest for videographic transects (*Standard-counts* and *MaxN*) and shortest for slate transects. The proportion of individuals rendered unidentifiable was higher for photographic transects (*Standard-counts* and *MaxN*) than for videographic transects (Fig. 4.5). Slate transects displayed the least efficient characteristic in this regard in that one cannot determine the proportion of fish unidentifiable.

Ecological group affinity

All transect and media analysis techniques showed similar proportions of benthic, epibenthic and pelagic individuals (Fig. 4.6). Epibenthic species constituted the majority of counts for all transects with the lowest proportion of epibenthic individuals occurring in photographic (*MaxN*) transects at 0.72 (Fig. 4.6).

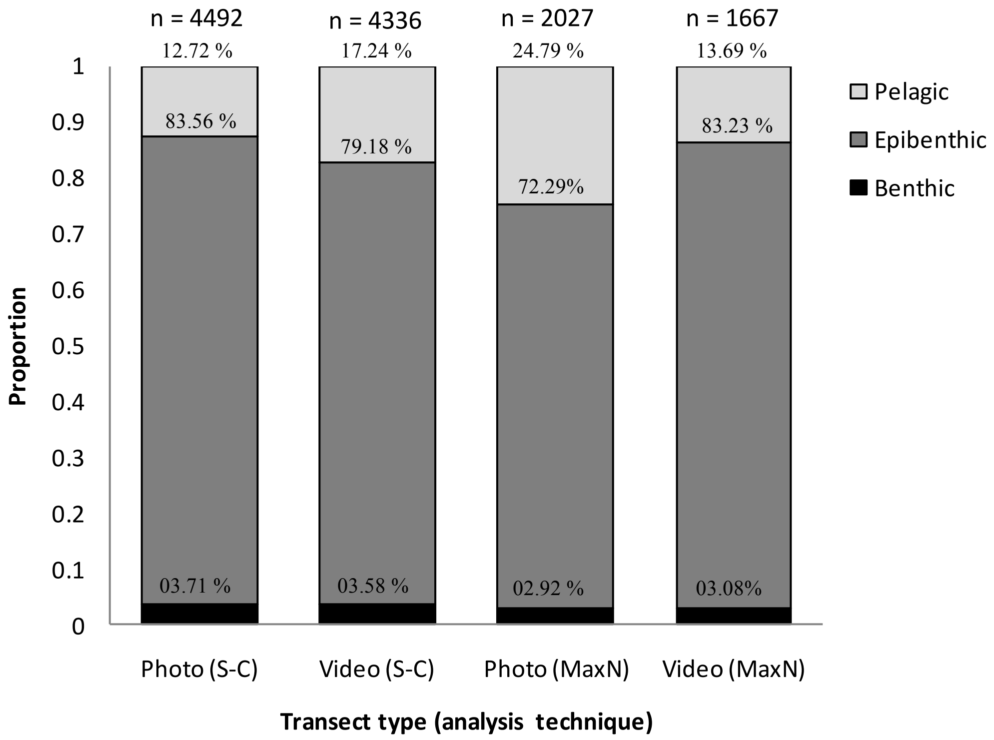


Figure 4.6: Proportional abundance of individuals categorised as Benthic, Epibenthic and Pelagic as defined in the present study for each transect and analysis technique. S-C = *Standard-counts* and MaxN = *MaxN* media analysis techniques.

Count validation

The majority of transects required only two stages of analysis, and a maximum of three stages of analysis to meet percentage agreement requirements (Table 4.8). The number of stages of analysis required to meet percentage agreement requirements did not differ between transect or analysis techniques (Table 4.8). Instances where percentage agreements were not met in the first two stages of analysis were a result of no more than 12% error in abundance estimates.

Table 4.8: Summary of the number of transects requiring two and three stages of analysis for *Standard-count* and *MaxN* analysis techniques.

		Stages of analysis	
Transect technique	Analysis technique	Two	Three
Photo	*Standard-count*	13	2
Video	*Standard-count*	14	1
Photo	*MaxN*	14	1
Video	*MaxN*	13	2

4.4 Discussion

The most appropriate transect technique is that which produces the highest estimates of richness and abundance (Samoilys and Carlos 2000), exhibits the lowest variability (Sale 1991, Bennett 2008), is representative of community diversity (Magurran 1988, Watson *et al.* 2005), requires the least replication (Samoilys 1997, Watson *et al.* 2005), is the most efficient (Pelletier *et al.* 2011) and also surveys the intended ecological group within a community (Tessier *et al.* 2005). This study found that videographic transects outperformed slate and photographic transects in all criteria. Assessment of the *Standard-count* and *MaxN* techniques for analysing digital footage showed that the *Standard-count* technique was more appropriate to analysing footage from transects.

Videographic transects estimated the largest area standardised estimates of richness and abundance for both *standard-counts* and *MaxN* techniques. Both Buxton (1987) and Samoilys and Carlos (2000) note that UVC methods can either under- or over-estimate density (e.g. Watson *et al.* 1995), but overall, generally underestimate abundance (e.g. Brock 1982; Sale and Sharp 1983; Sale 1991). Sale and Sharp (1983) further deduce that in assessing the relative accuracy of various UVC methods it can be assumed that higher estimates of density imply greater accuracy. Based on these conclusions, therefore, videographic transects are the most appropriate transect technique for achieving the best estimates of richness and abundance. Richness and abundance estimates do, however, possess an associated level of variability, which must also be compared.

Samoilys and Carlos (2000), Langlois *et al*. (2010), Chater *et al*. (1995), and this study have all reported similarly high estimates of variability. Sale (1991) and Williams (1991) propose that high variability usually evident in reef fish communities is possibly a result of the contiguous nature of data distributions for the many species sampled. It has been shown for UVC studies that as richness and abundance increase so too does the variability (Thresher and Gunn 1986, McArdle *et al.* 1990, Underwood 1992). Despite producing the highest area standardised richness and abundance estimates, videographic transects (*standard-counts* and *MaxN*) were the least variable in estimating richness and abundance. The variability in videographic transect data is more favourable than that from either the slate or photographic transect techniques. When comparing videographic transects analysed by *Standard-counts* or *MaxN*, transects analysed by *MaxN* produced lower estimates of variability. Because of the linear relationship between the size of estimates and its variability (Thresher and Gunn 1986, McArdle *et al.* 1990, Underwood 1992), and because *MaxN* analyses will always produce significantly more conservative estimates of richness and abundance, one cannot determine which of the two analysis methods is more appropriate based on variability alone. Therefore, with all other factors considered equal, it is preferable to reduce error by selecting the least variable estimate (Winer *et al.* 1991). Videographic transects are therefore recommended as the most suitable transect technique.

Transects analysed using the *standard-count* approach (slate, photographic and videographic) showed no difference in diversity between one another but produced significantly lower estimates than transects analysed using the *MaxN* technique. Shannon‟s *H'*, one possible proportional abundance metric, takes into account both species richness, and evenness (Magurran 1988). The maximum diversity (H_{max}) would occur in a situation where all the species in a community are equally abundant (Traas 2009). Therefore *H'* for a community of *x* species will be lower if that community possesses a few dominant and rare species than if that same community consisted of only evenly represented species (Magurran 1988). When comparing techniques, therefore, differences in *H'* values do not necessarily reflect that one technique is more appropriate than the other but can provide information on the way a particular technique is representing the community in question.

The lower *H'* estimates of slate and *standard-count* transects indicate a lower evenness, while the higher *H'* estimates of *MaxN* transects indicate the converse. These trends are evident when one examines the list of the five most common species sampled. Using the *standard-counts* technique *C. dimidiata*, a usually non-aggregating and evenly distributed species (Kuiter and Tonozuka 2001), is shown to be the most common species. *MaxN* analysis showed that *C. dimidiata* is replaced by *C. weberi*, which usually occurrs in large aggregations in specific areas (Allen 1991). Using the *MaxN* technique, both *C. weberi* and *C. dimidiata* dominated counts. The reason for this discrepancy between *Standard-count* and *MaxN* results can be explained by the *MaxN* technique that selects a single frame as the total count for each species in each replicate. By conducting counts in this manner, it is inevitable that aggregating or shoaling species, such as *C. weberi*, will be more strongly represented because it is almost certain that when the species is detected in a transect, it is detected in large numbers. By contrast, non-aggregating and evenly distributed species, such as *C. dimidiata,* will almost always be represented in low numbers. Similar results were found by

Floros (2010a) who, using the slate roving diver method, found *C. dimidiata* to be more abundant than *C. weberi* on Two-Mile Reef. Counts using the *MaxN* technique will therefore always be biased towards shoaling species in comparison to non-shoaling species when used in conjunction with the transect technique.

When using the *MaxN* technique for the purpose it was developed – conducting fish counts using footage from stationary, usually baited underwater video stations – it is unbiased towards species with particular aggregating/non-aggregating behaviours as even non-aggregating species will aggregate at the bait zone (Priede *et al.* 1994; Willis and Babcock 2000). The technique has also been shown to be successful when conducting UVC video point counts as it reduces the risk of duplicate counts of schooling fishes that may repeatedly enter the stationary survey area (Gledhill *et al.*1996; Francour *et al.*1999, as cited by Tessier *et al.* 2005). Transect surveying requires the observer to move forwards in a single direction at a constant speed to reduce the occurrence of duplicate counts and the associated error (Watson *et al.* 2005; Bennett 2008). Videographic transects in particular afford further reduction of this error as it is possible to track individuals/shoals throughout a video clip such that the large majority of double counts are avoided. Watson *et al.*(2005) used this approach in their UVC study stating that while the use of *MaxN* as a relative density measure for remote techniques avoids repeated counts, it can be assumed that the transect technique avoids duplicate counts.

Based on the differences in *H'* diversity, the representation of the five most common species by *Standard-count* and *MaxN* analysis techniques, and the theoretical considerations drawn for the applicability of the benefits of *MaxN* analysis to transecting, it can be concluded that the *standard-count* technique is the most appropriate, representative method for analysing digital transect footage of diverse reef fishes.

The number of replicates required to achieve an acceptable level of statistical power of results is an essential component to any study design (Samoilys 1997, Utts and Heckard 2007). UVC methods are innately field intensive thus amplifying the trade off between statistical rigour and the maximum number of samples that can be reasonably collected within financial and logistical constraints (Green 1979; Samoilys 1997). When comparing transect techniques the most appropriate technique for future implementation will therefore be the technique requiring the lowest number of replicates to achieve the predetermined power level. This study found that to achieve a statistical power of 80%, videographic transects required the most favourable sample size of 17 replicates, as opposed to 37 and 27 replicates required for slate and photographic transects, respectively. Based on the required sample size to achieve 80% power it is therefore recommended that future studies investigating coral reef fish communities in the iSimangaliso Wetland Park use videographic transects. These studies should attempt to conduct at least 17, but more conservatively 20, replicates at each level of stratification. Sample size is, however, not the only factor determining the time and cost involved in conducting a UVC, as not all techniques will exhibit equal efficiency per replicate. The efficiency of each transect technique was therefore compared.

Many authors have shown that survey time is one of the most important considerations when implementing a UVC study (Bortone *et al.* 1986; 1989). This is relevant to this study both in terms of training time and the time taken to analyse the media. Observer training time for slate transects was triple that of other two transect techniques based on six available divers – 3 buddy pairs. Future research expeditions will likely require adjusted sampling designs based on available field time. The number of dives a diver can safely conduct in a day is limited by residual nitrogen accumulation. Because of this constraint, as available field days decrease the number of divers required to conduct sufficient replicates increases. When conducting slate transects, this trend is associated with an increase in the required number of trained fish identification observers and hence an increase in training time. Transects which record digital footage, such as photographic and videographic transects, are not restricted in the same way as only a single fish identification observer is required to analyse footage. Based on observer training time, photographic or videographic transects are recommended for use in future reef fish transect surveys.

Total analysis time was lowest for slate transects and longest for videographic transects. The low analysis time for slate transects is because no further counts are conducted; once transects are complete, data are simply captured. Digitally-based photographic and videographic footage must be analysed and re-analysed. Total analysis time for photographic transects was shorter than for video transects (*standard-counts* and *MaxN*) as the time taken to analyse still images is shorter than the time taken to analyse continuous video. Based purely on analysis time, slate transects are the most favourable technique while photographic transects are the most favourable digital transect technique.

The ability to identify fishes is a contentious UVC techniques issue (Pelletier *et al.* 2011). The proportion of detected individuals that cannot be identified provides a good proxy for the ability of a digital transect technique to successfully count fishes. A technique that has a larger proportion of individuals that cannot be identified is less efficient than a technique with a lower proportion (Bortone *et al.* 1991, Samoilys and Carlos 2000, Pelletier *et al.* 2011). One of the shortcomings of the slate transect technique is that transects cannot be reviewed (Langlois 2010, Pelletier 2011). Slate transects are therefore the least favourable technique because one cannot estimate the magnitude of identification error and therefore cannot account for it. Of the digital media transect types videographic transects (*standard-counts* and *MaxN*) exhibit a lower proportion of unidentifiable species than photographic transects. This difference is probably due to the continuous nature of video footage which allows the image of a fish to be viewed through a number of frames and also allows for closer inspection of behavioural aspects such as swimming (Watson *et al.* 2005). Photographs do not afford this flexibility and are hindered by obstructions such as reef topography or large fishes/shoals of fishes permanently obscuring other individuals from photos. Videographic transects are therefore the most efficient sampling technique for successfully identifying fishes as they can be reviewed repeatedly and allow greater identification flexibility.

All digital transect techniques used in this study show similar proportions of each ecological group. Epibenthic species dominated either because the transects were best suited to the epibenthic species group or because the fish communities on Two-Mile Reef are dominated by the epibenthic species. Regardless of the reason, no single technique is different from the others indicating that all transects techniques are equally favourable in terms of their affinity for particular ecological groups.

Although the percent agreement approach has been scrutinised as a method of validation in ichthyofaunal ageing studies (Campana 2001), the technique can provide a simple, rapid method of validating richness and abundance estimates from digital UVC media. The results of the present study indicate that, provided a sufficiently experienced observer is available, doubling the time required to analyse digital media for the sake of validation is perhaps not necessary because original estimates of richness and abundance are, for the most part, sufficiently accurate. Where observers analysing digital media are inexperienced, however, validation as implemented in the present study could be in a training and/or quality control component to any study to confirm the probability of accurate estimates.

4.5 Conclusions

To conclude, this study was the first to: compare slate, photographic and videographic techniques for UVC transecting; directly compare *Standard-count* and *MaxN* media analysis techniques propose a method for validating estimates of richness and abundance estimates generated by reviewing digital imagery.

Photographic transects required less replication than slate transects but more replication than videographic transects. In terms of time taken to analyse digital media, photographic transects outperformed videographic transects. The photographic transect technique produced lower abundance and richness estimates, higher variability of estimates and a higher proportion of unidentifiable individuals when compared to videographic transects. The photographic transect technique as implemented in this study is therefore not recommended for future studies. It is, however, recommended that further research be undertaken to develop photographic-based transect methods as the techniques may be valuable in volunteer observer-based monitoring programs where observers are more likely to own photographic cameras than sufficient resolution video cameras.

Videographic transects were shown to outperform the other two techniques in nearly all aspects of the six criteria assessed. The only criterion in which videographic transects under-performed was in the time taken to analyse videographic media. This was outweighed by the benefits of better richness and abundance data, low estimates of variability, the representative diversity observed, and the lower minimum required sample size.

It is recommended that for all future studies investigating the epibenthic coral-reef fishes of high-latitude WIO coral reefs, videographic transects be employed. It is anticipated that this technique will be equally applicable to low-latitude WIO reefs. The most representative description of fish communities was achieved using the *Standard-count* media analysis technique, which it is recommended as the most appropriate analysis approach.

Chapter 5:

Seasonal- and depth-related community dynamics of fish assemblages on a large, high latitude coral reef in the Western Indian Ocean

5.1 Introduction

Many organisms have been shown to exhibit predictable spatial distributions associated with the proximate biotic and physical structure of their habitat (Sebens 1990; Sale 1991; Garpe and Öhman 2003). Coral reef fishes are no different. Distinct fish assemblages have been found between different spatial and temporal zones such as various depths or years (Öhman *et al.* 1997; Wilson *et al.* 2003), or, associated with specific habitat features such as the structural architecture of a reef and the type of benthic community (Bergman *et al.* 2000; Munday 2000). These assemblages can be altered by a complex suite of often species-specific external factors and by phenomena such as ontogenetic- or metamorphosis-related processes (Fulton *et al.* 2001; Feary *et al.* 2007; Lecchini *et al.* 2007).

While the distribution of coral-reef fishes and their relationship to habitat structure have been thoroughly investigated in areas such as the western Pacific Ocean and the Caribbean, fish assemblages in many biogeographic regions still remain poorly studied (Garpe and Öhman 2003). An area yet to be thoroughly investigated is the biogeographic subregion known as the Western Indian Ocean (WIO) which possesses a large variety of reef environments (Sheppard 1987). While the coral-reef fish assemblages of the WIO island states have been quantitatively described (Table 5.1), the assemblages of the East African Coast have, for the most part, been only superficially surveyed (Garpe and Öhman 2003).

One poorly studied area is the north-eastern coast of South Africa that contains the highest latitude coral-reefs in the southern hemisphere (Schleyer 1999). As anthropogenic influences on the marine environment continue to escalate, pristine systems, unaltered by human impacts, are becoming rare (Jackson *et al.* 2001; Friedlander and DeMartini 2002; DeMartini *et al.* 2008). South Africa is fortunate in that many of the coral reefs of the north-east coast have long histories of protection. These relatively undisturbed areas may be considered to be baseline areas for comparison with similar reefs in the region that are unprotected and are exposed to anthropogenic resource pressures (Floros 2010a). While the high-latitude South African reefs are marginal, they are, for all intensive purposes, similar to reefs in lower-latitude WIO areas (Schleyer 1999; Schleyer and Celliers 2003a, b). Given their high-latitude location and their position on the continental shelf, however, ichthyofaunal species compositions could differ from lower-latitude continental areas and WIO island state reefs (Schleyer 1999; Garpe and Öhman 2003).

Table 5.1: Summary of all studies outside of South Africa investigating the full community dynamics of coral reef fishes in the Western Indian Ocean. Studies presented only at symposia are not included.

Study	Study type	Country/ Island	Site description	Depths (m)	Survey method	Families	Species
Letourneur (1996)	Community dynamics	Réunion	Multiple reef flats	<2 m	Roving diver and strip transects	44	217
Chabanet (2002)	Community dynamics	Comoros	Multiple fringing and barrier reefs	<6 m	Strip transects	35	225
Garpe and Öhman (2003)	Community dynamics	Mafia Island	Multiple reefs of various sizes	3–10	Strip transects	56	394
Durville *et al.* (2003)	Community dynamics	Glorieuses Islands	Fragmented fringing reef	0–15	Roving diver	57	332
Gillibrand *et al.* (2007)	Community dynamics	Madagascar	Single fringing reef, Single barrier reef and single patch reef	10–30	Roving diver and timed counts	58	334
Floros (2010a)*	Community dynamics	Mozambique & South Africa	Multiple reefs of various sizes	12–15	Slate roving diver	50	284

* Some aspects of study conducted in South Africa

It has been hypothesised that the high-latitude nature of the South African coral reefs makes them excellent candidates as early-warning systems of climate change trends. Events apparent on the South African coral reefs could precede similar events in lower-latitude areas (Schleyer 1999). Already Schleyer and Celliers (2005) note that temperatures are fast approaching the local coral bleaching threshold, an event which could have significant effects on their ichthyofaunal assemblages (Chabanet 2002). Previous ichthyofaunal surveys on the north-east South African coast have compiled only species checklists (Smith *et al.* 1980; Chater *et al.* 1993; Polack 2007), investigated the dynamics of selected taxa (Chater *et al.* 1995; Currie 2005; Floros 2010c) (Table 5.2), or investigated ichthyofaunal community structure and dynamics in a single depth/ habitat zone (Floros 2010b).

Table 5.2: Summary of all studies investigating fish communities, or aspects of fish communities, on South African coral reefs.

Study	Study type	Site description	Depths	Survey method	Families	Species
Smith *et al.* (1980)	Community checklist	Haphazard sampling of multiple marine habitats	Intertidal–30 m	Various destructive and non-destructive methods	150	732
Chater *et al.* (1993)	Community checklist	Multiple reefs of various sizes	8–45 m	Roving diver and angling	73	399
Chater *et al.* (1995)	Selected species dynamics	Multiple reefs of various sizes	10–40 m	Slate strip transects	13	43
Currie (2005)	Selected taxa dynamics	Multiple patch reefs	10–16 m	Slate point counts	24**	
Polack (2007)	Community checklist	Haphazard sampling of multiple marine habitats	Intertidal–200 m	Various destructive and non-destructive methods	-	1257
Floros (2010a)*	Community dynamics	Multiple reefs of various sizes	12–15 m	Slate roving diver	50	284
Floros (2010b)*	Selected species dynamics	Multiple reefs of various sizes	12–15 m	Slate point counts	12	25
This study	Community dynamicls	Single, large patch reef	6–30 m	Videographic strip transects	41	209

* Some aspects of study conducted in adjacent Mozambique, ** Data are for study-defined taxa

The aim of this chapter was to survey the coral reef assemblages of the area using non-destructive underwater visual census (UVC). As coral reefs generally occur at lower equatorial latitudes, the effect of season in structuring the ichthyofaunal communities of these systems has largely been overlooked. The possible effects of season in structuring high-latitude coral-reef fish communties was examined. As previous replicated research in the area excluded the effect of depth in structuring fish communities by sampling only a narrow depth range (Floros 2010b), depth-related drivers to changes in community structure were investigated. This study examined the relationship between depth, its associated habitat and benthic community characteristics, and the structure of coral reef fish assemblages by categorising the complex reef system into smaller, strongly interacting subsystems based primarily on depth, and secondarily on habitat and benthic community variables. In addition to the assessment of ichthyofaunal communities as a whole, all species were categorised into functional groups to allow for an explanation of group-specific patterns.

5.2 Materials and Methods

5.2.1 Study site

Epibenthic coral reef fish surveys were conducted on Two-Mile Reef in the iSimangaliso Wetland Park, on the north-eastern South African coast (Chapter 2). Two-Mile Reef is a large, accessible, high-latitude patch reef which spans a wide depth range (approximately 6–30 m, Fig. 5.1) and possesses a wide variety of intricate habitats. Two-Mile Reef was selected because its characteristics suggest it can be considered representative of other high-latitude WIO coral reefs (Ault and Johnson 1998).

5.2.2 Sampling methodology

Videographic transecting was accomplished using a DX 1200 HD Camera (© SEA&SEA SUNPAK Co., Ltd.). Transect width was 4.79 m, the width of the camera‟s field of view at a distance of 5 m. Video transects, 50 m in length, were captured at an elevation of 1 m above the reef and using a swimming speed of 6.25 $m.min^{-1}$ that translates into a total time of 8 $min.transect^{-1}$. Total transect area was therefore 239.5 m^2. Planned total dive time was a maximum of 21 minutes for all dives. Video footage was analysed using the *Standard-count* technique. Further transect and fish assemblage details are outlined in Chapter 3.

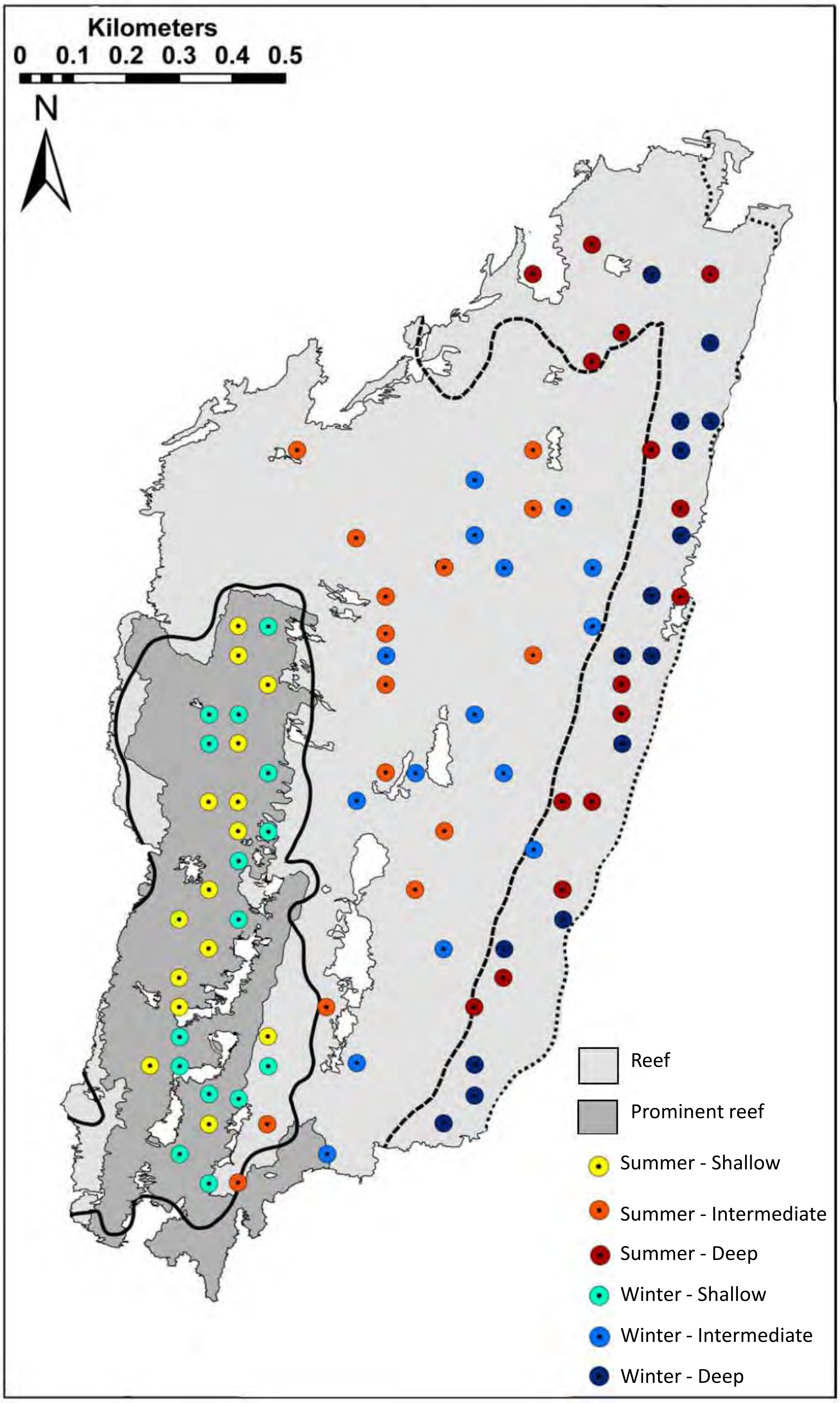


Figure 5.1: Transect start points for shallow (6–14 m), intermediate (14–22 m) and deep (22–30 m) transects conducted during winter and summer on Two-Mile Reef in the Central Reef Complex in the iSimangaliso Wetland Park, South Africa. Depth countour and geological data were obtained from the Marine Geoscience Unit of the Council for Geoscience, South Africa. Spatial reference system: Transverse Mercator (Central Meridian +27.00).

5.2.3 Sampling design

The most suitable sampling design maximises the number of replicates logistically feasible, taking into consideration both the time and funding available, such that there is a realistic trade-off between accuracy and precision (Winer *et al.* 1991). Although it has been shown that 17 videographic transects are required at each level of stratification to achieve a statistical power of 80% (Chapter 4), the maximum number of replicates that were logistically feasible, 15 per stratum, would have resulted in a statistical power of approximately 76% (Chapter 4). This was considered adequate for the present study as Samoilys and Carlos (2000) note that for typical reef fish, which tend to be highly variable, a minimum of ten replicates should be conducted.

A total of 90 videographic transects were conducted over winter and summer and across three depth categories – shallow (6–14 m), intermediate (14–22 m), and deep (22–30 m) – using a balanced sampling design (Table 5.3). Depth categories were selected *a priori* based on dividing the depth extent of Two-Mile Reef into equal categories. Categories were assigned to test the null hypotheses that (1) there is no change in epibenthic coral reef fish community structure from winter to summer, and (2) there is no change in epibenthic coral reef fish community structure from shallow, to intermediate, to deep depths.

Table 5.3: Outline of the balanced sampling design for 90 videographic transects conducted over two seasons and three depth categories on Two-Mile Reef.

Seasons	Seasonal samples	Depths		Depth samples
Winter	45	Shallow	6–14 m	15
		Intermediate	14–22 m	15
		Deep	22–30 m	15
Summer	45	Shallow	6–14 m	15
		Intermediate	14–22 m	15
		Deep	22–30 m	15

Sampling occurred over six days in winter from the 31 July 2010 to 05 August 2010 and over six days in summer from 01 March 2011 to 06 March 2011 (Table 5.4). To minimise the effects of short-term shifts in environmental conditions (e.g. wind, tide, moon phase), a single replicate per depth category was conducted and repeated until all 45 transects were completed

(Table 5.4). Due to diving regulations and the number of divers available, sampling was optimised by devoting one day in each season to sampling only deep sites (Table 5.4).

The randomised allocation of sample sites, quantification of supplementary variables, establishment of trasect dimension and fish assemblage parameters, standardisation pertaining to observers, and transect deployment protocol are described in Chapter 3.

Table 5.4: Sampling dates and study design for transects at Shallow (Sh), Intermediate (I), and Deep (D) in winter and summer on Two-Mile Reef as conducted in this study.

Season	Date	Transects/day	Sampling structure
Winter	31/07/2010	8	I,Sh,D,I,Sh,D,I,Sh
	01/08/2010	8	I,Sh,D,I,Sh,D,I,Sh
	02/08/2010	8	I,Sh,D,I,Sh,D,I*,Sh
	03/08/2010	8	I,Sh,D,I,Sh,D,I,Sh
	04/08/2010	8	I,Sh,D,I,Sh,D,I,Sh
	05/08/2010	6	D,D,D,D,D,I
Summer	01/03/2011	5	D,D,D,D,D
	02/03/2011	8	I,Sh,D,I,Sh,D,I,Sh
	03/03/2011	8	I,Sh,D,I,Sh,D,I,Sh
	04/03/2011	8	I,Sh,D,I,Sh,D,I,Sh
	05/03/2011	8	I,Sh,D,I,Sh,D,I,Sh
	06/03/2011	8	I,Sh,D,I,Sh,D,I,Sh
Total		90	Sh & D (n = 30); I (n = 31)

* Indicates a failed transect

5.2.4 Accounting for supplementary variables

Depth, profile, rugosity, habitat complexity, substrate, current speed and direction, visibility, water temperature, turbidity and time of day were measured, estimated, and/ or categorised for each transect according to the methods outlined in Chapter 3.

To avoid statistical over-fitting, the qualitative variables in habitat complexity and substrate were reduced from five categories to three (Götz 2006).

Although precise transect start times were recorded, time was organised into three categories to determine whether or not time affected any observed patterns. These categories were Category 1 (09:00–10:20), Category 2 (10:20–11:40), and Category 3 (11:40–13:00).

As sampling was conducted by depth across different benthic community types, the benthic community of each transect was qualitatively assessed by methods described by Götz *et al.* (2009). Benthic community categories were assigned by categorising the descriptions of Celliers and Schleyer (2008), and in-field observations from reconnaissance dives, into three categories – coral 1, coral 2 and sponge (Table 5.5).

To assign each transect to a benthic community category, five still-frames were randomly selected from each transect video. Each frame was then assigned a benthic community category. For each transect the benthic community category observed at the highest frequency was selected as the category for that transect. In instances where an even frequency distribution of benthic communities was observed, that transect was re-analysed by adding five more randomly selected frames to the original set until the appropriate benthic community could be determined.

5.2.5 Functional groups

To investigate the community dynamics of different functional groups, species were placed into one of six functional groups based on their diet and feeding behaviour (Pratchet 2005; Wilson *et al.* 2006; Wilson *et al.* 2008; Floros 2010b; Froese and Pauly 2011). Species known to have strong associations with live coral, including obligate coral feeders but excluding facultative coral feeders, were categorised as coral-dependent. The other functional groups were territorial and roving feeders of the epilithic algal matrix (EAM feeders), planktivores, invertivores, piscivores.

5.2.6 Statistical analyses

Identification of suitable predictive factors

As UVC surveys typically have relatively low sample sizes, it is usually necessary to reduce the number of predictor variables where appropriate (Clarke and Gorley 2006). Categorical variables from all 90 transects were therefore ordinated using Principal Component Analysis (PCA), where clustered variables in the 2D ordination space show a strong relationship to one another.

Table 5.5: Revised characteristics of shallow, intermediate and deep depth categories for Two-Mile Reef in terms of their associated habitat complexity, benthic community (Celliers and Schleyer 2008) and algal cover (Anderson *et al.* 2005). For reference images of habitat complexity categories see Appendix 1.

Depth category	Depth range	Habitat complexity	Benthic community	Algal community characteristics
Shallow	6–14 m	Highly complex: detailed habitat structure. Refuge habitat extensively available - nooks and crannies within and between caves and crevasses extensive.	Coral 1: Extensive, diverse coral cover. High abundance of Scleractinia and Alcyonacea. Characteristic species = e.g. *Pocillopora, Acropora, Porites and Sacrophyton spp.*	Highest diversity, biomass and percent cover*. Extensive patches of algal turf and few foliose algal species.
Intermediate	14–22 m	Moderately complex: detailed habitat structure. Refuge habitat readily available. Large complex areas interspersed with lower complexity areas.	Coral 2: Moderately extensive and diverse coral cover. Abundant Scleractinia and Alcyonacea. Same characteristic species as Coral 1 but *Pachyseris, Sinularia* and *Sarcophyton spp.* more frequent.	Intermediate diversity, biomass and percent cover*. Biotic cover dominated by algal turfs.
Deep	22–30 m	Homogenous: Undetailed habitat structure. Comparatively flat and barren with minimal refuge habitat for fishes. Large areas of rubble present.	Sponge: Sparse living benthic community. Only Infrequent corals such as *Porites* and *Pachyseris* spp. .Living cover dominated by gorgonians and sponges.	Lowest diversity, biomass and percent cover*. Bare substrate present with patches of turf algae interspersed with occasional foliose individuals.

* For specific data see Anderson *et al.* (2005)

Bray–Curtis similarity matrices were calculated using root-root transformed fish assemblage data (Bray and Curtis 1957). The matrices were ordinated using non-metric Multi-Dimensional Scaling (MDS) (Shepard 1962; Kruskal 1964; Field *et al.* 1982) to determine if fish assemblages showed patterns relating to the variable in question. If no relationship was evident, the categorical variable was dropped as a factor. Where possible relationships were evident, the representative variable was retained for further examination and stratified for subsequent analyses.

General community characteristics

Proportional family abundances were graphically displayed as pie graphs and associated with total and within-family richness data for visual comparison at all levels of stratification. Ranked species abundance (dominance) curves were produced by ranking the abundance of each species and expressing ranked abundance as a percentage of the total abundance of all taxa. To facilitate better visualisation of dominant species, species ranks were log-transformed (Clarke and Warwick 2001).

Identification of fish communities

The total abundance for transect i was calculated as the sum of all individuals j:

$$N_i = \sum_j n_{ij}$$

Richness for each transect was calculated as the total number of observed species in each transect.

Diversity of each transetc was calculated using Shannon‟s diversity index, Shannon‟s *H'* (Shannon and Weaver 1949), calculated as

$$H_i' = -\sum_j p_{ij} \ln p_{ij}$$

where $p_i = (n_{ij}/N_j)$, the proportional abundance of the j^{th} species in transect ij, with n_{ij} being the abundance of species i in transect j and where N_i.

Univariate assessment of fish assemblages

Univariate analyses of fish assemblage data were conducted at all levels of stratification. Abundance, richness and diversity data required root-root transformation at certain levels of stratification before the assumptions of normality and homoscedacity were met. Normality and homoscedacity of residuals was tested using Shapiro-Wilk‟s normality test and Levene‟s test respectively. Treatment and interaction effects of season and depth on abundance, richness and diversity were assessed using a two-way ANOVA. Tukey‟s HSD test was used for *post hoc* pairwise comparisons if a significant difference was noted.

Multivariate assessment of fish assemblages

Multivariate analyses of fish assemblage data were conducted at all levels of stratification. Bray–Curtis similarity matrices were constructed for transect-specific abundance, richness and diversity data. Matrices were ordinated using MDS and an analysis of similarity (ANOSIM) (Clarke and Green 1988) was used to test for differences in community structure between categories at all levels of stratification. The MDS visualised the samples while ANOSIM, a non-parametric permutation procedure, was used to test differences in (rank) similarities underlying the similarity data (Clarke and Warwick 2001).

Observable differences evident using both the univariate and multivariate approaches was used as the proxy to identify discrete community structures within the fish assemblage data from all 90 transects.

Multivariate dynamics of fish communities

The similarity percentages (SIMPER) routine, described by Clarke and Warwick (2001), was used to determine the relative contribution of individual species to within-community similarity and between-community dissimilarity to ascertain which species were the dominant contributors to the observed clusters. An extension of the SIMPER routine was used to determine those species responsible for discriminating between community structures at all levels of stratification. All 209 species were included in analyses.

No species or groups of species exhibited disproportionately large contributions to between-category dissimilarity. The two most appropriate discriminating species were therefore selected for each pairwise comparison according to the criteria outlined by Clarke and Gorley

(2006). These were (1) a low standard deviation resulting in a favourable average contribution: standard deviation ratio, and (2) a comparatively high contribution to between-category dissimilarities.

To assess which combination of environmental variables best explained patterns in fish communities, Spearman"s rank correlations (ρ_s) between environmental and fish community matrices were calculated using the BEST analysis with the BIOENV algorithm (Clarke and Warwick 2001). BIOENV permutation tests were used to test the null hypothesis that there was no agreement in observed multivariate patterns.

Dynamics of functional groups

The contribution of functional groups to observed temporal and spatial differences between communities was assessed using the SIMPER routine and calculated as the summed contribution of each species to overall functional group contribution.

Treatment and interaction effects of season and depth on all functional groups were assessed using Two-Way ANOVA of root-root transformed abundance data for each functional group. Significant differences were investigated *post hoc* using a Tukey"s HSD test.

The open source statistical environment in R 2.11.0 (© R development core team 2010), Excel 2007 (© Microsoft Office), or PRIMER V6.0 (© Plymouth Marine Laboratory) were used to conduct all analyses.

5.3 Results

5.3.1 Identification of suitable predictive factors

Principal Component Analysis (PCA) of all categorical variables accounted for 56% of the cumulative variation in the first two principal components (PCs). Factor loadings indicated that there was a relationship between categorical depth, habitat complexity and benthic community data. Season, time, divers, and substrate categories showed no relationship to any other categorical factor.

As the effect of season on ichthyofaunal communities was of interest, season was retained as a factor in all analyses.

Due to the relationship between depth, habitat complexity and benthic community, the ease of conducting balanced sampling by depth categories, and the availability of continuous depth data, analyses were stratified by categorical depth to holistically represent the three categorical factors depth, habitat complexity and benthic community. A revised summary of the characteristics of each depth category is presented in Table 5.5.

Divers and time categories were implicitly accounted for within the sampling design of the study by systematically rotating pairs of divers conducting transects in each depth category during a narrow, mid-morning time window (Chapter 3). MDS ordination confirmed that divers, time and substrate could not be associated with any observed patterns in fish assemblages (Fig. 5.2). As divers were sufficiently randomised, and time sufficiently restricted, these categories were excluded from analyses.

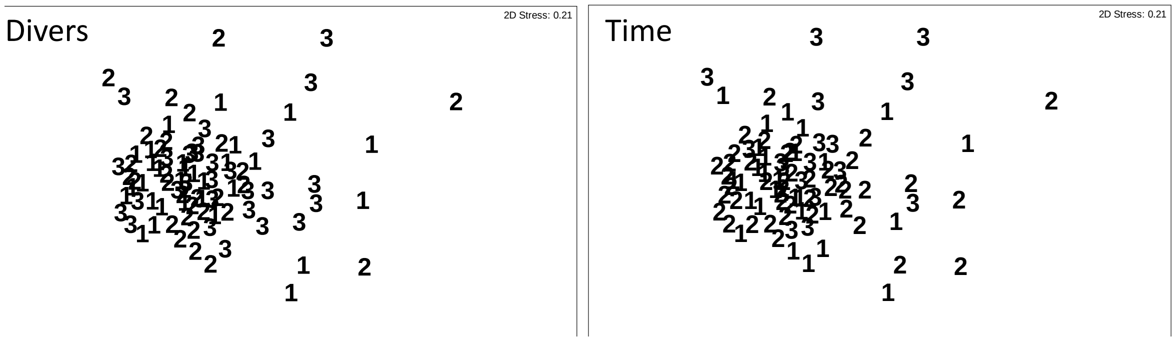


Figure 5.2: MDS ordination of 90 videographic transects, conducted on Two-Mile Reeef, with superimposed divers and time categories.

Although algal cover was not quantified during the present study, the algal communities of Two-Mile Reef have been previously examined (Anderson *et al.* 2005). Anderson *et al.* (2005) found that from the shallowest sections of the reef to 26 m the algal communities on Two-Mile reef are dominated by small red turf algae interspersed with larger, upright individuals from other taxa. Diversity, biomass and percentage algal cover on Two-Mile Reef decreases significantly with depth, possibly due to lower light penetration or different grazing patterns (Anderson *et al.* 2005). As algal cover shows a gradient with depth, the factor can be included in the description of depth categories in the present study as shallow sites have been shown to possess the greatest algal biomass and cover followed by intermediate and then deep sites (Anderson *et al.* 2005) (Table 5.5).

Only Category 1 substrate was encountered for all replicates. Substrate was therefore excluded from analyses.

All analyses were stratified by season and by depth resulting in eight levels of comparison, specifically; season (all depths pooled), depth (seasons pooled), shallow depths in winter, shallow depths in summer, intermediate depths in winter, intermediate depths in summer, deep depths in winter, and deep depths in summer.

5.3.2 General community characteristics

Family groups

A total of 18172 fish from 209 species and 41 families were identified from the videographic transects (Fig. 5.3a).

No biogeographic trends were evident in number of families from winter to summer (Fig. 5.3b-g).

All depth categories in both seasons were dominated by pomacentrids except for the deep depth category in winter when the serranids dominated (Fig. 5.3f). Five families contributed to >3% abundance at all levels of stratification. These were the Pomacentridae, Acanthuridae, Serranidae, Labridae and Chaetodontidae (Fig. 5.3).The Labridae consistently displayed the highest species richness in both seasons over all depths with a total of 37 species encountered during the study and twice as many as the two next most rich families, the Acanthuridae and Chaetodontidae, with 17 species each (Fig. 5.3a).

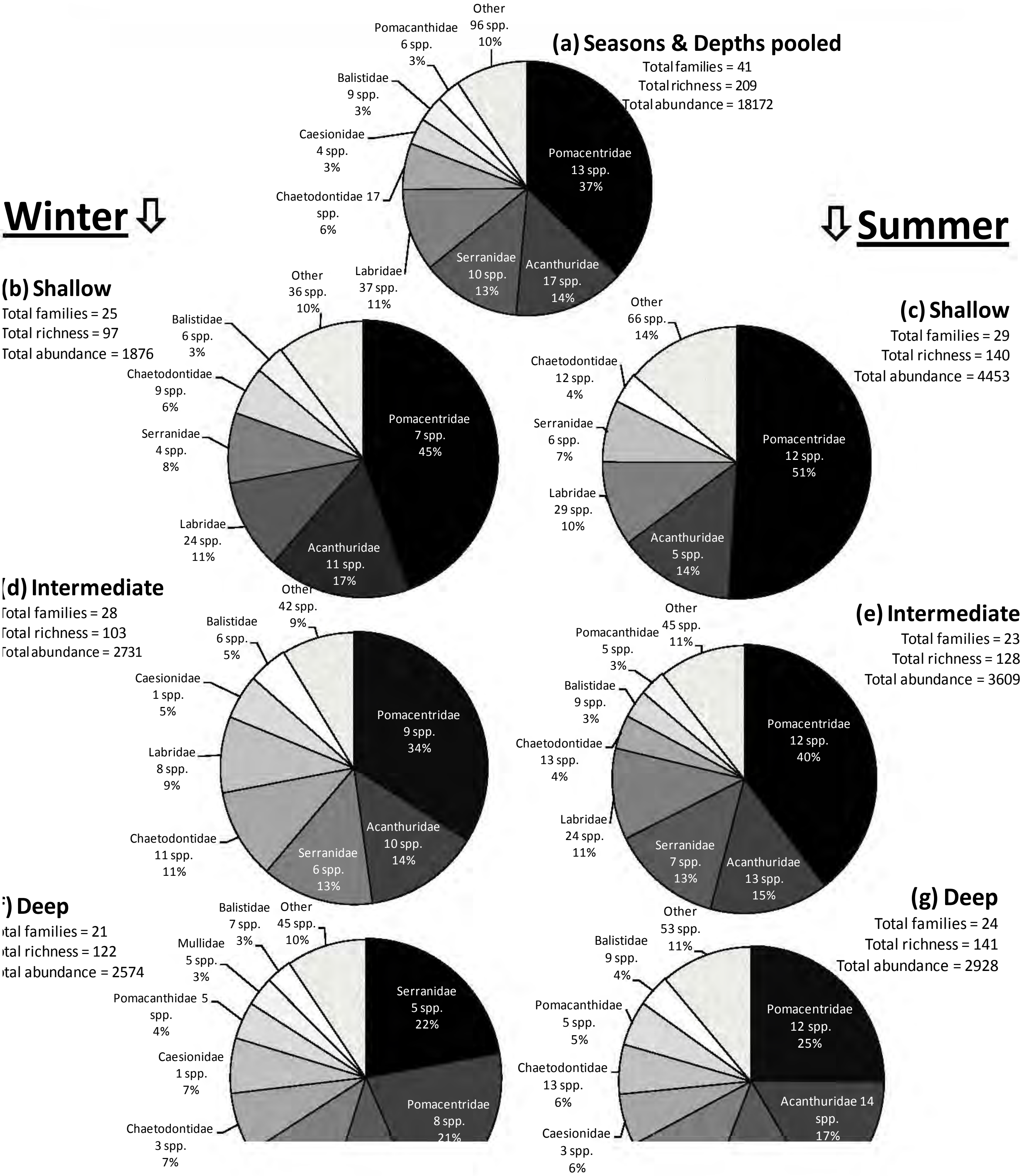


Figure 5.3: Families contributing >3% to total abundance on Two-Mile Reef. The remaining species are categorised as 'other' and may vary between charts. Percentages refer to % abundance for respective families.

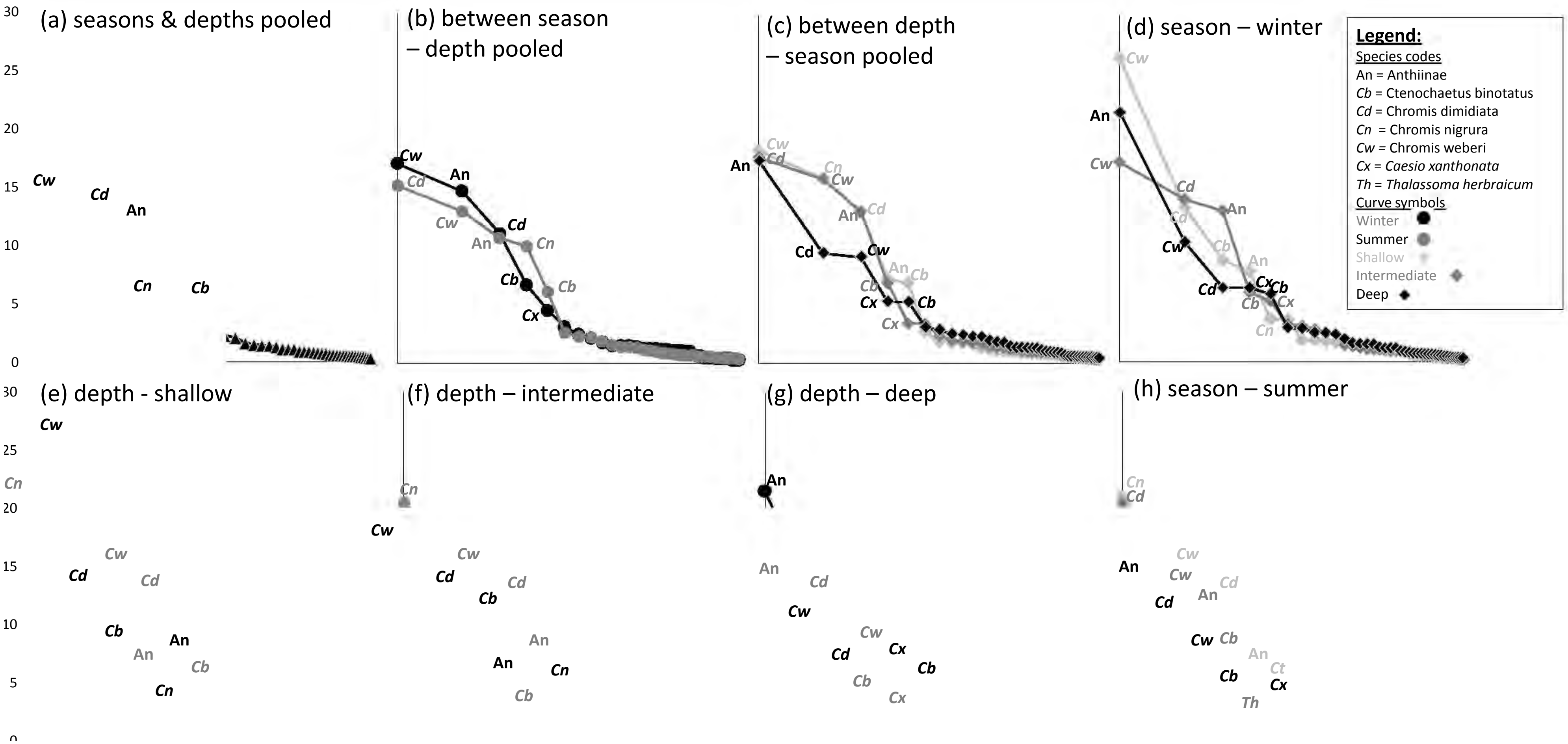


Figure 5.4: Ranked species abundance, or dominance, plots of fishes occurring on Two-Mile Reef indicating the five most dominant species by log-transformed rank in both seasons for all depths. Species ranks are log-transformed and truncated at 1.6 for visual interpretation. Only those species contributing at least 4% to total abundance at one or more levels of stratification are named in the legend.

Species dominance

Dominance plots indicate that at all levels of stratification only five species exhibited >4% dominance indicating no biogeographic shifts to a disproportionately dominant species in a particular season or at a particular depth. *Chromis weberi, C. dimidiata, C. nigrura*, and the Anthiinae consistently comprised the most dominant species while, *Ctenochaetus binotatus* and *Caesio xanthonata* comprised the remainder of those species frequently exhibiting >4% dominance (Fig. 5.4a-h). *Thalassoma herbaicum* appears only once as the fifth most dominant species at intermediate depths in summer (Fig. 5.4h).

Trends within seasonal data were that shallow and intermediate depths exhibited similar rank-dominance curves for pooled seasons and during summer (Fig. 5.4c, d), but intermediate and deep depths were similar during winter (Fig. 5.4h).
Shallow and intermediate depths were dominated by pomacentrids during both seasons but showed a change in the dominant species; *C. nigrura* dominated during winter while *C. weberi* dominated during summer (Fig. 5.4e, f). Deep depths were dominated by the Anthiinae and showed no change in species dominance between seasons.

5.3.3 Identification of fish communities

Univariate analysis of abundance, richness and diversity

Abundance

Spatial representation of the relative abundances of fishes for each transect conducted during the study are displayed in Fig. 5.6.

The effect of season had a significant effect on abundance ($F_{1,84} = 14.60$, $p<0.01$, Fig. 5.5a, Table 5.6) with abundance higher during summer at shallow and intermediate depths, but not at deep depths (Tukey‟s HSD test, $p<0.05$).

Overall depth did not have a significant effect on abundance ($F_{2,84} = 1.27$, $p = 0.29$, Fig. 5.5a, Table 5.6) despite abundance being significantly higher at shallow depths than at deep depths during summer (Tukey‟s HSD test, $p<0.05$).

Although only marginally, the interaction between season and depth on abundance was significant ($F_{2,84} = 3.24$, $p = 0.045$, Fig. 5.5a, Table 5.6).

Richness

Spatial representation of the relative richness"s of fishes for each transect conducted during the study are displayed in Fig. 7.

The effect of season had a significant effect on richness ($F_{1,84} = 13.39$, $p<0.01$, Fig. 5.5b, Table 5.6) with abundance higher during summer at shallow depths but not at intermediate and deep depths (Tukey"s HSD test, $p<0.05$).

Overall depth did not have a significant effect richness ($F_{2,84} = 1.27$, $p = 0.29$, Fig. 5.5b, Table 5.6), despite richness being significantly higher at shallow depths than at deep depths during summer (Tukey"s HSD test, $p<0.05$).

The interaction between season and depth on richness was not significant ($F_{2,84} = 1.64$, $p = 0.20$, Fig. 5.5b, Table 5.6)

Diversity

Spatial representation of the relative diversity of fishes for each transect conducted during the study are displayed in Fig. 5.8.

Neither season ($F_{2,84} = 2.67$, $p = 0.12$, Fig. 5.6c, Table 5.6), depth ($F_{2,84} = 0.78$, $p = 0.46$, Fig. 5.5c, Table 5.6), nor the interaction between season and depth ($F_{2,84} = 0.10$, $p = 0.90$, Fig. 5.5c, Table 5.6) had a significant effect on diversity.

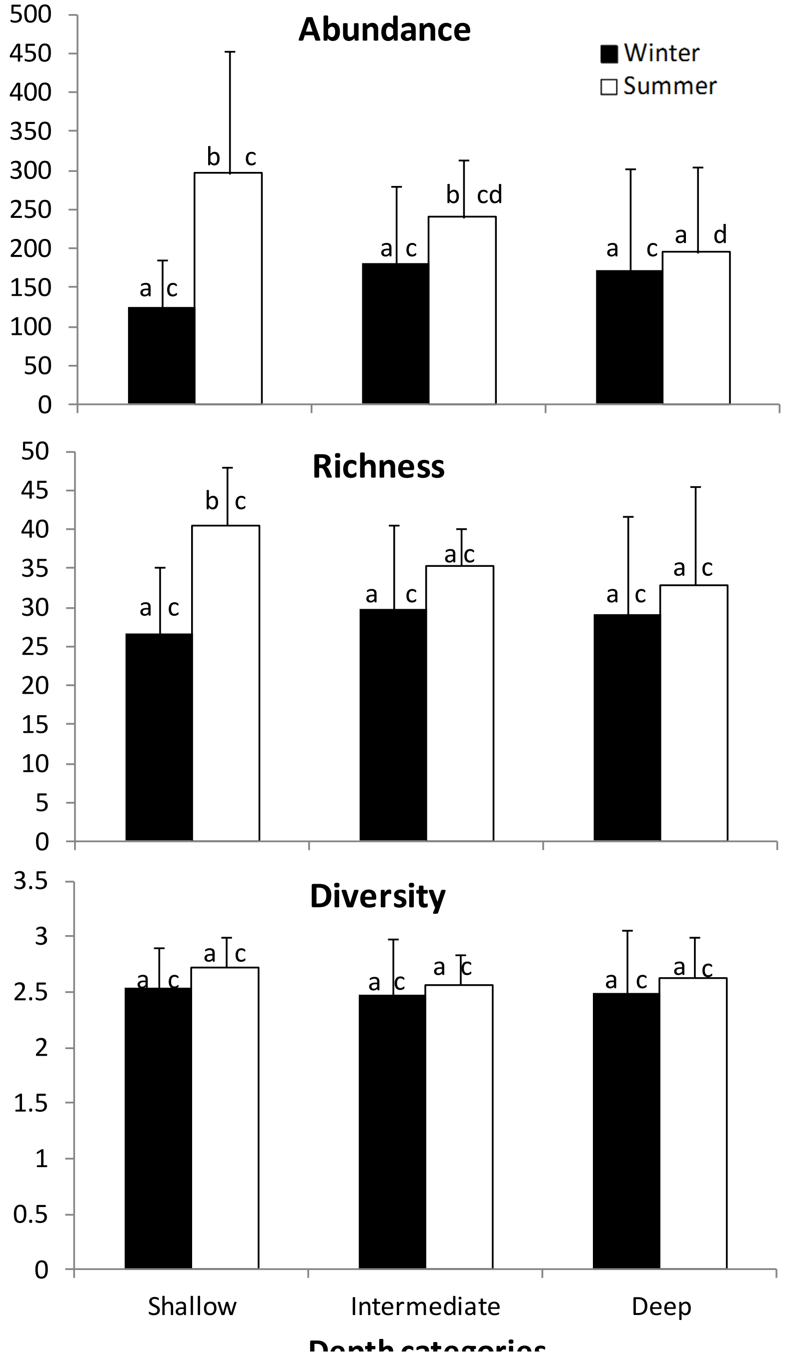


Figure 5.5: Abundance, richness and diversity of reef fishes for each level of stratification on Two-Mile Reef as a function of season and depth. Statistical results from Two-way Analysis of Variance presented in Table 5.6. Common superscripts depict statistically homogenous groups ($p>0.05$). Untransformed data are displayed while the analysis of abundance and richness was conducted using root-root transformed data.

Table 5.6: Results from Two-way Analysis of Variance on abundance, richness and diversity as a function of season and depth on Two-Mile Reef. *Post hoc* pairwise comparisons were conducted using Tukey's HSD test. Significant differences (p<0.05) are indicated with bold text.

	Season		Depth		Season × Depth		Pairwise comparisons						
	$F_{1,84}$	p	$F_{2,84}$	p	$F_{2,84}$	p	Depth pooled	Season pooled	Winter (W)	Summer (S)	Shallow (Sh)	Intermediate (I)	Deep (D)
Abundance	14.60	**<0.01**	1.27	0.29	3.24	**0.03**	**W<S**	D=I=Sh	D=I=Sh	D=I,I=Sh,**D<Sh**	**W<S**	**W<S**	W=S
Richness	13.39	**<0.01**	0.78	0.46	1.64	0.2	**W<S**	D=I=Sh	D=I=Sh	D=I,I=Sh,**D<Sh**	**W<S**	W=S	W=S
Diversity	2.67	0.12	0.54	0.58	0.10	0.90	W=S	D=I=Sh	D=I=Sh	D=I=Sh	W=S	W=S	W=S

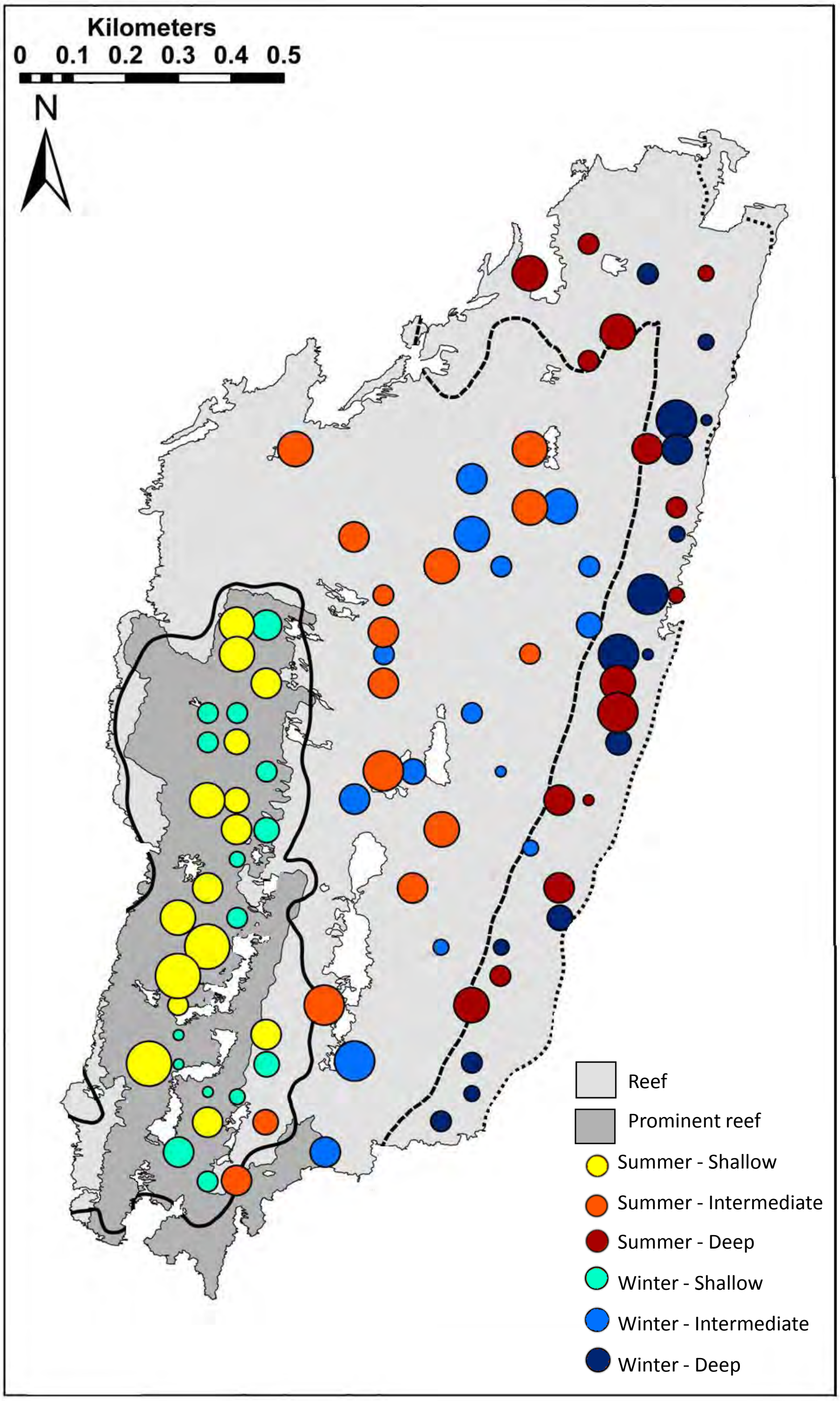


Figure 5.6: Ichthyofaunal abundance observed at shallow (6–14 m), intermediate (14–22 m) and deep (22–30 m) transects conducted during winter and summer on Two-Mile Reef in the Central Reef Complex in the iSimangaliso Wetland Park, South Africa. Bubble size is proportional to observed abundance for each transect. Depth contour and geological data were obtained from the Marine Geoscience Unit of the Council for Geoscience, South Africa. Spatial reference system: Transverse Mercator (Central Meridian +27.00).

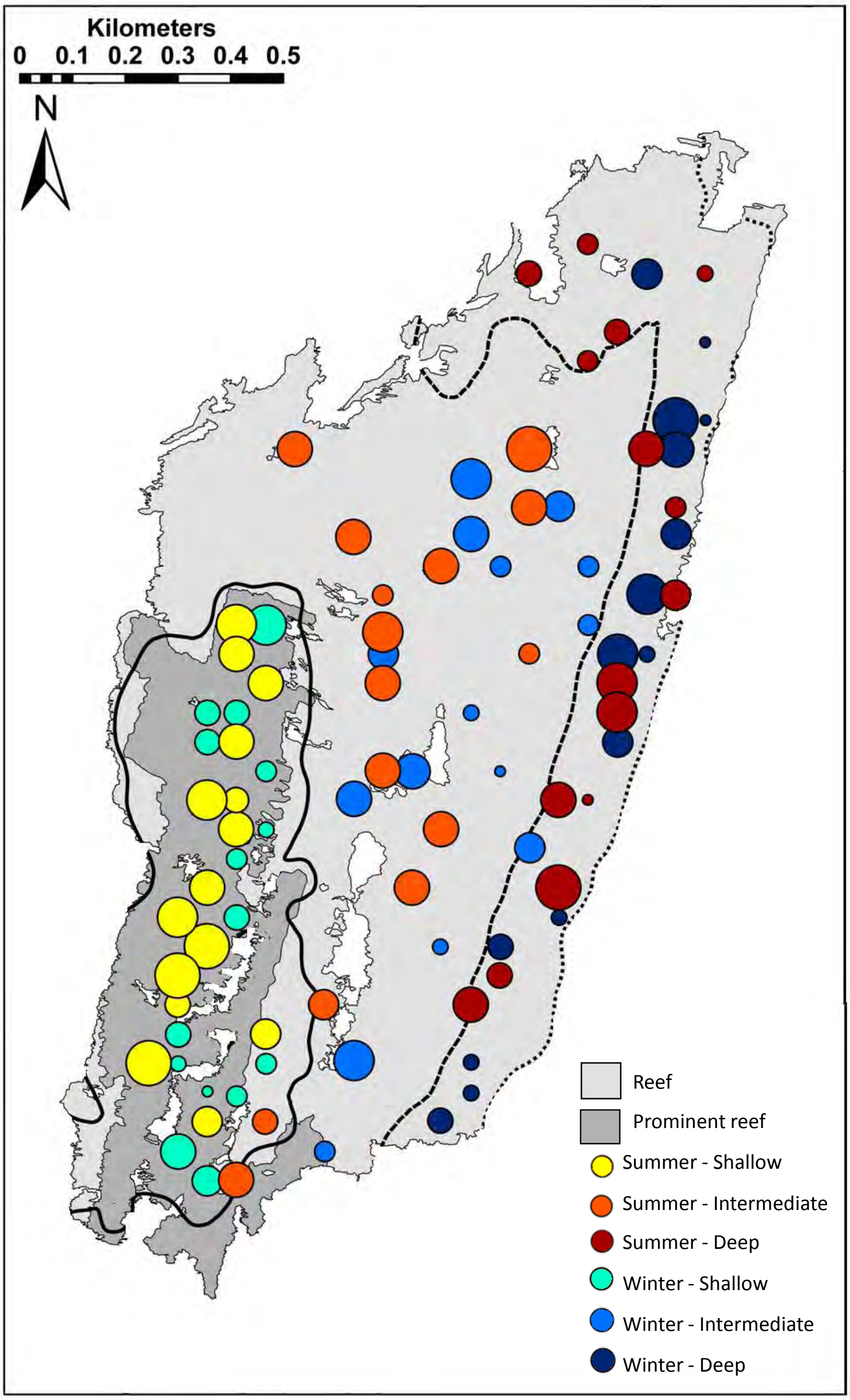


Figure 5.7: Ichthyofaunal richness observed at shallow (6–14 m), intermediate (14–22 m) and deep (22–30 m) transects conducted during winter and summer on Two-Mile Reef in the Central Reef Complex in the iSimangaliso Wetland Park, South Africa. Bubble size is proportional to observed richness for each transect. Depth contour and geological data were obtained from the Marine Geoscience Unit of the Council for Geoscience, South Africa. Spatial reference system: Transverse Mercator (Central Meridian +27.00).

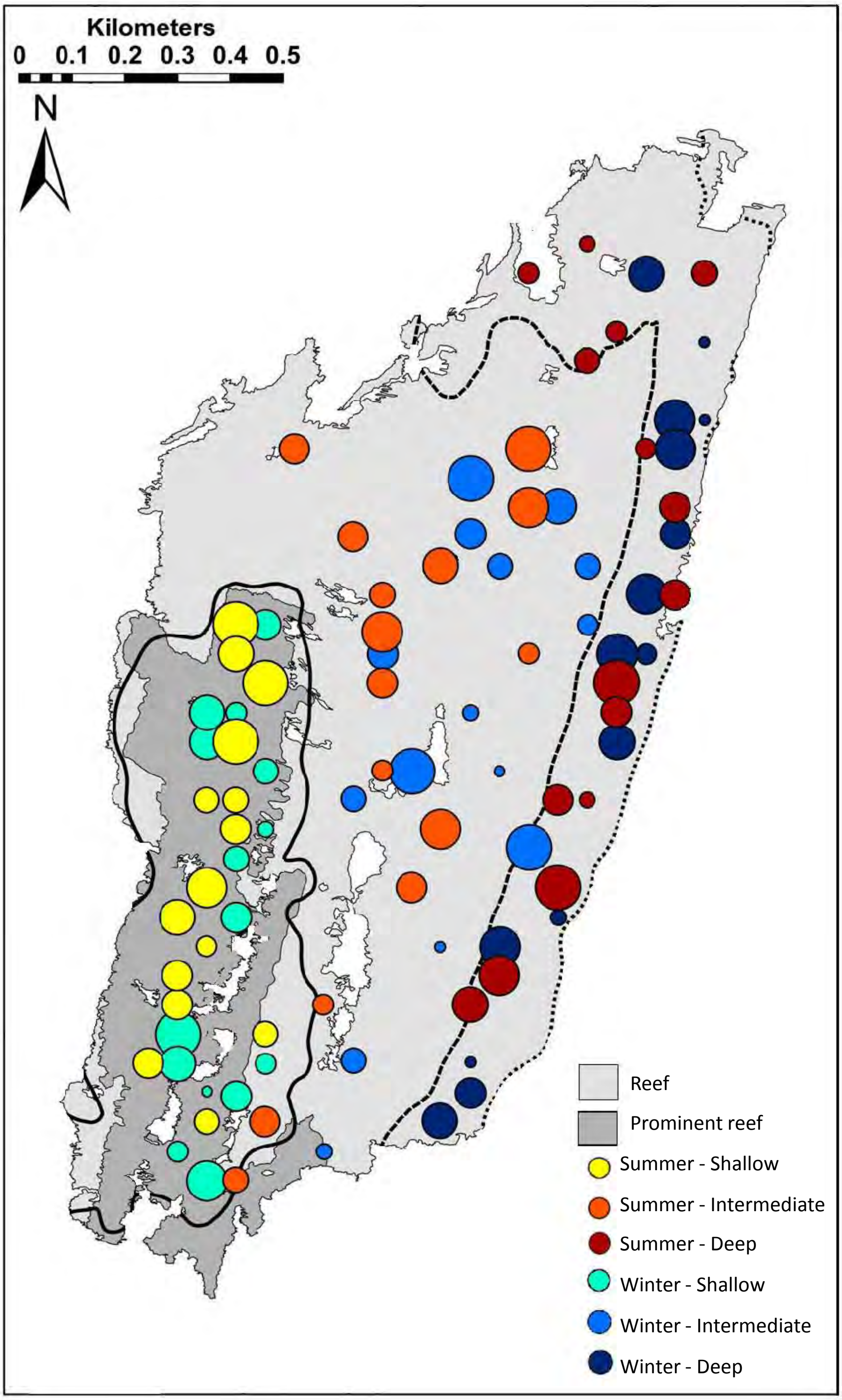


Figure 5.8: Ichthyofaunal diversity, Shannons *H'*, observed at shallow (6–14 m), intermediate (14–22 m) and deep (22–30 m) transects conducted during winter and summer on Two-Mile Reef in the Central Reef Complex in the iSimangaliso Wetland Park, South Africa. Bubble size is proportional to observed diversity for each transect. Depth contour and geological data were obtained from the Marine Geoscience Unit of the Council for Geoscience, South Africa. Spatial reference system: Transverse Mercator (Central Meridian +27.00).

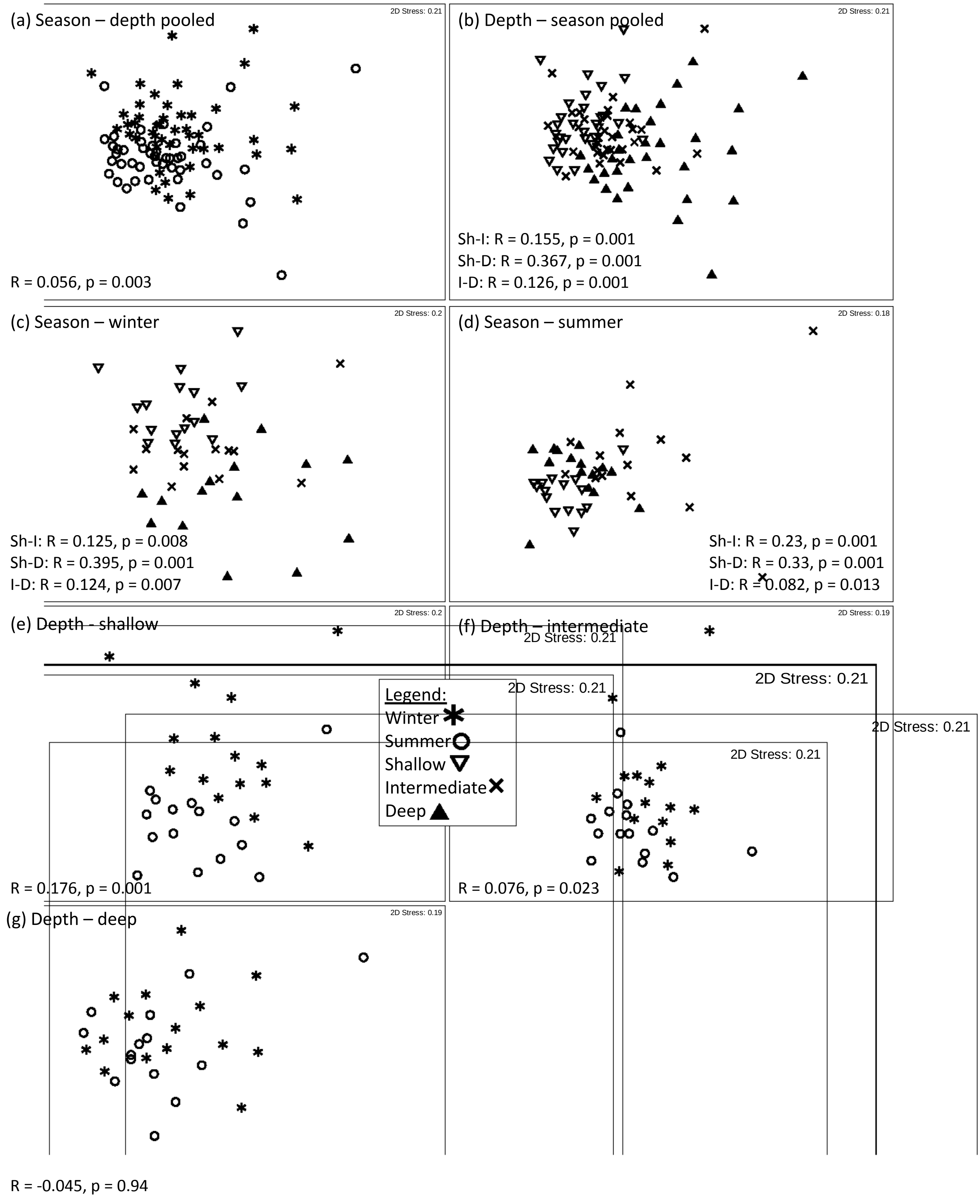


Figure 5.9: MDS ordination of the Bray–Curtis similarity matrices illustrating relationships between depth and season. ANOSIM summary statistics for fish assemblages on Two-Mile Reef provided for each depth-season combination.

Multivariate assessment of fish assemblages

MDS ordination revealed that there were differences in fish assemblage structure between seasons and depths that were confirmed using ANOSIM. There were significant differences in fish assemblages between seasons and depths at all levels comparison (ANOSIM, R >0. 2, p<0.05) except between seasons at deep depths (Fig. 5.9).

5.3.4 Multivariate dynamics of fish communities

Within-community similarity

Seasonal similarity was high for both winter and summer at all levels of stratification (Table 5.7). A trend of decreasing within-category similarity with increasing depth was evident in that shallow depths that had the greatest within-category similarity (48.65% and 52. 37% for winter and summer, respectively), while deep depths exhibited the lowest within-category similarity (39.48% and 37.74%, respectively) (Table 5.7).

Depth similarity was high for both shallow and intermediate depths (always >46% similarity) but lower for deep depths (<40% similarity) in both seasons (Table 5.7).

No species showed >13% contribution to the within-category similarity and no species showed a disproportionately large contribution to within-category similarity.

Between-community dissimilarity

Dissimilarity for shallow and intermediate depths was high between seasons (52.97%) and deep depths (60.66%) between seasons (Table 5.8).

Depth dissimilarity was smallest when comparing shallow and intermediate depths (54.42% and 51.45% in winter and summer, respectively) and greatest when comparing shallow and deep depths (63.34% and 61.93% in winter and summer, respectively) suggesting a trend of increasing dissimilarity with increasing difference in depth (Table 5.8).

Table 5.7: Similarity summary statistics from the SIMPER analysis for all levels of stratification on Two-Mile Reef. The two taxa with the highest percentage contribution to within-category similarity for each category are presented. Seasons = Winter (W) and Summer (S), Depths = Shallow (Sh), Intermediate (I) and Deep (D).

Stratification category	Within-category similarity		Similarity species	% Contribution
Season – depth pooled	W = 42.34 S = 44.01	W:	*Ctenochaetus binotatus*	9.15
			Chromis weberi	7.96
		S:	*Ctenochaetus binotatus*	7.26
			Chromis dimidiata	7.17
Depth – season pooled	Sh = 48.71 I = 47.68 D = 38.99	Sh:	*Chromis weberi*	9.11
			Ctenochaetus binotatus	8.50
		I:	*Chromis weberi*	9.01
			Ctenochaetus binotatus	8.81
		D:	Anthiinae	10.32
			Sufflamen chrysopterus	7.87
Season – Winter	Sh = 48.65 I = 46.85 D = 39.48	Sh:	*Chromis weberi*	10.58
			Ctenochaetus binotatus	10.21
		I:	*Ctenochaetus binotatus*	9.55
			Chromis weberi	9.39
		D:	Anthiinae	12.46
			Sufflamen chrysopterus	7.40
Season – Summer	Sh = 52.37 I = 49.88 D = 37.74	Sh:	*Chromis weberi*	7.39
			Chromis nigrura	7.10
		I:	*Chromis dimidiata*	8.57
			Chromis weberi	8.35
		D:	*Sufflamen chrysopterus*	8.44
			Anthiinae	8.42
Depth – Shallow	W = 48.65 S = 52.37	W:	*Chromis weberi*	10.58
			Ctenochaetus binotatus	10.21
		S:	*Chromis weberi*	7.39
			Chromis nigrura	7.10
Depth – Intermediate	W = 46.85 S = 49.88	W:	*Ctenochaetus binotatus*	9.55
			Chromis weberi	9.39
		S:	*Chromis dimidiata*	8.57
			Chromis weberi	8.35
Depth – Deep	W = 39.48 S = 37.74	W:	Anthiinae	12.46
			Sufflamen chrysopterus	7.40
		S:	*Sufflamen chrysopterus*	8.44
			Anthiinae	8.42

Table 5.8: Dissimilarity summary statistics for the SIMPER analysis for all levels of stratification on Two-Mile Reef. Two primary discriminating species (or taxa), as determined by the SIMPER routine, are presented with dissimilarity contribution (Diss): standard deviation (SD) ratio and percentage contribution to dissimilarity results.

Stratification category	Between-category dissimilarity	Discriminating Species			
			Species	Diss:SD	% contribution
Between season – depth pooled	W – S = 58.9		*Chromis dimidiata*	1.05	2.35
			Anthiinae	1.04	2.26
Between depth – season pooled	S – I = 53.83 S –D = 63.20 I – D = 59.1	S-I:	*Chromis nigrura*	1.32	3.12
			Achanthurus leucosternon	1.38	1.99
		S-D:	*Chromis nigrura*	1.34	2.95
			Acanthurus leucosternon	1.49	1.97
		I-D:	*Chromis weberi*	1.17	2.82
			Chromis dimidiata	1.07	2.71
Season – Winter	Sh – I = 54.42 Sh – D = 63.34 I – D = 58.91	S-I:	Anthiinae	1.26	3.13
			Achanthurus leucosternon	1.43	2.17
		S-D:	Anthiinae	1.30	2.83
			Acanthurus leucosternon	1.49	1.97
		I-D:	*Chromis weberi*	1.18	3.24
			Chromis dimidiata	1.20	1.77
Season – Summer	Sh – I = 51.45 Sh – D = 61.93 I – D = 58.28	S-I:	*Chromis nigrura*	1.83	3.72
			Acanthurus leucosternon	1.59	1.91
		S-D:	*Chromis nigrura*	2.11	3.93
			Acanthurus leucosternon	1.61	1.84
		I-D:	*Chromis weberi*	1.17	2.44
			Achanthurus tennenti	1.20	1.71
Depth – Shallow	W – S = 52.97		*Chromis nigrura*	1.70	4.01
			Anthiinae	1.24	2.36
Depth – Intermediate	W – S = 52.97		Anthiinae	1.05	2.46
			Ctenochaetus strigosus	1.28	2.02
Depth – Deep	W – S = 60.66		*Chromis dimidiata*	1.12	2.71
			Chromis weberi	1.21	2.58

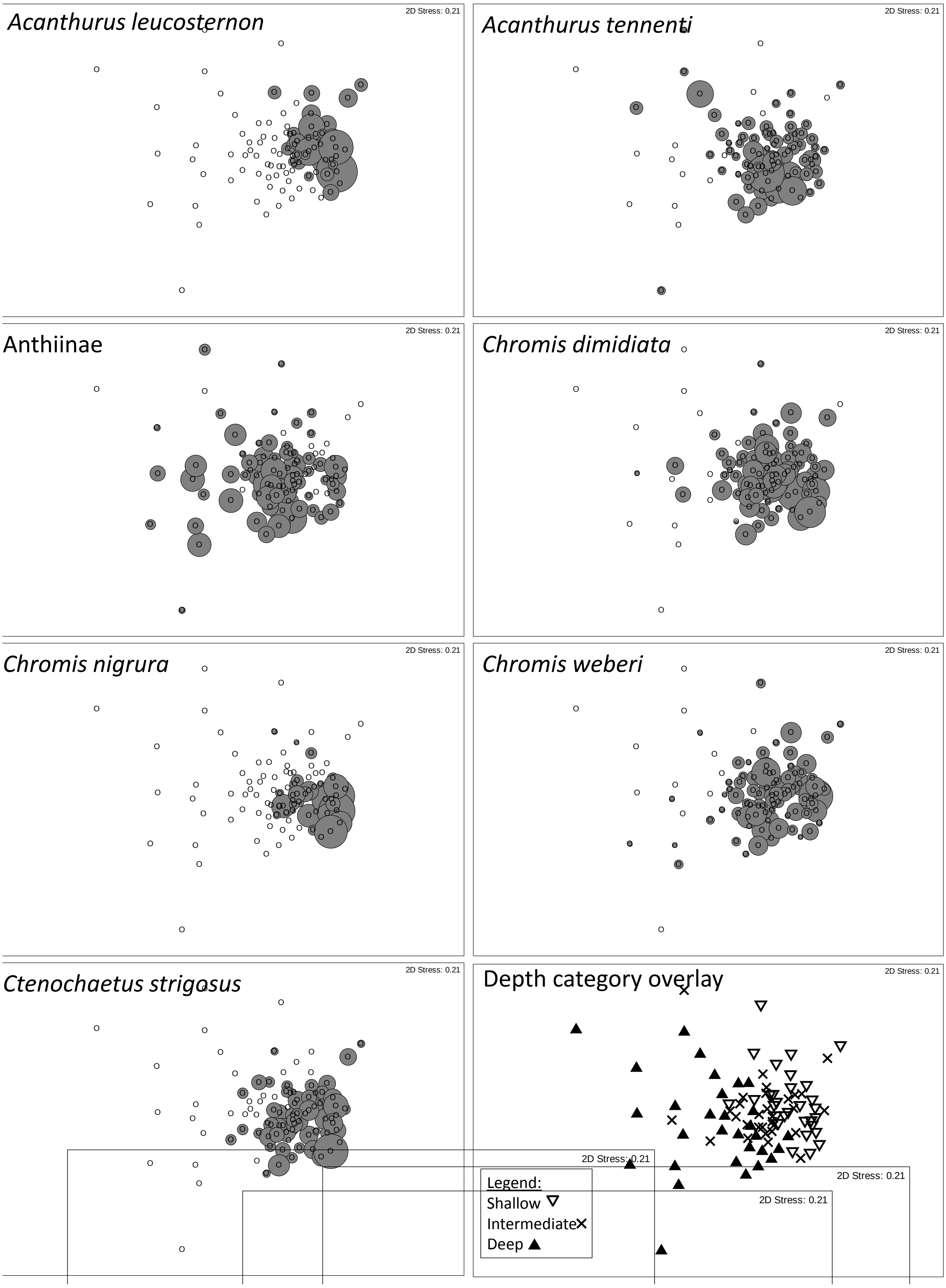


Figure 5.10: MDS ordination of 90 videographic transects with superimposed relative abundances of seven discriminating species for Two-Mile Reef as determined by SIMPER analysis (Table 5.8). Sizes of bubbles indicate relative abundance for that particular species, but are not comparable between species, while empty bubbles represent the ordinated position of transects with inconsequential abundance for that species. An overlay of all depth categories to the spatial distribution of transects within the plot is also illustrated.

Discriminating taxa

Although no single taxa contributed >3% to spatial and temporal differences, a total of seven taxa comprised the top two discriminating species (or taxa) for all between-category comparisons. These were *Acanthurus leucosternon, A. tennenti,* Anthiinae, *C. dimidiata, C. nigrura, C. weberi* and *Ctenochaetus strigosus* (Table 5.8). MDS plots of these seven taxa show that areas of higher counts are variable but for the most part occur to the central-right of the ordination space (Fig. 5.11a–f). *Acanthurus tennenti* and the Anthiinae exhibited the greatest spatial variation in counts within plots, with relatively large counts occurring throughout the ordination space. An overlay of depth categories applied to the ordinated spatial distribution of transects shows deep transects are to the left, while shallow and intermediate transects are clustered to the central-right (Fig. 5.10e). This pattern shows that for the seven discriminating species depth is the variable responsible for structuring. High *A. leucosternon, C. dimidiata, C. nigrura, C. weberi* and *C. strigosus* counts are clustered around shallow and intermediate depth transects (Fig. 5.10a, d–g). *A. tennenti* and Anthiinae abundance, however, did not appear to differ between depth categories (Fig. 5.10b, c).

5.3.5 Relating community-level trends to environmental variables

Profile and rugosity were strongly correlated ($R^2 = 0.86$, $p < 0.01$). Bennett (2008) found a similar result and concluded that of the two measures rugosity provided more desirable information. Profile was therefore excluded from the BIO-ENV procedures (Clarke and Gorley 2006).

Of all possible combinations of the six remaining continuous environmental variables (depth, current speed, visibility, temperature, rugosity and turbidity), depth alone correlated strongest with the observed pattern in fish community structure, albeit weakly ($\rho_S = 0.31$, $p<0.01$, Table 5.9).

Initial analyses for pooled data indicated that temperature was not a component of any parsimonious combination of environmental variables (Table 5.9). Temperature in winter ranged from 19 ˚C–21 ˚C in winter and 25˚C–26 ˚C in summer. Temperature was thus removed as a factor by stratifying the BIO-ENV procedure by season. BIO-ENV analysis stratified by season did not result in any improved relationships indicating that temperature has little influence on fish community structure of fish (Table 5.9).

For pooled and seasonally stratified BIO-ENV results, depth formed a component of all combinations that all displayed weak, but significant, correlations (Table 5.9).

Table 5.9: Results from BEST analysis within the BIO-ENV algorithm, run for the fish assemblages on Two-Mile Reef, yielding the five most parsimonious combinations of abiotic and biotic variables for each *k*, as determined by the Spearmans rank correlation ρ_S.

Stratification	*k*	Best variables	ρ_S	Overall p-value
Season pooled	1	Depth	0.314	p<0.01
	2	Depth, Current Speed	0.307	p<0.01
	3	Depth, Current Speed, Visibility	0.279	p<0.01
	3	Depth, Current Speed, Rugosity	0.273	p<0.01
	2	Depth, Rugosity	0.270	p<0.01
Winter	1	Depth	0.325	p<0.01
	2	Depth, Current Speed	0.307	p<0.01
	3	Depth, Current Speed, Turbidity	0.283	p<0.01
	2	Depth, Turbidity	0.274	p<0.01
	2	Depth, Rugosity	0.252	p<0.01
Summer	1	Depth	0.307	p = 0.02
	2	Depth, Visibility	0.289	p = 0.02
	3	Depth, Visibility, Rugosity	0.281	p = 0.02
	2	Depth, Rugosity	0.279	p = 0.02
	4	Depth, Visibility, Rugosity, Turbidity	0.236	p = 0.02

5.3.6 Dynamics of functional groups

Of the 209 species sampled, coral dependents and EAM territorials had the lowest species abundance and richness with 0.6 and 0.5% total abundance, respectively, and 9 species each (Fig. 5.11). Piscivores occurred at low abundance (4%) despite the presence of a high number of species (46 spp.) (Fig. 5.11). EAM Rovers consisted of 26 species that comprised 15% of the total abundance. The richest functional group was the invertivores (82 spp.) which occurred at relatively low abundance (22%) (Fig. 5.11) but still consistently contributed the highest proportion to between-category dissimilarity (Fig. 5.12). The planktivores occurred at the greatest abundance (58%) and consisted of 37 species (Fig. 5.11).

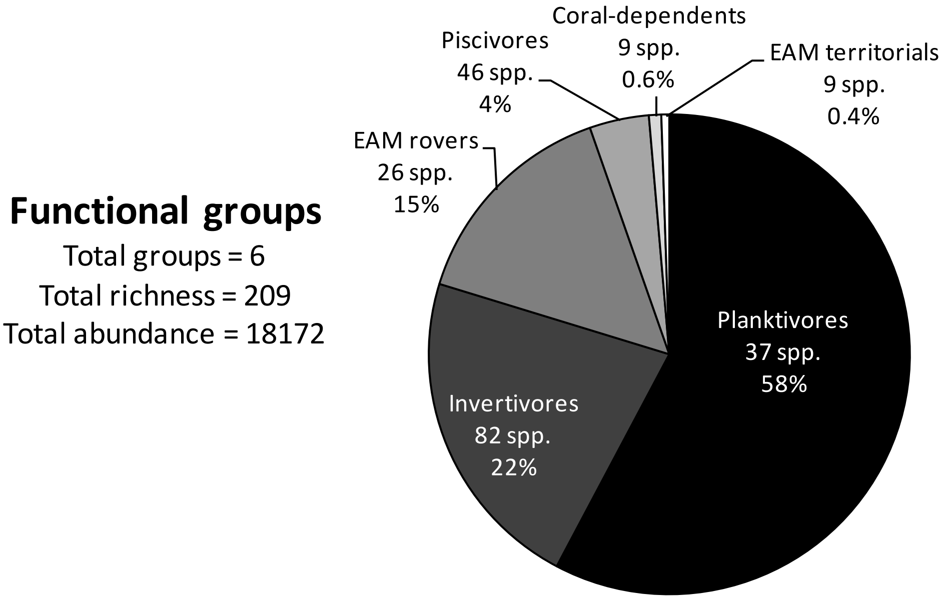
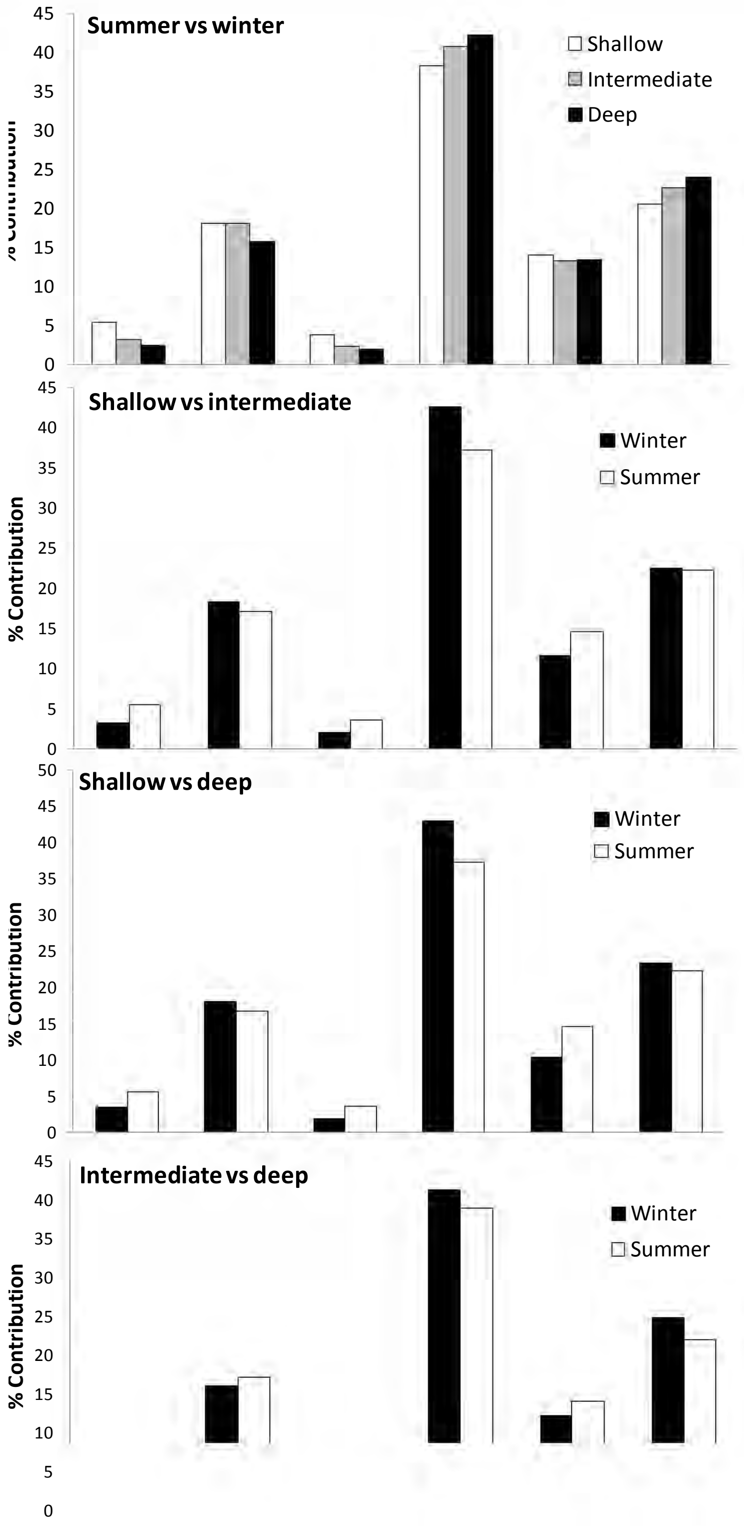


Figure 5.11: Percent contribution of each functional group to overall abundance on Two-Mile Reef.

Seasonal trends of functional groups

Seasonal trends in functional group abundance revealed that over all depths all functional groups are more abundant during summer ($F_{1,84}$ >4.3, $p<0.05$, Table 5.10). This appears to be amplified at shallow depths in that planktivores, EAM rovers, EAM territorials and invertivores ($p<0.05$) were in significantly greater abundances at shallow depths in summer (Fig. 5.13 b-e , Table 5.10).



Summer vs winter

Shallow vs intermediate

Shallow vs deep

Intermediate vs deep

Figure 5.12: Percent contribution of functional fish groups to temporal and spatial differences in fish assemblages on Two-Mile Reef based on values from SIMPER analysis and calculated as the summed contribution of each species to overall functional group contribution.

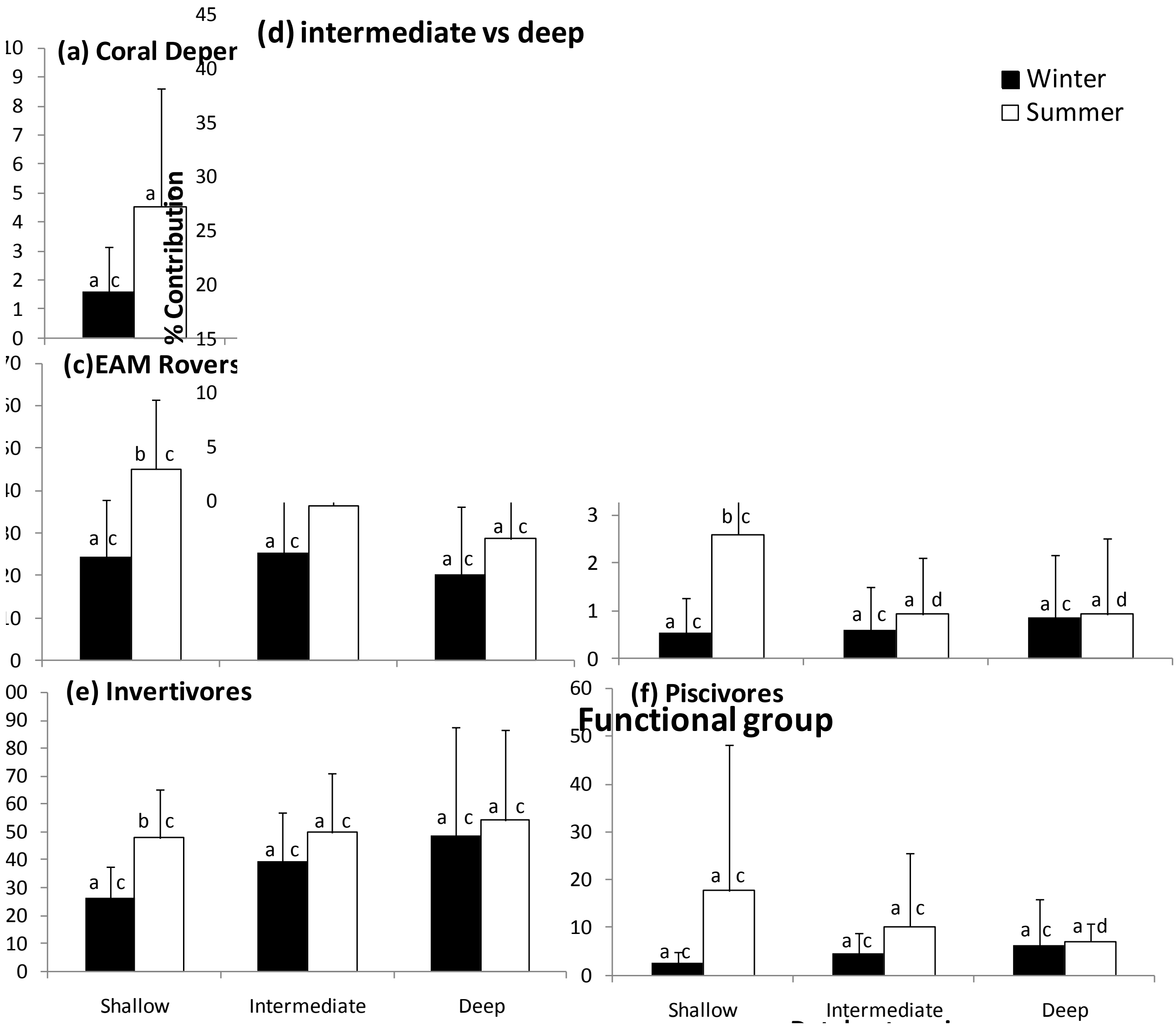


Figure 5.13: Spatial and temporal variation in abundance estimates (average + one standard deviation) of functional fish groups for all three depth categories. Common superscripts depict statistically homogenous groups. Untransformed data are displayed while analyses were conducted on root-root transformed abundance data.

Depth trends of functional groups

Each functional group showed different depth-related patterns (Fig. 5.12 and Fig. 5.13).

Coral-dependents are significantly affected by depth ($F_{2,84} = 6.60$, $p<0.01$, Table 5.10), declining in abundance with depth in summer. Intermediate depths in winter had lower abundances ($p<0.05$) but in summer shallow and intermediate depths had significantly greater ($p<0.05$) abundances than deep depths.

Planktivores and invertivores are not significantly affected by depth (Table 5.10).

EAM rovers, are significantly affected by depth ($F_{2,84} = 5.4$, $p = 0.01$, Table 5.10), exhibiting higher abundances at shallow depths in summer (Tukey‟s HSD test, $p<0.05$).

EAM territorials, are not significantly affected by depth overall (Table 5.10). Shallow depths in summer, however, exhibit significantly greater abundances than intermediate and deep depths (Tukey‟s HSD test, $p<0.05$).

EAM territorials, are not significantly affected by depth overall (Table 5.10). Shallow and intermediate depths in summer, however, exhibit significantly greater abundances than deep depths (Tukey‟s HSD test, $p<0.05$).

Interaction trends (season × depth) of functional groups

The interaction effect of season and depth had a significant effect on the planktivores ($F_{2,84} = 3.51$, $p = 0.04$) and on the piscivores ($F_{2,84} = 3.50$, $p = 0.04$) but did not significantly affect any other group (Table 5.13).

Table 5.10: Statistical results from Two-way Analysis of Variance for each functional fish group. *Post hoc* pairwise comparisons were conducted using Tukey's HSD test. Significant differences (p<0.05) are indicated with bold text.

	Season		Depth		Season × Depth		Pairwise comparisons						
	$F_{1,84}$	p	$F_{2,84}$	p	$F_{2,84}$	p	Depth pooled	Season pooled	Winter (W)	Summer (S)	Shallow (Sh)	Inter-mediate (I)	Deep (D)
Coral dependents	10.82	**<0.01**	6.63	**<0.01**	0.54	0.58	**W<S**	**D<I**,I=Sh,**D<Sh**	**D>I,I<Sh**,D=Sh	**D<I**,I=Sh,**D<Sh**	W=S	**W<S**	W=S
Planktivores	6.51	**0.01**	2.07	0.13	3.51	**0.04**	**W<S**	D=I=Sh	D=I=Sh	D=I=Sh	**W<S**	W=S	W=S
EAM Rovers	6.13	**0.02**	5.41	**<0.01**	0.64	0.58	**W<S**	D=I,I=Sh,**D<Sh**	D=I=Sh	D=I=Sh	**W<S**	W=S	W=S
EAM Territorials	4.32	**0.04**	1.52	0.22	2.01	0.15	**W<S**	D=I=Sh	D=I=Sh	D=I,**I<Sh**,**D<Sh**	**W<S**	W=S	W=S
Invertivores	8.72	**<0.01**	1.61	0.22	1.73	0.33	**W<S**	D=I=Sh	D=I=Sh	D=I=Sh	**W<S**	W=S	W=S
Piscivores	17.71	**<0.01**	0.23	0.84	3.41	**0.04**	**W<S**	D=I=Sh	D=I=Sh	D=I=Sh	W=S	W=S	W=S

5.4 Discussion

This study represents the first holistic, replicated survey of the community structure of South African coral reef fishes between seasons and across depths, habitats, and benthic communities. A total of 18172 individuals from 209 species and 41 families were surveyed on Two-Mile Reef. While both abundance and richness were structured by both season and depth, no structure in diversity was evident. Higher abundance and richness was observed in summer but only at shallow depths. Distinct community structures were noted between season and depth. At shallow depths, winter and summer communities were distinct, and the communities of each depth category were different. Specific drivers of structural changes to community assemblages could not, however, be determined, but it is likely a product of a combination of abiotic (e.g. environmental variables) and biotic (e.g. reef benthic community) factors acting at different seasonal and depth scales. Assessment of the functional species groups present on the reef indicated that more diverse groups contributed more to between-community dissimilarities regardless of abundance. The distribution of abundances within each functional group was not uniform suggesting niche partitioning. This high-latitude system was found to be comparable to other low latitude coral reefs in the WIO.

5.4.1 General fish assemblage characteristics

The observed ichthyofaunal family richness was comparable to previous studies investigating full coral-reef fish community dynamics in the WIO. Previous studies have reported observing between 35 (Chabanet 2002) and 58 (Gillibrand *et al.* 2007) families at various WIO locations. Across all these studies, including this study, labrids dominated species richness followed by the Acanthuridae, Chaetodontidae, Pomacentrida and Serranidae (Letourneur 1996; Chabanet 2002; Garpe and Öhman 2003; Durville *et al.* 2003; Gillibrand *et al.* 2007; Floros 2010b). While scarids may have been cited as one of the species-rich families in other areas (Durville *et al.* 2003; Gillibrand *et al.* 2007), on Two-Mile Reef this family was represented by relatively low species numbers and abundance.

The 209 species observed on Two-Mile Reef was lower than, but comparable to, other WIO (Letourneur 1996; Chabaneet 2002; Garpe and Öhman 2003; Durville *et al.* 2003; Gillibrand *et al.* 2007) and South African (Floros 2010a) studies (Table 5.1). These studies span approximately 20 degrees of latitude and 18 degrees of longitude which would allow for biogeographic influences. Bellwood and Wainwright (2002) and Wilkinson (2008) investigated the influence of biogeography on the community structure of fishes, these studies along with Hobson (1994) all noted that species richness decreases with increasing latitude. This has been attributed to many interrelated factors such as ocean currents and reef structure, but deviations are often observed (Floros 2010a). Although the role of latitude in structuring WIO fish communities has not been adequately addressed, latitude is a possible explanation for the present study observing lower species numbers than other studies. There are three other factors which, along with biogeography, could contribute to observed differences. These are survey site characteristics, survey techniques employed and level of replication.

The survey sites for all previous WIO studies were spread over multiple reefs with a comparatively large spatial distribution often including various reef types. In concordance with island biogeography theory (MacArthur and Wilson 1967; Freeman 2005), the survey of a single reef and its associated habitats will almost certainly produce lower species counts than an assortment reefs, reef types and the associated plethora of habitats. Floros (2010 b) reported a total of 284 species observed over a number of South African and Southern Mozambican coral reefs. Although this is greater than the present study, Floros (2010 b) reported observations of only 189 species on Two-Mile Reef – 20 species less than observed in this study. This difference can be accounted for by survey site characteristics in that Floros (2010) only surveyed a narrow depth range. In contrast to Floros (2010b), this study surveyed the complete depth extent of Two-Mile Reef that included numerous benthic communities (Celliers and Schleyer 2008) and therefore comprised higher species numbers.

UVC studies of full coral-reef fish community dynamics in the WIO have generally used one of two techniques; roving diver or strip transects. Letourneur (1996), Durville *et al.* (2003), Gillibrand *et al.* (2007) and Floros (2010b) all used the roving diver technique - flexible in survey area and not particularly species selective – that probably partly accounts for the higher number of observed species richness. The strip transect protocol implemented by Chabanet (2002), Garpe and Öhman (2003) and this study strongly favours the epibenthic reef fish group and has been shown to select against cryptic and benthic fishes (Chapter 1). Chabanet (2002), using similar techniques to this study in the Comoros, observed six less families but 16 more species. In contrast, the study of Garpe and Öhman (2003) at Mafia Island observed vastly greater numbers of fish taxa - 56 families and 394 species. Although biogeography probably plays a role in these differences, as Mafia Island and Two-Mile reef are separated by almost 20 degrees of latitude, differences in the level of replication could be a factor. Garpe and Öhman (2003), who conducted 110 strip transects - the highest level of replication of all studies in the WIO region, observed greater species numbers at Mafia Island than any other WIO area. In comparison to the present study, the 22% more replicates conducted by Garpe and Öhman (2003) probably contributed to the greater number of observed species.

This study found lower species numbers in comparison to other studies as only a single patch reef was surveyed, the more selective strip transect technique was implemented, and despite replication being adequate, fewer replicates were conducted. Floros (2010b) estimates that the number of reef-associated fishes, including cryptic species, on the South African coral reefs is likely somewhere between the 399 species observed by Chater *et al.* (1993) and 1000. Although the initial estimate of 399–1000 species proposed by Floros (2010b) is probably valid, intensive, destructive survey methods would be required to ascertain an approximation of total richness on high-latitude WIO coral reefs. The similarities in the general characteristics of Two-Mile Reef compared to other WIO areas are also evident in species dominance characteristics.

This study, together with Letourneur (1996), Chabanet (2002), Garpe and Öhman (2003) and Floros (2010b), found that the Pomacentridae and Serranidae (Subfamily Anthiinae) dominated (Table 5.11). Only Letourneur (1996) and Floros (2010b) reported dominance of species unique at the family level. Letourneur (1996),working in Reunion, reported dominance of *Thalassoma purpureum* (Labridae) which is a species that was not observed in the present study but has been identified in South African waters (Smith and Heemstra 2003). Floros (2010b), working on a small patch reef in Mozambique, reported the dominance of *Parapriacanthus ransonneti* (Pempheridae). Although *P. ransonneti* was not observed in transects in this study, photographic records (Wartenberg pers. obs. 2010) confirm that this species currently occurs on Two-Mile Reef, but only at moderate to low abundances. The ommision of *P. ransonneti* from the observations of the present study is probably because strip transects select against this shoaling, but cave-dwelling reef fish (Heemstra and Heemstra 2003). In agreement with the findings of Floros (2010b), the patterns of species dominance on Two-Mile Reef therefore appear similar to other, lower latitude, WIO coral reefs.

Table 5.11: Summary of studies investigating community dynamics in the WIO which have reported family or species dominance as either relative or absolute abundance.

Study	Country/ Island	Reef type	Dominant taxa: Species and/or (Family)
Chater *et al.* (1995)	South Africa	Patch reefs	*Thalassoma herbraicum**
Letourneur (1996)	R'eunion	Outer reef flat	*Chrysiptera unimaculata* (Pomacentridae) *Plectroglyphidodon imparipennis* (Pomacentridae) *Plectroglyphidodon leucozonus* (Pomacentridae) *Thalassoma purpureum* (Labridae)
Chabanet (2002)	Comoros	Barrier reef	*Chromis dimidiata* (Pomacentridae) *Chromis nigrura* (Pomacentridae) *Pseudanthias cooperi* (Anthiinae) *Pseudanthias squamipinnis* (Anthiinae)
Garpe and Öhman (2003)	Mafia Island	Patch, channel and fringing reefs	(Pomacentridae)
Floros (2010b)	South Africa and Mozambique	Patch reefs	*Chromis dimidiata* (Pomacentridae) *Chromis weberi* (Pomacentridae) *Nemanthias carberryi* (Anthiinae) *Parapriacanthus ransonneti*(Pemphiridae) *Pseudanthias squamipinnis* (Anthiinae)
This study	South Africa	Patch reef	*Chromis dimidiata* (Pomacentridae) *Chromis nigrura*(Pomacentridae) *Chromis weberi* (Pomacentridae) (Anthiinae)

5.4.2 Identification of fish communities

Factors that influence the abundance and composition of fish assemblages include, but are not restricted to, season (Götz 2006), depth (Bell 1983; Shpigel and Fishelson 1989; Friedlander and Parrish 1998; Sherman *et al.* 1999; Garpe and Öhman 2003; Durville *et al.* 2003), habitat complexity (Öhman *et al.* 1997; Friedlander and Parrish 1998; Jones and Syms 1998; Öhman and Rajasuriya 1998) and benthic community (Galzin *et al.* 1994, Jennings *et al.* 1996; Garpe and Öhman 2003; Wilson *et al.* 2009). Of these, it is not usually possible to determine a single predictor variable as most variables measured to determine community structure tend to be correlated (Götz 2006). In this study, fish assemblages showed substantial variation among sites categorised by season and depth – depth being a proxy for habitat complexity and benthic community. Significant trends in univariate and multivariate approaches, with a meta-analysis of previously published literature, were used to determine where community differences are most likely to occur.

Seasonal trends were evident in abundance, richness, diversity and multivariate metrics particularly at shallow depths. Although the effect of season on coral reef fish assemblages has been poorly studied, the influence of this factor has been investigated in other aquatic ecosystems. Grossman *et al.* (1982) noted that season had a significant effect in structuring ichthyofaunal assemblages of freshwater stream fishes in Indiana, USA, over a 12 year period. These authors attributed this effect to the reproductive cycles of fishes and variable flooding events. The study was unable to predict seasonal shifts in assemblages between years as the variability and frequency of flooding events aided/ inhibited the reproductive success of different species each year. Patterns in freshwater systems such as streams are of little relevance to the marine environment, but the study of Grossman *et al.* (1982) highlights the variability and interrelatedness of possible predictive factors associated with season. The effect of season on structuring temperate marine fish assemblages has been studied. Aburto-Oropeza and Balart (2001) and Pérez-Espãna *et al.* (1996) both investigated rocky reefs in the Gulf of California. These studies found that fish assemblages differed significantly with season and that species numbers were significantly higher during warmer seasons. In agreement with this Robinson (1973), Alvarez-Borrego and Schwartzlose (1979), Roden and Emilsson (1980), Maluf (1983) and Aburto-Oropeza and Balart (2001) all found that the biggest differences in fish assemblages with season were observed between winter and summer seasons. Although these authors did not determine which factors associated with season were responsible, they concluded that high seasonality observed in a host of abiotic and biotic variables were likely to drive change over similarly structured reefs. This northern hemisphere example is in agreement with a study conducted by Götz (2006) investigating temperate reef fishes in a South African MPA.

Götz (2006) noted that season had a significant effect on the abundance of selected species, diversity (Shannon‟s *H'*, Margalef‟s richness index, and the Taxonomic diversity index), and, overall community structure. Although Götz (2006) described the fish assemblage of that area as a typical, temperate South African reef fish community, the factors driving seasonal changes in fish assemblages could not be determined. A seasonal change in fish behaviour was however speculated as a possible driver. The study of Götz (2006) was conducted in the Goukama MPA in the Western Cape – approximately seven degrees of latitude further south from Two-Mile reef. It is possible that seasonal trends in abiotic and biotic factors, or the drivers of those trends, are similar for Goukama MPA and Two-Mile Reef. This would account for the similar patterns with season observed between these studies despite fish assemblages from different bioregions having been investigated.

Despite noting seasonal changes is it still unclear why there is a detectable difference in community structure between seasons and why, within this difference, one season will consistently produce higher community metrics such as richness and abundance even though highly resident coral reef fishes are unlikely to migrate with season (Götz 2006). Harmelin-Vivien *et al.* (1985) suggest that this is relevant to the seasonal observations of the present study, as well as previous studies, because although fish might not be leaving the reefs in winter, a significant decrease in ichthyofaunal activity, and therefore UVC visibility, can result in the survey of less individuals and species.

Seasonal change has been shown to determine the community structure of fish assemblages (Thomson and Lehner ,1976; Thomson and Gilligan 1983; Castro-Agruirre *et al.* 1995; Aburto-Oropeza and Balart 2001; Götz 2006) while Thomson and Lehner (1976), Thomson and Gilligan (1983), Castro-Agruirre *et al.* (1995) have shown that abiotic and biotic cycles related to season are the main sources of variability in reef fish assemblages. Changes of seasons are associated with annual ecosystem scale cycles in a host of potential predictive factors such as temperature, photoperiod, currents, swell, plankton densities, and fish behaviour (Bertram *et al.* 2001).

No trends in abundance, richness and diversity were evident during winter but all these metrics decreased with increasing depth in summer. Multivariate analyses indicated that the fish assemblages surveyed at each depth in this study show separation between each depth in winter and summer. Friedlander and Parrish (1998) noted that depth seems to be an important habitat variable affecting abundance and distribution of fishes. Although „depth" is often the overriding term used in the literature, increasing depth is most often associated with trends in other factors. This highlights the difficulty in separating out the effects of environmental factors (McCoy and Bell 1991; Parker *et al.* 1994). Several factors cannot be acceptably measured within reasonable logistical constraints and must therefore be qualitatively assessed, such as habitat complexity. As the qualitative descriptions of habitat complexity and benthic community typically change with depth, no attempt was made in this study to separate out the individual effects of these variables by further stratification of analyses. „Depth“ was therefore selected a proxy representing specific depth, habitat complexity and benthic community traits.

Bell (1983), comparing protected and fished rocky reefs in the north-western Mediterranean, investigated the effect of depth on fish assemblages and found that the occurrence and relative abundance of species was greater at shallow (7–10 m) than at deep (15–20 m) sites. Bell (1983) found that although habitat complexity could explain differences between fished and reserve sites within depth categories, differences between depths were likely due to increased algal cover at shallow sites. Anderson *et al.* (2005) surveyed algae up to a depth of 26 m on Two-Mile Reef and found that as algal cover decreases with depth. It is therefore possible that algal cover affects the ichthyofaunal communities on Two-Mile Reef.

Using methods similar to this study, Friedlander and Parrish (1998), investigating Hawaiian coral reefs, assigned depth categories associated with specific reef-structure characteristics to describe depth trends in fish assemblages. In contrast to this study, Friedlander and Parrish (1998) noted, at five depth categories between two and 15 m – a much shallower depth range than the present study (6–30 m), greater abundance and richness at shallow sites. They ascribed lower abundance and richness at shallower sites to severe wave-action and ascribed higher abundance at deep sites to more complex habitat types. Garpe and Öhman (2003) compared two discrete depth categories (3–5 m and 7–10 m) on exposed and sheltered reefs that exhibited unclear benthic zonation patterns and found that although differences in fish assemblages between the two depth categories were clear, these differences could probably be attributed to habitat structure, including substrate compositions, architecture and the presence of live coral cover. The ultimate drivers of changes in community structure were not clear. On Two-Mile Reef as more complex habitats occur at shallower depth –higher abundance and richness was observed. This suggests that habitat complexity is an important factor structuring reef fish communities and is in agreement with Bell (1983), Öhman *et al.* (1997), Friedlander and Parrish (1998), Jones and Syms (1998), and Garpe and Öhman (2003). A lack of clear trends in diversity is similar to Bell (1983) and Friedlander and Parrish (1998). It is anticipated that the inability to detect differences in diversity between categories is a result of the choice of the index used which provides only low-resolution information for highly diverse areas, and the relatively low sample sizes innate in most UVC surveys (Magurran 1998).

Sherman *et al.* (1999) and Kruer and Causey (1992), working in southeast Florida in the USA, attempted to partition those factors influencing fish community structure with depth by placing identical artificial reefs at different depths – 7 m and 21 m (Sherman *et al.* 1999), 14 m and 24 m (Kruer and Causey 1992). Although Sherman *et al.* (1999) observed higher abundance and richness of fishes at deeper sites, Kruer and Causey (1992) observed the opposite, i.e. higher abundance and richness at shallow sites. Sherman *et al.* (1999) concluded that although depth clearly had an effect on fish communities, the factors driving differences might be linked to the flow dynamics of the area, while Kruer and Causey (1992) concluded that the creation of microhabitats resulting from increased scouring at shallow sites accounted for differences. Both these studies emphasise that although differences in fish communities can occur between depth categories, „depth" is most often associated with other unmeasured factors. This is relevant to this study in that as depth increases on Two-Mile Reef, habitat complexity decreases, benthic coral communities shift from dense coral domination to more sparse sponge domination, and benthic algal community biomass and cover decreases.

Habitat complexity, or the degree of structural architecture, may, buffer the effects of physical stress, inhibit foraging predators and interfering competitors, and/or alter the availability of resources and their rate of acquisition (Safriel and Ben-Eliahu 1991; Sherman *et al.* 1999; Aburto-Oropeza and Balart 2001). Furthermore, habitat complexity provides refuges and barriers that fragment a reef, resulting in more heterogeneous fish assemblages (Sebens 1991). Although the correlation between habitat complexity and fish communities is not always consistent (Roberts and Ormond 1987), it is generally accepted that community abundance on different reefs increases with habitat complexity (Gladfelter and Gladfelter 1978; Callum and Ormond 1987; Reñones *et al.*1997). The influence of habitat complexity on fish assemblages is therefore often contributed to by the dominant benthic community (Beukers and Jones 1998).

Benthic coral communities can contribute to the structural architecture of a reef. As small changes in the coverage of live coral may produce significant changes in the abundance, richness and community structure of fish (Bell and Galzin 1984; Carpenter *et al.* 1981

Chabanet *et al.* 1997; Feary *et al.* 2007). Where coral cover is high, structural complexity is also generally high, resulting in abundant shelter for resident reef fishes (Sano *et al.* 1987; Beukers and Jones 1997; Lewis 1997; Lirman 1999; Feary *et al.* 2007), particularly smaller species such as the pomacentrids (Garpe and Öhman 2003). On smaller scales, certain species, such as the hawkfish *Cirrhitichthys oxycepalus*, are closely related with different sizes of coral heads (Aburto-Oropeza and Balart 2001) a behaviour displayed on Two-Mile Reef (pers. obs). Besides adding to the structural complexity of a reef, coral and/or algal communities can influence fish assemblages by attracting grazing corallivores. For example, coral-dominated benthos can attract species such as obligate corallivores such as some chaetodontids, while algal-dominated benthos can attract grazing herbivores such that some acanthurids, which will in turn attract predators (Bell 1983; Pratchet 2005). Currie (2005), who specifically assessed the effect of benthic community in structuring fish assemblages on a number of reef sites in the same area as Two-Mile Reef, noted that the type of benthic community had a significant effect on fish assemblages. Currie (2005) showed that coral-dominated sites were different to sites dominated by foliose algae, sponges or turf algae. This is a possible explanation for why shallow and intermediate depths, comprising of coral benthic communities, show similar community structures but deep depths, comprising of sponge and gorgonian benthic communities, differ.

Based on these findings the fish assemblages on Two-Mile Reef can therefore be described by four relatively distinct communities; a shallow-winter community, a shallow-summer community, a year-round intermediate community and a year-round deep community.

5.4.3 Multivariate dynamics of Two-Mile Reef fish communities

Each of the four communities are characterised by unique species dominance and multivariate dynamics in terms of within-category similarity, between-category dissimilarity, and those species that can be considered discriminating species. Although there is no change in the dominant species from shallow to intermediate depths, which were pomocentrid dominated, deep depths exhibited a dominance of the anthiines. Finding a similar pattern, Aburto-Oropeza and Balart (2001) showed that the deepest, most homogenous habitats harbour fewer species but that these species are highly characteristic of these habitats. Aburto-Oropeza and Balart (2001), in agreement with Larson and DeMartini (1984), Holbrook and Schmitt (1989) and Harmelin (1990), concluded that this pattern was probably a result of certain species being more selective of various habitats, ecological processes such as resource competition, and trophic considerations. Both the anthiines and pomacentrids feed primarily on zooplankton and occupy similar habitats (Smith and Heemstra 2003; Froese and Pauly 2011). Although pomacentrids occur in high abundance at deep depths, the change to the anthiine dominance indicates that factors related to niche partitioning such as reproductive requirements or predation pressure, could be driving change (Shpigel and Fishelson 1989).

This study observed that all species contributed only low proportions to within-category similarity and between-category dissimilarity, and that no species exhibited a disproportionately large contribution to overall similarity and dissimilarity. This trend has been previously observed for WIO (Garpe *et al.* 2006) and South African (Floros 2010b) coral reef fish assemblages and is likely a product of high species, habitat and resource diversity (Clarke and Gorley 2006).

Discriminating species are often used as a guide for determining which species can be considered good indicator species of factors such as anthropogenic pressure or reef health (Bennett 2008; Floros 2010b). Information on selected species may provide insight into changes in community structure as a result of these factors when information on the whole community is not available or difficult to obtain (Pajak 2000). Not all species which are dominant, or which exhibit high proportional contributions to community similarities and dissimilarities, constitute good discriminating species (Clarke and Gorley 2006).
Floros (2010b) identified 26 discriminating species over a number of South African and Southern Mozambican coral reefs for a narrow depth range comprising a single benthic community. Barring *A. leucosternon,* the Anthiinae, and *C. dimidiata*, none of the species identified as discriminating species using stricter criteria appeared there. Discriminating species are partly determined by their contribution to between-category dissimilarity (Clarke and Gorley 2006). Floros (2010b) determined discriminating species for different reefs and protection regimes. In contrast, this study found discriminating species for different seasons and depths. The reason for these differences is probably the different factors that were compared.

Floros (2010c) proposes a list of 25 indicator species based on an initial baseline survey, the trends observed in MDS plots, species appropriateness to diving and fishing usage, subjective assessment of the impact of each species to the ecosystem, and ease of identification. Floros (2010c) proposed that this list of 25 indicator species be incorporated into a long-term monitoring program for the WIO high latitude coral reefs. Suitable indicator species for long-term monitoring are easier, cheaper or more accurate to measure, show an earlier response to an impact, or changes in environmental variables (Vos 2000). Floros" (2010c) list is proposed purely for the monitoring of usage while the discriminating species determined in the present study could indicate changes to environmental variables associated with season and depth. It is therefore proposed that the list of 25 species proposed by Floros (2010c) be expanded to incorporate *A. tennentii*, *C. nigrura*, *C. weberi*, and *C. strigosus*. It must be noted though that choice of which indicator variables or species to measure is not a simple one (Keough and Quinn 1991; Degnbol and Jarre 2004) and often requires relatively uneducated subjectivity. Indicator species should ideally convey as much information as possible with respect to the effects of anthropogenic impacts and environmental change on the health of coral reef communities (Green 1979; Hodgson 1999). A balance needs to be found between the efficiency of surveying less species, and the information that can be obtained from tracking shifts in more species (Pajak 2000; Thompson and Mapstone 2002). A cost-effective, accurate and efficient method for surveying the whole community of an area, such as the videographic transect technique, would therefore be more appropriate than the survey of a selected list of species. This is especially true for areas such as the iSimangaliso Wetland Park for which only sparse baseline information exists.

5.4.5 Relating community-level trends to environmental variables

The potential driving pathways of coral reef ecosystems are complex (Wilson *et al.* 2008). In this study every effort was made to keep variation in environmental factors random. Any relationships between the seven variables depth, profile, rugosity, temperature, visibility, currents speed and turbidity can therefore be considered real. Depth, as an abiotic factor, accounted for the greatest contribution to observed patterns in fish assemblages. Rugosity, temperature, visibility, current speed and turbidity showed only a weak relationship to observed patterns and did not aid in improving BEST results.

Based on the assessment of continuous factors using BEST analysis, it is unclear which continuous factors might also be contributing to observed changes in the ichthyofaunal communities. The high complexity inherent in diverse ecosystems such as coral reefs means that a wide array of factors from climatic to behavioural changes has the potential to influence coral reef community structures in any combination of ways (Wilson *et al.* 2008). Aburto-Oropeza and Balart (2001) and Pérez-Espãna *et al.* (1996) noted significant changes with season but were also unable to determine which factors were drivers of change and conluded that seasonal changes were probably a result of seasonal cycles in a hose of environmental and biotic variables. This study was also unable to determine the direct/indirect influences of environmental variables. Information in this regard would prove useful to the implementation of a long-term monitoring program and to the provision of management. Further research in this regard is therefore recommended.

5.4.6 Dynamics of functional groups

It has been proposed that habitat can have a bottom-up cascade effects on the abundance of higher trophic groups (Garpe *et al.* 2006; Wilson *et al.* 2008). In general, species at lower trophic levels are directly affected by resources such as habitat availability and structural complexity (Garpe *et al.* 2006). Bell (1983) noted that samples from the same depth were similar because the majority of species showed a preference for either deep or shallow areas, and that the biology of several species indicated that feeding requirements dictated depth preferences. This is a logical assumption as the abundance of a specialist"s food source would have considerable influence on its distribution (Öhman *et al.* 1997). This hypothesis has however been disputed, as reef fish species are not all specialists and can vary from small species that may be associated with single coral species to those that may be found ubiquitously(Williams 1991; Munday *et al.* 1997; Munday 2000). To clarify which hypothesis applies to the fish communities of Two-Mile Reef, the contribution of each functional group to between category dissimilarity and overall trends in abundance were assessed.

The abundance of species within the same functional group is often controlled by different drivers, effectively reducing the overall strength of the response at the functional level (Wilson *et al.* 2008). Community level responses to change in habitat are therefore likely to be attenuated when there is high diversity within a functional group (Schmitz *et al.* 2000). This is supported by the finding that despite exhibiting relatively low abundance, the highly diverse invertivore group (82 spp.) accounted for more the greatest (>35%) between category dissimilarity at all levels of stratification.

The presence of important functional groups of reef fishes is important for coral reef health (Harmelin-Vivien 1979). Tracking their changes can provide information on possible reef degradation or recovery (Harmelin-Vivien 1979). All functional groups showed increased abundances during summer, although in most cases this was only significant at shallow depths. Each functional group showed unique patterns with depth.

Declines in fish that either feed or shelter within live coral can be directly attributed to coral loss, a result that has been widely reported in the literature (Wilson *et al.* 2006). Garpe and Öhman (2003) and Findley and Findley (2001) found that the majority of the variation in obligate corallivores was explained by live coral cover. Subtle shifts in the benthic coral community can favour some species over others, influencing the composition of coral-dependent fish communities (Berumen and Pratchett 2006). Decreasing coral cover and a change in the coral community with increasing depth are therefore likely to account for decreasing abundance of coral-dependent species with depth. The abundance of a number of other families and feeding categories were also determined by the amount of live coral cover, a finding observed by Garpe and Öhman (2003) at Mafia Island also in the WIO.

Garpe and Öhman (2003) have suggested that hydrodynamic factors may regulate fish community composition particulary for plankton feeders which rely on exposure and surges to replenish resources. Hobson (1991) observed greater planktivore abundance along more exposed reef edges and in deeper water. This study found that planktivore abundance is significantly affected by the interaction of season and depth, and their associated factors. It is therefore possible that this interaction stems from factors that determine the abundance of plankton in the water column generating increased plankton densities at deeper depths in winter and the converse in summer. To understand this relationship, it is recommended that future work also investigates plankton density.

Complex habitats are essential for the survival of many small-bodied reef fish (Graham *et al.* 2006). On Two-Mile reef, the greatest habitat complexity is observed at shallower depths. Wilson *et al.* (2008) showed that EAM-feeding pomacentrids also show a preference for habitats with skeletons of branching corals (Wilson *et al.* 2008). This may explain why the generally small, resident EAM territorials are observed at significantly higher abundances at shallow sites. The abundance of the generally large bodied EAM rovers showed no significant pattern with depth suggesting the utilisation of more diverse habitats (Graham *et* *al.* 2006). Food availability could also play a role in the distribution of these algal feeders as algal cover and biomass is higher in shallower depths (Anderson *et al.* 2005). The lack of observable patterns in invertebrate-feeding fish indicates that the distribution of this group on Two-Mile reef was not related to depth and its associated habitat characteristics. The even distribution of invertivore abundance may partially relate to the greater mobility of these fish compared with similar-sized species from other trophic groups (Wilson *et al.* 2008). For example, Ceccarelli *et al.* (2005) showed that EAM-feeding pomacentrids have territories that are 1 m^2 while Jones (2005) showed that invertebrate-feeding wrasses of similar size forage in areas between 30 m^2 and 150 m^2 that often include areas of low complexity such as coral rubble. The lack of observable pattern in the abundance of invertivores can therefore be explained in that this group are habitat generalists (Vazquez and Simberloff 2002).

Predators such as piscivores are important species in coral reef ecosystems because of their role in regulating fish abundance (Steele *et al.* 1998; Pala 2007). Although top-level predators are the focus of fishing effort, and therefore receive most of the attention in the literature, it is important to understand the dynamics of all predators (Hixon 1991). This study found that piscivore abundance decreased with depth and was influenced significantly by the interaction of season and depth. Hixon and Beets (1993) state that a pleuralistic approach to explaining the distribution of predator abundances must be taken as abundances are not only related to prey availability but also prey refuges and predator habitat. The interaction of piscivores with season and depth could therefore be a product of a combination of these factors.

The distribution of functional groups on coral reefs is complex (Williams 1991; Munday *et al.* 1997; Munday 2000). Garpe *et al.* (2006) states that with decreasing habitat availability all taxa and functional groups are likely to decrease in total abundance and taxonomic richness. This claim has been generally accepted as fishes are viewed as being versatile and opportunistic in their use of available resources (Pratchett *et al.* 2001; Bellwood *et al.* 2003). More recent research does not, however, support this view as it has been suggested that some functional groups are highly specialised in their use of certain prey and/or habitat types (Munday 2004; Prachett 2005; Gardiner and Jones 2005). For example, the distribution and abundance of more specialised functional groups on Two-Mile Reef, such as the coral-dependants or EAM territorials, appears constrained by the availability of specific resources. More versatile species, such as the EAM rovers, do not mimic this pattern as abundances appear more evenly distributed – a finding also observed by MacNally (1995). Despite previous claims that the distribution of abundances of reef fishes is directly related to habitat availability, the present study indicates that the application of this rule is dependent on how specialised/generalist each functional group is.

5.5 Conclusions

This study is the first to have documented the fish assemblages of a high-latitude WIO coral reef across seasonal and depth-related scales.

The fish assemblages on Two-Mile Reef are a highly diverse community comparable to other, lower latitude WIO areas in terms of family abundance and species richness. Season had a significant effect on structuring fish communities. The effect of season was strongest at shallow depths and decreased in magnitude with increasing depth. Depth also had a significant influence in structuring fish communities and the magnitude of this effect was highest between shallow and deep sites. Each distinct community was dominated by a different species. The shallow-winter community was dominated by *C. nigrura*, shallow-summer community by *C. weberi*, intermediate community by *C. dimidiata*, and the deep community by the subfamily Anthiinae.

The factors season and depth were found to be good predictor variables for changes in community structure but the actual drivers of change could, unfortunately, not be determined. It is likely that changes in community structure with depth are primarily structured by a combination of depth, habitat complexity, and benthic community type, and secondarily shaped by a combination of environmental and behavioural factors.

Of the six functional ichthyofaunal groups investigated, the relatively low abundance but disproportionately high diversity invertivores accounted for the largest proportion of between-community differences. Variation in the abundances of functional groups indicates that the importance of various habitats is unique to each group. It is likely that spatial and temporal areas of high abundance for each group are a product of factors such as habitat complexity, food availability, competition, predation, environmental variables and the complex interaction between these factors.

Chapter 6:

Implications, recommendations and conclusions for the underwater visual census of epibenthic coral reef fishes in the Western Indian Ocean

6.1 Research implications

Marine ecosystems are complex and dynamic natural units which provide goods and services beyond those specific to fisheries (Food and Agriculture Organization (FAO 2003). Coral reefs, for example, are amongst the most productive and biologically diverse ecosystems on earth and supply vast numbers of people with food, recreational possibilities, coastal protection, and aesthetic and cultural benefits (Moberg and Folke 1999). Despite the value of these systems, most coral reefs are characterised by long histories of degradation that is a direct result of anthropogenic influences or an indirect result of climate change (Hughes 1994; Pandolfi *et al.* 2005). Biological information on the fauna and flora of coral reefs is critical if management measures, particularly those necessary for coral-reef conservation, maintenance and restoration, are to be efficiently implemented. One group of organisms which is known to contribute significantly to the structure and function of coral reefs, and one that is of high commercial value, is the fishes (Friedlander *et al.* 1998).

The literature has identified two major concerns regarding the survey of coral-reef fishes. First, it is widely accepted that the most appropriate means of surveying these fishes is by non-destructive underwater visual census (UVC). Over many years a wide array of UVC techniques has developed, which are unfortunately most-often implemented interchangeably. UVCs now lack sufficient coherence, and common design and analysis protocols (Langlois *et al.* 2010). Long-term and between-study comparisons are therefore difficult and often entirely confounded. Standardised protocols are therefore required. Second, while coral reef fishes in some parts of the world have been relatively well-documented, coral reef fish assemblages in some biogeographic regions remain inadequately studied (Garpe and Öhman 2003). To address these two concerns, this thesis adopted a four-phase approach that was logically divided into separate chapters.

The first phase, presented in Chapter 1, reviews the multiple facets of UVCs. The chapter reviewed available methodological and analysis techniques, the available modes with which UVCs can be conducted, the media technology with which UVCs can be conducted, and the shortcomings of UVCs. While this review was not study-specific, the advantages and limitations of the various methods were discussed such that researchers wishing to conduct UVCs in the future can assess the most appropriate methods for their study based on available resources and logistical constraints.

Chapters 2 and 3 constituted the second phase and adopted a study-specific approach to addressing the more pertinent shortcomings of UVCs that may confound studies. It was identified that the biogeographic subregion, the continental south Western Indian Ocean (WIO), was a unique biodiversity hotspot for which the ichthyofauna have only been superficially surveyed. A broad-scale review of the available physical and biological literature pertaining to the high-latitude coral reefs of the south WIO in a South African context was provided. Based on accessibility and reef characteristics, it was decided that Two-Mile Reef on the north-eastern South African coast was an appropriate, representative, high-latitude coral reef on which to conduct the field work for this study. The importance of, and requirement for, implementing standardised approaches to conducting UVCs was emphasised. Building on the physical and biological information, presented in Chapter 2, and based on suitability to long-term monitoring, it was identified that high-resolution information pertaining to epibenthic coral reef fishes on high-latitude coral reefs was of greatest immediate importance. In accordance with this requirement, a standardised protocol for surveying high-latitude epibenthic reef fishes by transecting was proposed. Although strip transects were identified as the most appropriate technique for conducting UVCs, it was envisaged that use of digital photographic and videographic media could further assist refining the strip transect technique such that it is feasible for long-term monitoring.

Phase 3 of this thesis, presented in Chapter 4, refined underwater transecting such that these techniques can be used to provide a high resolution survey of the ichthyofauna of Two-Mile Reef and used in future long-term monitoring programs. Although numerous techniques for the survey of reef fishes have been compared, and further comparisons may seem redundant, technological advancements afford an opportunity to refine previously successful techniques. Comparing transects conducted using digital photographic and videographic technology to traditional slate methods, it was shown that digital technology can be used to improve the accuracy and precision of the data collected and therefore the implementation efficiency of UVCs. It was shown that for the survey of high-latitude epibenthic coral reef fishes, strip transects making use of digital videographic media not only produce better quality data, but are the most efficient to conduct and analyse. Two analysis techniques for analysing digital footage were compared. It was found that the *MaxN* technique was not suitable for analysing footage generated from transects, and that the *Standard-count* technique provided data that was more representative of actual ichthyofaunal community composition. In addition, a simple approach to validating counts from digital footage was proposed. It was found that when sufficiently trained and/or experienced observers are used to conduct counts, validation is perhaps unnecessary. This first attempt at a method of validation could, however, have applications in training inexperienced observers.

The last phase, presented in Chapter 5, aimed to directly address the concern that coral reefs in some biogeographic regions remain poorly studied, resulting in a lack of understanding of their structure and function. This chapter presented high-resolution videographic transect surveys of a large, high-latitude WIO ocean patch reef that was stratified by seasons and depths. General community characteristics indicate that coral-reef fish communities on Two-Mile Reef are not only similar to those of lower-latitude WIO coral reefs in terms of family and species numbers, but are also dominated by similar, and often the same, species. This trend mirrors that of the high-latitude benthic coral communities that represent a biodiversity peak south of the equator (Benayahu and Schleyer 1995; 1998). Although season is generally overlooked as a driver of changes to coral-reef fish assemblages, this study found that on high-latitude reefs season had a significant effect on the ichthyofaunal community structure at shallow depths. Furthermore, within each season, fish community structure was shown to differ significantly with depth, particularly during summer. Meta-analysis of the literature confirmed that depth-related shifts are likely the result of changes in available habitat, the dominant benthic community and algal cover, but specific driving pathways associated with both season and depth could not be determined.

Although the specific drivers of seasonal and depth-related shifts to ichthyofaunal community structure could not be determined, the obvious changes in community structure associated with these factors have significant implications for the design of future UVCs and application of management. It is essential that future ichthyofaunal studies and long-term monitoring programs on high-latitude coral reefs are stratified by these factors to ensure unconfounded results. The consistency of the results achieved in this study, using both univariate and multivariate community analyses, discriminating species, and functional groups, underlines the potential of this experimental design and videographic transects to detect relatively small changes to community structure. It can therefore be concluded that the protocol outlined in Phases 2 and 3, and implemented in Phase 4 of this study, was well-suited to surveying high-latitude epibenthic reef fishes. Because the univariate and community-level metrics of the fish assemblages on Two-Mile Reef are similar to those of lower latitude areas, it is likely that this protocol is relevant to other WIO reef complexes that exhibit similar ichthyofaunal diversity. As the protocol proposed in this study is based on reviewable digital media, and provides a means for rapidly and holistically assessing epibenthic ichthyofaunal community structure, it would probably be appropriate to efficiently compare the health and status of reefs under different usage regimes or to assess longer term changes in community structure between years. This is of immediate applicability to monitoring recovery of degraded WIO coral reefs of Kenya and Tanzania (Watson and Ormond 1994; McClanahan *et al.* 1999) and the implementation of other long-term monitoring programs.

6.2 Management recommendations and conclusions

It is becoming widely accepted that an ecosystem-based approach is the most appropriate means of obtaining biological information for management of marine systems (Shannon *et al.* 2006). Data need to be collected over the necessary spatial and temporal scales. The establishment of long-term monitoring sites in predetermined key areas is therefore critical so that the effects of anthropogenic usage can be quantified and hopefully understood (van Jaarsveld and Biggs 2000). The iSimangaliso Wetland Park is inscribed as a world heritage site primarily because of its unique biodiversity (iSimangaliso Wetland Park Authority 2009). The marine sector of the park contains diverse high-latitude coral reefs that have different protection regimes: sanctuary reefs, SCUBA diving only reefs, and restricted usage reefs. The sanctuary reefs are impacted only by environmental changes, the SCUBA diving reefs are impacted by disproportionately high levels of SCUBA diving, and the restricted usage reefs are impacted by recreational fishing and some SCUBA diving (Schleyer 1999). These characteristics indicate that the coral reefs of the iSimangaliso Wetland Park are not only valuable because of their high biodiversity, but are appropriate study sites for determining the impacts of usage. The SCUBA diving and restricted usage reefs are already known to face direct pressure from a rapidly expanding recreational diving and fishing industry (Schleyer

1999), while all reefs, including the sanctuary reefs, could soon face large-scale bleaching events as the sea surface temperature approaches the local coral bleaching threshold (Schleyer *et al.* 2008). The coral reefs in the park are excellent candidates for quantifying the long-term effects of different usage regimes, but also for the collection of baseline data such that the impacts of potential disturbance events can be quantified.

In the past, UVC survey techniques utilising videographic technology were seen as inappropriate because equipment was not readily available, the techniques were expensive, and trained operators were required (Bennett 2008). Long-term monitoring with UVCs has therefore been developed around survey techniques using traditional slate methods. Based on slate techniques, Floros (2010c) suggests a selected list of species for monitoring reefs of different protection regimes in the park. The primary benefits of monitoring only a selected list of species include reduced bias and variability due to increased simplicity of survey (Vos 2000). Although criteria for the selection of indicator species, such as percent contribution to community-level differences, appropriateness to usage, and ease of identification, are usually outlined, the ultimate decision of which species are to be considered „indicator species" is largely subjective (Kremen 1992). The selection of indicator species for long-term monitoring has therefore been widely criticised, because trends in relatively few subjectively selected species are often extrapolated to whole communities (Kremen 1992; Noss 1990). Indicator species should ideally convey as much information as possible with respect to the effects of anthropogenic impacts, and environmental change, on the health of coral-reef communities (Green 1979; Hodgson 1999). If a long-term monitoring program based on indicator species is to be successful, a balance needs to be found between the efficiency of surveying fewer species, and the information that can be obtained from tracking shifts in more species (Pajak 2000; Thompson and Mapstone 2002). One solution to this trade-off is, instead of reducing the number of species surveyed, to optimise the survey techniques used to survey more species.

In recent years videographic technology has become readily available, is cheaper, and easier to operate. While slate and videographic techniques possess many of the same limitations, the innate standardisation afforded by video removes much of the variability associated with UVCs as survey conditions remain constant through time and do not change with observers and skill-levels (Pelletier *et al.* 2011). Watson *et al.* (1995) states that, in terms of long-term monitoring, bias associated with the descriptions of fish communities is not necessarily problematic if biases remain constant over time.

The primary applications of any ichthyofaunal long-term monitoring program are either to determine if there are long-term shifts in the structure of fish assemblages, to compare protected and unprotected areas, to compare data from a locality previously unprotected but that is now protected, or to compare data from before and after a disturbance event, such as pollution or coral bleaching. All of these applications could be relevant to various aspects of long-term monitoring in the park, require baseline information, and require the implementation of standardised methods to ensure spatial and temporal comparability. It is proposed that the videographic transect technique, as implemented in this study, be applied under the proposed protocol presented in Appendix 4, for the immediate development and implementation as a long-term monitoring program of the coral-reef fishes of the iSimangaliso Wetland Park. While the proposed videographic transect-based long-term monitoring program should continue to monitor the elements investigated in this study, it is recommended that a comparison of reefs under different protection regimes also be included.

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Appendix 1 -
Dive Logistics

Dive planning

All dive planning, outlined in Table A1.1, was conducted in accordance with the diving regulations outlined by the South African Occupational Health and Safety Act (85/1993; Government Gazette 32907). A conservative approach to planning was implemented by taking the maximum sampling depth of 30 m as the planned depth for all dives in the study.

Table A1.1: The General dive plan and estimated dive time for all transects conducted during the study

Activity	Description	Estimated time (hh:mm:ss)
Descent	10 $m.min^{-1}$ to 30 m.	00:03:00
Create buffer zone	Swim 10 m from drop zone. Establish transect start point. Deploy transect line.	00:02:00
Swim transect	50 m distance at 6.25 $m.min^{-1}$.	00:08:00
Gather site parameters	See Chapter 3	00:05:00
Safety stop	Decompression stop at 5 m.	00:03:00
	Total dive time	00:21:00

Diver duties and equipment

Table A1.2: Summary of standardised diver duties and the necessary diver equipment for all dives conducted in this study. Regular SCUBA diving equipment is not included.

Chapter	Duty	Diver one duties	Diver one equipment	Diver two duties	Diver two equipment
All	Standard	- Swim straight line transect along single contour - Maintain transect swimming speed at 6.25 m.min-1 - Record necessary site parameters upon transect completion* - Collect water sample upon transect completion	- Dive computer - Site parameter slate - Water sample jar (80 ml)	- Establish transect start point - Follow diver one - Release transect line as transect length increases - Record depth at each 10 m interval - Notify Diver One when 50 m transect is complete - Record necessary site parameters upon transect completion*	- Transect line with shot attached to tag end - Dive computer - Site parameter slate
4	Visual Transect Duty	- Identify and count all species encountered	- Species identification slate	- None	- None
4	Photo Transect Duty	- Take photograph at 1 m intervals	- Camera	- None	- None
4	Video Transect Duty	- Record transect video	- Camera	- None	- None
5	Video Transect Duty	- Record transect video	- Camera	- None	- None

* See Chapter 3 for explanation

Reference images

Fig. A1.1 and Fig. A1.2 provide visual examples of each of the habitat complexity and substrate categories outlined in Table 3.2 (Chapter 3).

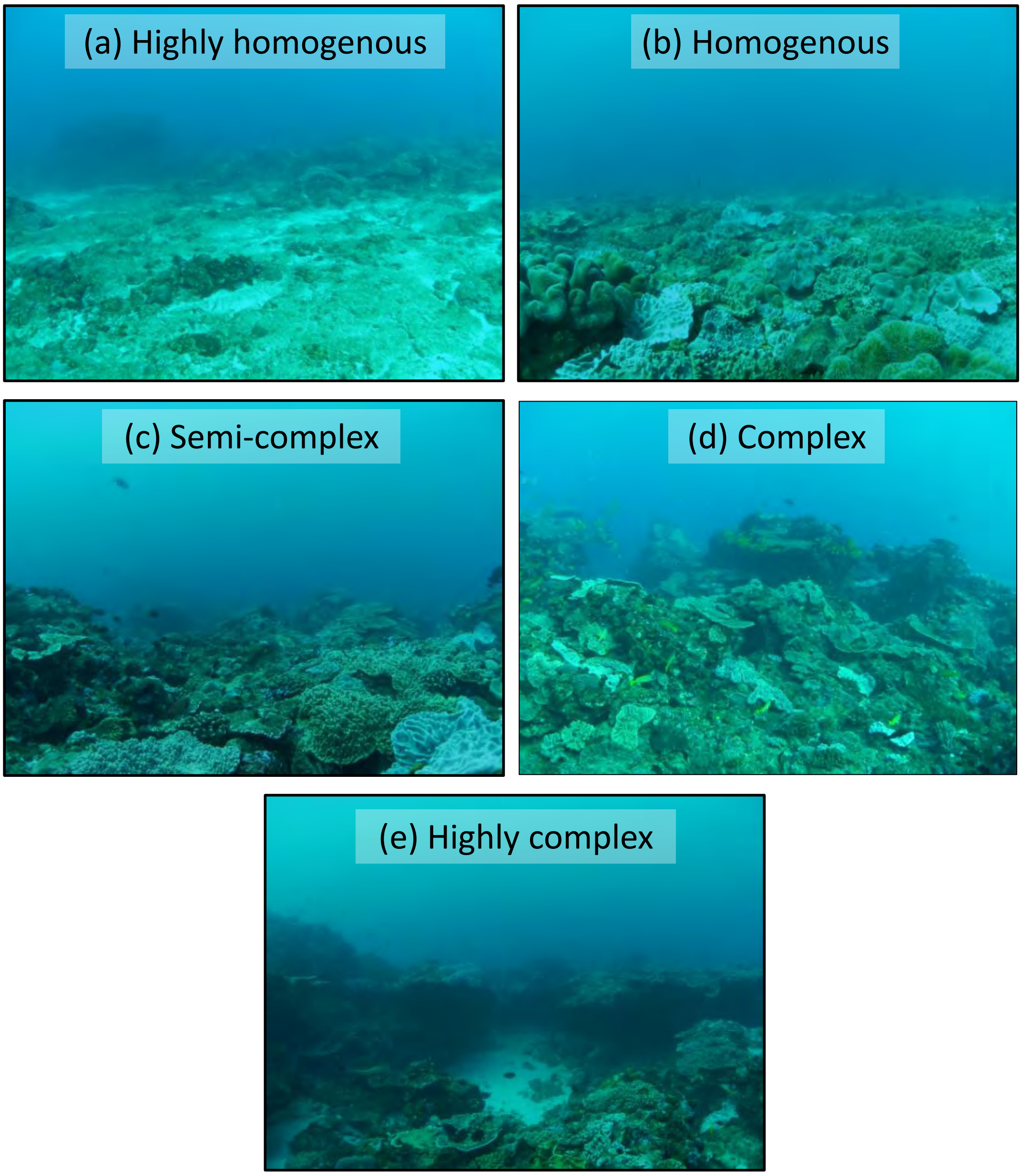

Figure A1.1: Representative examples of the five habitat complexity categories, as outlined in Table 3.2 (Chapter 3), used to quantify the substrate of each sample site. All photographs were taken on Two-Mile Reef.

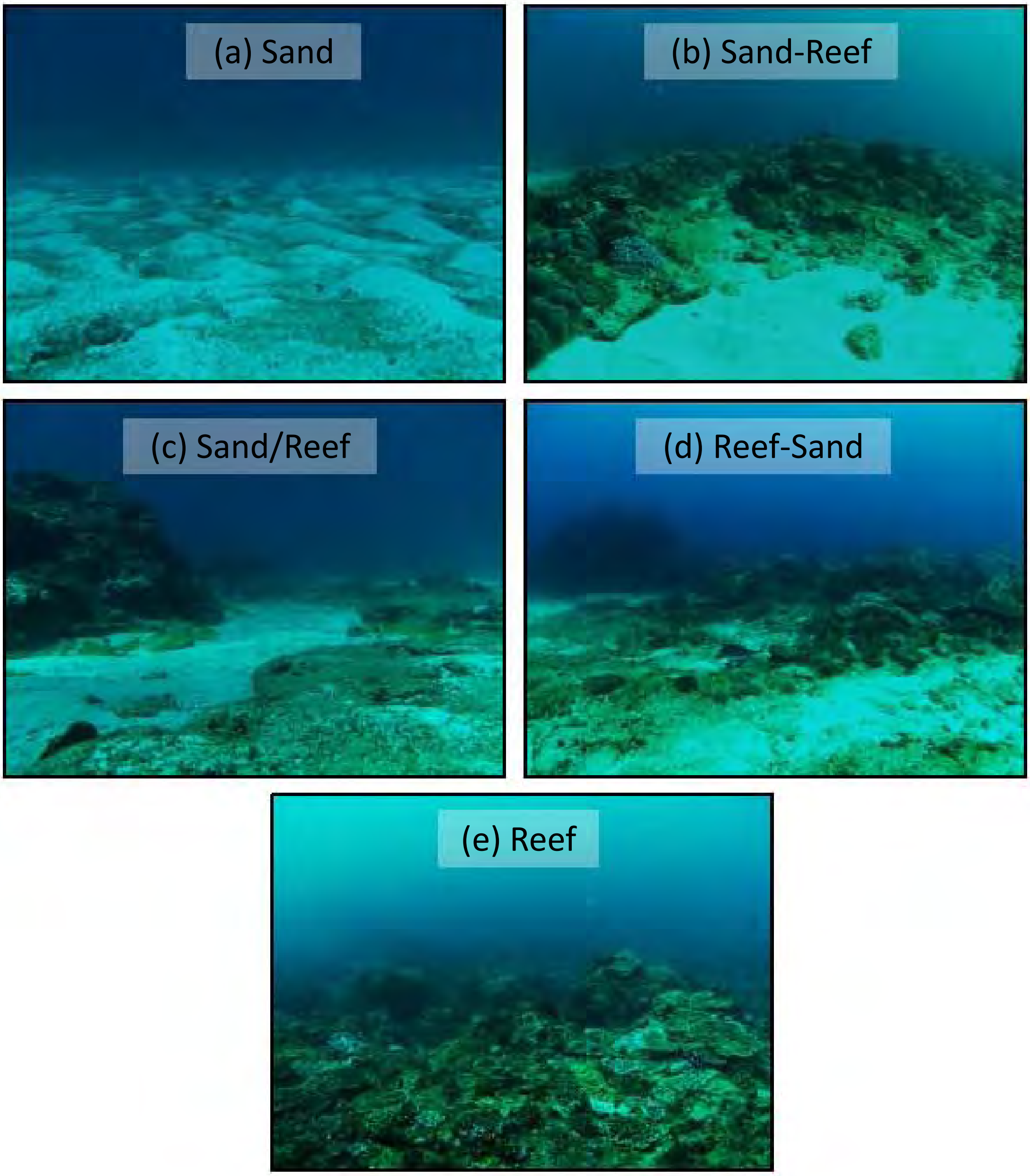

Figure A1.2: Representative examples of the five substrate categories, as outlined in Table 3.2 (Chapter 3), used to quantify the substrate of each sample site. All photographs were taken on Two-Mile Reef.

Appendix 2 -

Reducing observer-related bias and improving study standardisation by observer training

Introduction

Underwater survey research involves large amounts of fieldwork, over great geographic ranges, often over long periods of time. It is therefore inevitable that there will be inter- and intra-observer disparity and shifts in methodology, which can cause significant errors in the data obtained (Thompson and Mapstone 1997). Observer-related error in underwater surveys have been previously noted (Christensen and Winterbottom 1981, Sale and Sharp 1983, Sanderson and Solonsky 1986, Mapstone and Ayling 1998, Samoilys and Carlos 2000, Edgar 2004, Colvocoresses and Acosta 2007) and results in data variability from from observer/ methodological bias more often than random variability (Andrew and Mapstone 1987, Marsh and Sinclair 1989).

Despite a general acceptance that observer-related bias is a reality of UVC surveys, few authors attempt to reduce the problem. Thompson and Mapstone (1997) state that if the correct interpretation of results from UVCs is to be assumed, then observer-related error must, at a minimum, be documented and minimised wherever possible. Although Edgar (2004) states that time spent in the field with experienced observers is generally better spent collecting usable data, comprehensive diver training to minimise observer-related bias is an effective way to significantly improve the quality of data obtained by UVCs (English *et al.* 1994; Harvey *et al.* 2001b). This is particularly relevant to situations where only inexperienced observers are available, or new localities are being investigated. Mapstone and Ayling (1998) and Langlois *et al.* (2010) note that diver calibration will be of greatest importance in studies where temporal rather than spatial comparisons are of interest, such as long-term monitoring programs. Observer-training to ensure within- and between-study standardisation to reduce observer-related bias and to improve spatial and temporal comparability of data were therefore implemented in the present study.

The most common example of diver training is in the estimation of fish length, which aims at improving the quality of biomass data (e.g. Bell *et al.* 1985; Samoilys 1997; Kulbicki 1998;

Harvey *et al.* 2001a; Kadison *et al.* 2002; Edgar 2004; Colvocoresses and Acosta 2007). In this baseline thesis, however, no biomass component was incorporated as it was decided that sampling effort was more appropriately spent on obtaining higher-resolution ichthyofaunal assemblage data. This study, therefore, proposes simple, cost-effective approaches for training observers in terms of fish identification, the estimation of survey technique dimensions, observer swimming speed, and the quantification of some site-specific supplementary variables, all of which are aimed at decreasing data noise.

Materials and methods

Fish identification

Fish identification training was carried out for the single visual count observer (RW). The visual count observer first familiarised himself on the optical properties of fish coloration with increasing depth, and the ability of many fishes to change their coloration to relation to particular behaviours. Fish species images appearing on the video footage obtained during the reconnaissance dives, without the use of artificial light, were compared to photographs of those species taken using a strobe. Strobe-illuminated photographs were supplied by Mr Kobus Els, a recreational SCUBA instructor and photographer (Triton Dive Lodge 2010). All photographs obtained were captured no longer than 12 months prior to the commencement of this work. Familiarisation with the variability in fish coloration would have facilitated more efficient in-water identification during transects, thereby reducing observer task-loading. Once training in terms of the optical properties of fish coloration was complete, training in terms of species specific identification was conducted.

A list of all fishes known to occur, previously observed to occur, or which could possibly occur on the reefs of the study area was compiled from published scientific work and popular literature (Smith 1980; Chater *et al*. 1993; 1995; van der Elst 1993; King 1996; King and Fraser 2002; Smith and Heemstra 2003; Heemstra and Heemstra 2004; Compagno *et al.*2005; Van der Elst and King 2006; Floros 2010a) (Appendix 3, Table A3.1). All species occurring on the list were accompanied by photographs obtained from the same literature, Mr Kobus Els (Triton Dive Lodge 2010), or Froese and Pauly (2010). Wherever possible, images of all forms of each species, e.g. male form, female form, juvenile form, or intra-specific variations in coloration, were included. Where images of all forms of a species were not available, descriptive notes were annotated. The visual count observer was allowed time to study the list of 32 Chondrichthyes from 11 families and 541 Teleosts from 85 families.

The ability of the visual count observer to identify the fishes occurring on the list was then tested. Each test consisted of 50 randomly selected species, either in video or photographic image format, which had to be correctly identified. To ensure proficiency for sufficiently rapid identifications, a maximum time limit of 1 minute 5 seconds was placed on all tests. Only once the visual count observer was able to identify all 50 species correctly on three consecutive, independent tests, was the study commenced. To ensure the observer did not learn specific images, new images and video footage were used in testing wherever possible.

Transect width estimation

As the dimensions of photographic and videographic transects are predetermined by the field of view of the camera lens, only the visual count observer (RW) was required to undertake in-water transect width estimate training. A predetermined list of 30 transect widths, ranging from 2 m to 10 m was marked on a dive slate. Two divers used a fibre-glass tape measure to display the transect widths for the trainee estimate, each diver marking the outer edge of the transect. Randomisation was achieved by using a random number table to select the order of transect widths presented to the trainee. The distance of trainees to the model transect widths was kept variable and random at a distance of 1–6 m, the maximum distance that fishes were likely to be enumerated at. To ensure sufficient precision and accuracy in the estimation of actual transect widths, the visual count observer had to estimate the width of mock transects correct to within 0.1 m. Regression analysis was used to explain the variation between expected and observed transect widths. A coefficient of variation (R^2) >0.95 between expected and observed width estimates for the visual count observer was considered acceptable. Training was repeated until this minimum acceptable R^2 value was obtained. In instances where the minimum acceptable R^2 values were not achieved, data were reviewed such that the observer understood where biases were occurring.

Swimming speed

Unlike estimates of size and distance, the perception of speed will not change in an aquatic medium. Swimming speed training for all divers was therefore first carried out on dry land, and then continued in the water, in a 3-phase training regime. Only five of the divers used for this work were available for swimming speed training. Buddy pairs for this thesis were therefore assigned such that at least one diver in a buddy pair had undertaken swimming speed training. All trainees were trained independently to prevent trainees influencing one another during the learning phase.

During the first phase of training trainees were equipped with a countdown timer set to eight minutes and were required to walk a mock 50 m transect line at a constant speed of 6.25 $m.min^{-1}$, or eight minutes per transect (Chapter 3). Mock transects were conducted such that trainees could regulate speed according to the distance covered and the time elapsed. This phase of swimming speed training was repeated for each trainee until the task could be completed at an approximately constant speed. After this initial phase of land-based training the second land-based training phase commenced.

The second phase of land-based training was identical to the first except that trainees were denied access to a countdown timer. Instead trainees were required to walk the 50 m distance at a constant speed, reaching the finish in 8 minutes ± 24 seconds. This error of 24 seconds, 5% of the total eight minute transect time, can be considered an acceptable margin of error if variable current and surge are taken into account. Once all trainees completed the land based transects within the required time frame, the in-water phase of the training commenced.
The third phase of swimming speed training, in-water training, occurred in an area with sandy substrate to prevent divers having land marks as references of distance. In-water training was carried out in the same manner as the land-based second phase of training. Although divers would be equipped with stop watches and a transect line during actual transect deployments, it was anticipated that their heightened perception of the correct swimming speed gained from more restricted training would improve the ability of divers to conduct necessary activities without needing to constantly verify elapsed time and distance. Each diver was considered to have passed swimming speed training only after 3 consecutive in-water 50 m transects of 480 ± 24 seconds were completed.

Supplementary variables

Of the site parameters measured at each site (Chapter 3), habitat complexity, substrate, and current speed had the potential to possess subjectivity and were hence susceptible to bias. In an attempt to reduce potential bias, training in the estimation of these parameters was conducted in a discussion format. Training in the quantification of supplementary variables was conducted for all observers.

Habitat complexity and substrate categorisation

Training pertaining to the categorisation of habitat complexity and substrate was carried out using video footage obtained from reconnaissance dives.

Footage pertaining to each category of habitat complexity and substrate type (Chapter 3) was gathered during reconnaissance dives (Appendix 3, Fig. A1.1 and A1.2). Trainees were taken through the footage and the characteristics of each category discussed. This would have reduced the need for individual tuition and the subjectivity associated with the allocation of appropriate habitat complexity and substrate categories for each transect deployment.

Current speed estimation

Although estimations of current speed have the potential to be subjective, specific training in terms of current speed estimation is difficult. In an attempt to reduce inter- and intra-diver variability, the characteristics of current speed (Chapter 3) were demonstrated and discussed prior to commencement of the study.
As no formal scoring was assigned to supplementary variable training, training in this regard is not discussed further. Upon successful completion of all facets of diver training, the field work for the study was initiated.

Results and discussion

The inevitable error associated with observers working in an unnatural, aquatic medium, while conducting studies which are spatially and temporally variable, can be minimised by observer training (Thompson and Mapstone 1997, English *et al.* 1994, Harvey *et al.* 2002a, b). Improvements in observer ability were evident in all aspects of observer training conducted in this study; fish identification, transect width estimation and swimming speed regulation.

Fish identification

The single visual count observer for the study required six rounds of testing to achieve three consecutive scores of 100% (Table A2.1) for 50 randomly selected species from the full species list (Appendix 1, Table A1.1).

Table A2.1: Test scores for the single visual count observer. Each test was based on 50 randomly selected species from the full species list (Appendix 1, Table A1.1).

Test no.	Test score (%)
1	94
2	88
3	96
4	100
5	100
6	100

Thompson and Mapstone (1997) state that proficiency in the accurate and rapid identification of the full list of species to be surveyed in an area can be achieved by referencing the numerous photographic texts available. None of the studies which mention implementing this approach, however, provide a means of testing the ability of an observer to rapidly identify and obtain accurate data (e.g. Thompson and Mapstone 1997; Samoilys and Carlos 2000). The current study was therefore the first to conduct formal testing of the ability of an observer to successfully identify fishes. The results of testing conducted here indicate that, despite ample time for the observer to refer to the literature prior to testing, more than one round of testing, and hence re-learning, was required to achieve consecutive scores of 100%. This suggests that relying on observer „experience‟‟ gained by reviewing the appropriate literature is not sufficient to ensure correct identifications. Although the present results are based on only a single observer, it is recommended that fish identification training be implemented as a form of „quality control‟‟ in all UVC studies. Consecutive scores of 100% for the lists or randomly selected images of fishes, within a restricted time-limit, suggest that the single visual count observer was adequately proficient in fish identification in terms of the species likely to be encountered in this thesis.

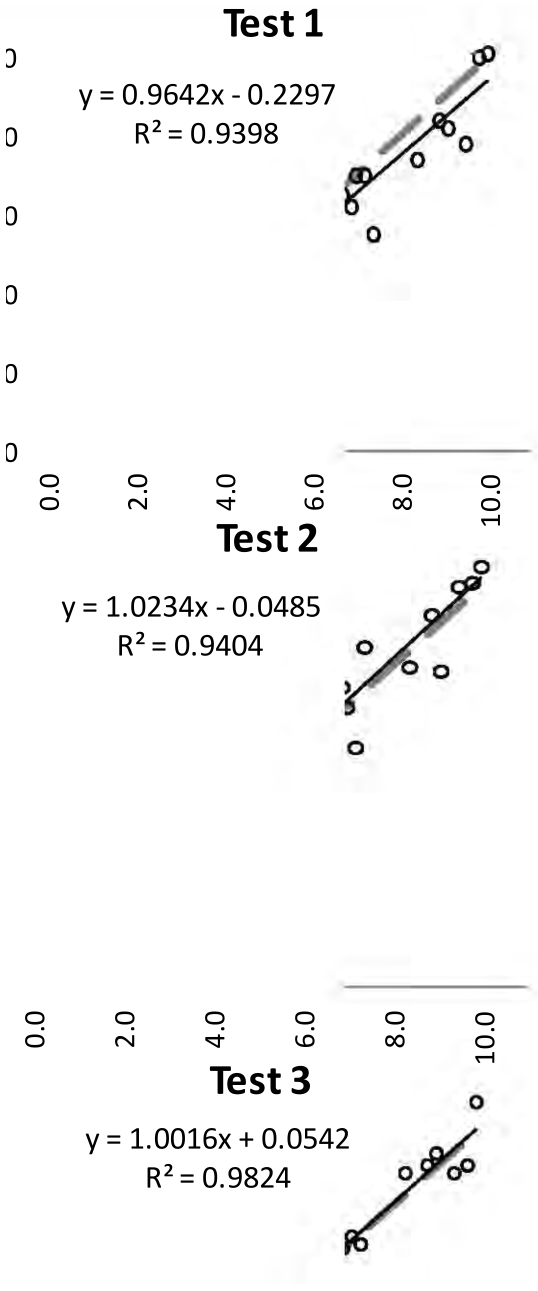

Figure A2.1: Linear regressions of expected vs. observed transect widths for tests 1–3 conducted using the single visual count observer. The solid line depicts the expected vs. observed relationship. Dashed line depicts a 1-1 relationship.

Transect width estimation

Several authors have examined the error associated with the estimation of the shape and dimension of sampling units (Sale and Sharp 1983, Fowler 1987, McCormick and Choat 1987, Mapstone 1988, Buckley and Hueckel 1989, Mapstone and Ayling 1998; Samoilys and Carlos 2000). These authors note that there is potential for observers to incorrectly estimate survey area boundaries which can have significant influences on density estimates. Strip transects, as used in this study, are subject to boundary estimation error because observers have to visually estimate transect boundaries at variable distances during a progressing transect (Samoilys and Carlos 2000). Bohnsack and Bannerot (1986) showed that without practice, an observer could estimate a distance of 7.5 m to within 0.5 m. Nolan and Taylor (1980) found similar results in that divers were able to estimate a 5 m distance to within 0.5 m without training and that with training divers could be accurate to within 0.2 m. Observer training was therefore the method of choice implemented in the present study to reduce the error associated with transect width estimation. The observer undertaking transect width estimate training required three rounds of testing to achieve an $R^2 > 0.95$ (Fig. A2.1) which confirms that, using this observer, training improved the estimates of transect width estimation. Harvey *et al.* (2004), who investigated the ability of observers to estimate distance, found that both inexperienced and experienced observers tended to underestimate distance initially, but that these underestimates were significantly reduced by training. It is therefore anticipated that all transect width estimates conducted for the present study are sufficiently standardised and are within an acceptable level of accuracy to ensure unbiased estimates of density. It is recommended that future studies using visually-based UVC techniques first incorporate survey dimension estimate training to ensure within- and between-study standardisation.

Swimming speed

The speed of an underwater census may bias visual estimates due to the efficiency and presence of the observer (Lincoln Smith 1988; Thresher and Gunn 1986; Fowler 1987) or due to the change in time of the survey (Bortone *et al.* 1988; 1989; Watson *et al.* 1995). The effects of different observer swimming speeds on observed fish density has previously been examined by Lincoln Smith (1988) who found that divers were able to accurately regulate swimming speed without training. However, the high accuracy of the swimming speeds obtained by Lincoln Smith (1988) were achieved by constructing transect lines on the reef substrate prior to conducting transects and then regulating swimming speed by gauging the distance swum along transect against elapsed time, while simultaneously counting and identifying fishes. As this approach is destructive, increases diver disturbance prior to conducting surveys, and adds to the task-loading of divers, the methods of Lincoln Smith (1988) were deemed inappropriate for use in the present study. Other studies which use a predetermined swimming speed to improve study standardisation provide no explanation of how swimming speed was controlled (e.g. DeMartini *et al.* 1989; Samoilys and Carlos 2000). The present study is therefore the first to have implemented some form of swimming speed training in an attempt to standardise a constant swimming speed.

The five observers who undertook swimming speed training required a minimum of four (n = 2) and a maximum of five (n = 3) attempts to achieve 3 consecutive transects within the predetermined 480 ± 24 seconds interval during in-water training (Table A2.2). The trend of improved results with increased training, evident in Table A2.2, indicates that even if training is conducted within the variable survey environment, variation in observer swimming speed can be reduced with repeated practice.

Table A2.2: Transect times for the in-water phase of swimming speed training. Only five observers were available to undertake swimming speed training. (-) depicts no further testing was necessary. The mean ($\bar{x}$) and standard deviation (s) are calculated using the last three transect times only.

	Transect Time (Seconds)						
Observer	1	2	3	4	5	$\bar{x}$	s
1	391	425	471	502	466	479.67	19.50
2	414	422	492	477	488	485.67	7.77
3	421	474	492	500	-	488.67	13.32
4	453	502	483	465	-	483.33	18.50
5	485	451	492	496	498	495.33	3.06

Conclusions

Laborious training of observers can carry substantial cost, which equates to additional time in the field or, more usually, less replication leading to reduced statistical power (Thompson and Mapstone 1997, Edgar 2004). A balance between the benefits of extensive training and the benefits of greater sample sizes must therefore be established (English *et al.* 1994). The training techniques presented here are quick, simple to implement, and effective and, as a result, do not jeopardise resources that would otherwise allow for increased replication. It is therefore recommended that future studies in the iSimangaliso Wetland Park, or the greater Western Indian Ocean, implement similar methods of training in terms of fish identification, transect width estimation, swimming speed and the quantification of supplementary variables such that standardised approaches to conducting UVCs are ensured. If a long-term monitoring program is to be implemented, then continuous retraining of observers every six months, as suggested by Samoilys and Carlos (2000) and Harvey *et al.* (2004), is recommended.

Appendix 3:
Species lists

The full list of fishes known to occur, previously observed to occur, or which could possibly occur in the iSimangaliso Wetland Park (Table A3.1) was compiled from published scientific work and popular literature (Smith 1980; Chater *et al*. 1993; 1995; van der Elst 1993; King 1996; King and Fraser 2002; Smith and Heemstra 2003; Heemstra and Heemstra 2004; Compagno *et al*. 2005; van der Elst and King 2006; Floros 2010a).

Table A3.1: List of 573 species of fishes from 96 families known to occur, previously observed to occur, or which could possibly occur, in the iSimangaliso Wetland Park. SSF = Family number assigned in Heemstra and Heemstra (2004). Families listed by SSF number while species within families are listed alphabetically.

SSF	FAMILY	Species	Common name
7	ORECTOLOBIDAE	*Stegostoma fasciatum*	Leopard/ Zebra shark
8	RHINCODONTIDAE	*Rhincodon typus*	Whale shark
9	CARCHARHINIDAE	*Carcharhinus leucas*	Zambezi/ Bull Shark
		Carcharhinus obscurus	Dusky shark
		Carcharhinus wheeleri	Shortnose blacktail reef shark
		Galeocerdo cuvier	Tiger Shark
		Triaenodon obesus	Whitetip reef/ blunthead shark
13	SPHYRNIDAE	*Sphyrna lewini*	Scalloped hammerhead
		Sphyrna zygaena	Smooth Hammerhead
19	ODONTASPIDIDAE	*Carcharias taurus*	Spotted ragged-tooth shark/ Grey Nurse Shark
22	PRISTIDAE	*Pristis microdon*	Largetooth Sawfish
23	TORPEDINIDAE	*Torpedo sinuspersici*	Marbled electric ray
27	RHINOBATIDAE	*Rhidobatos leucospilus*	Greyspot guitarfish/ sandshark
		Rhina ancylostoma	Bowmouth shark
		Rhinobatos annulatus	Lesser Guitarfish
		Rhynchobatus djiddensis	Giant guitarfish
28	MYLIOBATIDAE	*Aetobatus narinari*	Spotted eagleray
		Myliobatis aquila	Eagleray
		Pteromylaeus bovinus	Bull Ray
30	DASYATIDAE	*Dasyatis chrysonata*	Blue stingray
		Dasyatis kuhlii	Bluespotted stingray
		Gymnura natalensis	Diamond butterfly ray
		Himantura gerrardi	Sharpnose/ brown stingray
		Himantura uarnak	Honeycomb stingray
		Taeniura lymma	Bluespotted ribbontail ray
		Taeniura melansopilos	Round ribbontail/ Giant reef ray

Table A3.1 (Cont.): List of 573 species of fishes, from 96 families, known to occur, previously observed to occur, or which could possibly occur in the iSimangaliso Wetland Park. SSF = Family number assigned in Heemstra and Heemstra (2004). Families listed by SSF number while species within families are listed alphabetically.

SSF	FAMILY	Species	Common name
29	MOBULIDAE	*Manta birostris*	Manta
		Mobula eregoodootenkee	Longhorned mobula
		Mobula japanica	Spinetail mobula
		Mobula kuhlii	Devil Ray
		Mobula tarapacana	Spiny mobula
		Mobula thurstoni	Smoothtail mobula
40	CONGRIDAE	*Heteroconger hassi*	Spotted garden-eel
41	MURAENIDAE	*Echidna nebulosa*	Floral moray
		Gymnomuraena zebra	Zebra moray
		Gymnothorax breedeni	Blackcheek moray/ Masked moray
		Gymnothorax eurostus	Salt and pepper moray
		Gymnothorax favagineus	Honeycomb moray
		Gymnothorax flavimarginatus	Yellow-edged moray
		Gymnothorax javanicus	Giant moray
		Gymnothorax johnsoni	White-spotted moray
		Gymnothorax meleagris	Guineafowl moray
		Gymnothorax nudivomer	Starry moray
		Gymnothorax permistis	Riticulated moray
		Gymnothorax undulatus	Leopard moray
		Rhinomuraena quaesita	Ribbon eel
		Siderea grisea	Geometric moray
42	OPICHTHIDAE	*Myrichthys maculosus*	Ocellated/Spotted snake-eel
58	CHANIDAE	*Chanos chanos*	Milkfish
59	ARIIDAE	*Galeichthys* sp.	Natal seacatfish
60	PLOTOSIDAE	*Plotosus lineatus*	Striped ell-catfish
79	SYNODONTIDAE	*Saurida undosquamis*	Largescale lizardfish
		Synodus dermatogenys	Variagated lizardfish/ Reef lizardfish
		Synodus jaculum	Blacktail lizardfish
97	CARAPIDAE	*Encheliophis borabornesi*	Pineapple pearlfish
102	ANTENNARIIDAE	*Antennarius commerson*	Giant anglerfish
		Antennarius pictus	Painted anglerfish
		Histrio histrio	Sargassumfish
111	ATHERINIDAE	*Atherinomorus lacunosus*	Hardyhead silverside
113	BELONIDAE	Ablennes hians	Barred needlefish
		Strongylura leiura leiura	Banded/Yellowfin needlefish
		Tylosurus crococilus crocodilis	Crocodile needlefish
115	HEMIRAMPHIDAE	*Hemiramphus far*	Spotted halfbeak
		Hyporhamphus affinis	Tropical halfbeak/ Insular halfbeak
116	EXOCOETIDAE	*Exocoetus sp.*	Flying fish
		Exocoetus volitans	Two-wing flying fish
126	BERYCIDAE	*Centroberyx spinosus*	Short alfonsino
128	MONOCENTRIDAE	*Monocentris japonicus*	Pineapplefish
132	HOLOCENTRIDAE	*Myripristis chryseres*	Yellowfin soldier
		Myripristis kuntee	Epaulette/pearly soldierfish
		Myripristis melanostica	Pale Soldier/ Finspot soldier
		Myripristis murdjan	Blotcheye soldier/ Red soldierfish
		Myripristis violacea	Lattice/violet soldier
		Myripristis vittata	Immaculate/ white-tipped soldier
		Myripristris berndti	Bigscale soldier
		Neoniphon argenteus	Silver squirrelfish/ Clearfin squirrelfish
		Neoniphon sammara	Spotfin/ Bloodspot squirrelfish

Table A3.1 (Cont.): List of 573 species of fishes, from 96 families, known to occur, previously observed to occur, or which could possibly occur in the iSimangaliso Wetland Park. SSF = Family number assigned in Heemstra and Heemstra (2004). Families listed by SSF number while species within families are listed alphabetically.

SSF	FAMILY	Species	Common name
132	HOLOCENTRIDAE (Cont.)	*Sargocentron caudimaculatum*	Tailspot squirrelfish
		Sargocentron diadema	Crown squirrelfish
		Sargocentron spiniferum	Sabre/ long-jawed squirrelfish
143	AULOSTOMIDAE	*Aulostomus chinensis*	Trumpetfish
144	FISTULARIIDAE	*Fistularia commersonii*	Smooth flutemouth/ Cornetfish
148	CENTRISCIDAE	*Aeoliscus punctulatus*	Speckled razorfish/ Shrimpfish
145	SYGNATHIDAE	*Corythoichthys* sp.	Red-scribbled pipefish
		Dunckerocampus boylei	Banded pipefish
146	SOLENOSTOMIIDAE	*Solenostomus cyanospterus*	Ghost pipefish
149	SCORPAENIDAE	*Pterois antennata*	Broadbarred firefish/ Spotfin lionfish
		Pterois miles	Devil firefish/ Lionfish
		Pterois rediata	Clearfin firefish/ Radial lionfish
		Rhinopias eschmeyeri	Mauritius scorpionfish
		Rhinopias frondosa	Popeye scorpionfish
		Scorpaenopsis diabolus	False stonefish
		Scorpaenopsis gibbosa	Humpback scorpionfish
		Scorpaenopsis oxycephala	Tassled scorpionfish
		Scorpaenopsis venosa	Raggy scorpionfish
		Synanceia verrucosa	Stonefish
		Taenianotus triacanthus	Paperfish/ Leaf fish
153	CARACANTHIDAE	*Caracanthus madagascariensis*	Spotted croucher
		Caracanthus unipinna	Coral croucher
155	PLATYCEPHALIDAE	*Cociella crocodila*	Crocodile flathead
		Papilloculiceps longiceps	Longhead flathead
		Platycephalus indicus	Bartail flathead
157	TRIGLIDAE	*Chelidonichthys kumu*	Bluefin gurnard
		Chelidonichthys sp.	Bluewing gurnard
159	DACTYLOPTERIDAE	*Dactyloptena orientalis*	Helmut gurnard
163	AMBASSIDAE	*Ambassis natalensis*	Slender glassy
164	KUHLIIDAE	*Kuhlia mugil*	Barred flagtail
166	SERRANIDAE, Subfam. GRAMMISTIDAE	*Aulacocephalus temmincki*	Goldribbon soapfish
		Grammistes sexlineatus	Sixstripe soapfish
166	SERRANIDAE, Subfam. ANTHIINAE	*Nemanthias carberryi*	Threadfin goldie/ Threadfin anthias
		Pseudanthias connelli	Harlequin goldie
		Pseudanthias cooperi	Silverstreak goldie/Red-bar anthias
		Pseudanthias evansi	Yellowtail goldie/ Yellowback anthias
		Pseudanthias fasciatus	Onestripe anthias
		Pseudanthias squamipinnis	Sea goldie/ Goldie/ Lyretail anthias
166	SERRANIDAE, Subfam.EPINEPHALINAE	*Acanthistius sebastoides*	Koester (Check subfamily)
		Aethaloperca rogaa	Redmouth rockcod
		Anyperodon leucogrammicus	Slender rockcod
		Cephalopholis argus	Peacock rockcod
		Cephalopholis miniata	Coral rockcod/ coral hind
		Cephalopholis sonnerati	Tomato rockcod
		Cephalopholis urodeta/ nigripinnis	Duskyfin rockcod
		Epinephalus caeruleopunctatus	Whitespotted rockcod
		Epinephalus chlorostigma	Brownspotted rockcod
		Epinephalus malabaricus	Malabar rockcod
		Epinephalus merra	Honeycomb rockcod
		Epinephalus multinotatus	White blotched rockcod
		Epinephalus suillus	Orangespotted rockcod

Table A3.1 (Cont.): List of 573 species of fishes, from 96 families, known to occur, previously observed to occur, or which could possibly occur in the iSimangaliso Wetland Park. SSF = Family number assigned in Heemstra and Heemstra (2004). Families listed by SSF number while species within families are listed alphabetically.

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166	SERRANIDAE, Subfam. EPINEPHALINAE (Cont.)	*Epinephelus andersoni*	Catface rockcod
		Epinephelus fasciatus	Redbarred rockcod/ Blacktip grouper
		Epinephelus flavocaeruleus	Yellowtail rockcod
		Epinephelus lanceolatus	Brindle bass/ Giant grouper
		Epinephelus longispinis	Streakyspot rockcod
		Epinephelus macrospilos	Bigspot rockcod (Snubnose grouper)
		Epinephelus marginatus	Yellowbelly rockcod
		Epinephelus poecilonotus	Dot-dash rockcod
		Epinephelus polyphekadion	Marbled rockcod
		Epinephelus posteli	Tiger rockcod/ Striped fin rockcod
		Epinephelus rivulatus	Halfmoon rockcod
		Epinephelus tauvina	Greasy rockcod
		Epinephelus tukula	Potato bass
		Plectropomus punctatus	Marbled leopardgrouper
		Variola louti	Swallowtail rockcod
166	SERRANIDAE, Subfam. SERRANINAE	*Serranus cabrilla*	Comber
169	PSEUDOCHROMIDAE	*Pseudochromis dutoiti*	Dutoiti
		Pseudochromis melas	Dark dottyback
		Pseudochromis natalensis	Natal dottyback
173	TERAPONIDAE	*Terapon jarbua*	Thornfish
174	PRIACANTHIDAE	*Priacanthus cruentatus*	Glass bigeye
		Priacanthus hamrur	Crescent-tail bigeye
175	APOGONIDAE	*Apogon angustatus*	Broadstriped cardinalfish
		Apogon apogonides	Short-tooth/ Goldbelly cardinal
		Apogon aureus	Bandtail/ Ringtailed cardinal
		Apogon cookii	Blackbanded cardinal
		Apogon kallopterus	Spinyhead/ Irridescent cardinal
		Apogon taeniophorus	Ninestripe cardinal
		Archamia bleekeri	Golden capped cardinalfish
		Archamia fucata	Redbarred/ Orange-lined cardinalfish
		Archamia mozambicuensis	Mozambique cardinal
		Cheilodipterus artus	Wolf cardinal
		Cheilodipterus lineatus	Tiger cardinal
		Cheilodipterus quinquelineatus	Five-lined/sharptooth cardinalfish
178	POMOTOMIDAE	*Pomatomus saltatrix*	Elf/ Shad
179	HAEMULIDAE	*Plectorhinchus chubbi*	Dusky rubberlips
		Plectorhinchus flavomaculatus	Lemon rubberlip
		Plectorhinchus gaterinus	Blackspotted sweetlips
		Plectorhinchus gibbosus	Harry hotlips/ Gibbus sweetlips
		Plectorhinchus plagiodesmus	Barred rubberlips
		Plectorhinchus playfairi	Whitebarred rubberlip
		Plectorhinchus sordidus	Redlip rubberlips
		Plectorhinchus vittatus	Oriental sweetlips
		Pomadasys commersonni	Spotted grunter
		Pomadasys furcatum	Grey grunter
		Pomadasys kaakan	Javelin grunter
		Pomadasys maculatum	Saddle grunter
		Pomadasys multimaculatum	Cock grunter
		Pomadasys olivaceum	Piggy/ Pinky
		Pomadasys striatus	Striped grunter
180	DINOPERCIDAE	*Dinoperca petersi*	Cavebass

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SSF	FAMILY	Species	Common name
181	LUTJANIDAE	*Aphareus furca*	Blue Smoothtooth/ Smalltooth jobfish
		Aphareus rutilans	Red Smoothtooth/ Smalltooth jobfish
		Aprion virescens	Green Jobfish/ Kaakap
		Etelis coruscans	Ruby snapper
		Lutjanus argentimaculatus	River snapper
		Lutjanus bengalensis	Bluestriped/ Bluebanded snapper
		Lutjanus bohar	Twinspot snapper
		Lutjanus fulviflamma	Dory/ Blackspot snapper
		Lutjanus fulvus	Yellow-striped snapper
		Lutjanus gibbus	Humpback snapper
		Lutjanus kasmira	Bluestriped/ Bluebanded snapper
		Lutjanus lemniscatus	Sweetlip snapper
		Lutjanus lutjanus	Yellow/ Bigeye snapper
		Lutjanus monostigma	Onespot snapper
		Lutjanus notatus	Blue striped snapper
		Lutjanus rivulatus	Speckled snapper
		Lutjanus russelli	Russels snapper
		Lutjanus sanguineus	Blood/Humphead snapper
		Lutjanus sebae	Emperor snapper
		Macolor niger	Black beauty/ Black snapper
		Paracaesio sordidus	False fusilier snapper/ Fusilier snapper
		Paracaesio xanthura	Yellowtail false fusilier/ Protea beam
		Pristipomoides filamentosus	Rosy jobfish
182	CAESIONIDAE	*Caesio caerulaurea*	Blue and gold/ Scissor-tailed fusilier
		Caesio lunaris	Lunar fusilier
		Caesio varilineata	Yellowstriped fusilier
		Caesio xanthonota	Yellowback fusilier
		Pterocaesio sp.	Fusilier
		Pterocaesio tile	Neon/ Bluestreak fusilier
183	SPARIDAE	*Acanthopagrus bifasciatus*	Twobar seabream
		Argyrops filamentosus	Sodierbream
		Argyrops spinifer	King soldierbream
		Cheimerius nufer	Santer/Soldier
		Chrysoblephus anglicus	Englishman
		Chrysoblephus lophus	False englishman
		Chrysoblephus puniceus	Slinger
		Credidens crenidens	Karanteen
		Diplodus hottentotus	Zebra
		Diplodus sargus capensis	Blacktail
		Lithognathus mormyrus	Sand steenbras
		Pachymetopon aeneum	Blue hottentot
		Pengellus bellottii natalensis	Red Tjor-tjor
		Petrus rupestris	Red steenbras
		Polyamblyodon germanum	German
		Polyamblyodon gibbosum	Cristie
		Polysteganus curuleopuntatus	Blueskin
		Polysteganus praeorbitalis	Scotsman
		Polysteganus undulosus	Seventyfour
		Porcostoma denata	Dane
		Rhabdosargus holubi	Cape stumpnose
		Rhabdosargus sarba	Natal stumpnose

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SSF	FAMILY	Species	Common name
183	SPARIDAE (Cont.)	*Rhabdosargus thorpei*	Bigeye stumpnose
		Sarpa salpa	Strepie/ Karanteen
185	LETHRINIDAE	*Gnathodentex aureolineatus*	Glowfish
		Gymnocranius grandoculis	Bigeye emperor/ Rippled barenose
		Gymnocranius griseus	Grey barenose
		Lethrinus borbonicus	Snubnose emperor
		Lethrinus concyliatus	Redaxil emperor
		Lethrinus crocineus	Yellowfin emperor
		Lethrinus harak	Blackspot emperor
		Lethrinus mahsena	Cutthroat emperor
		Lethrinus microdon	Longnose emperor
		Lethrinus nebulosus	Blue/Spangled emperor
		Lethrinus rubrioperculatus	Spotcheek/ Redgill emperor
		Lethrinus variegatus	Variegated emperor
		Monotaxis grandoculis	Bigeye barenose
186	NEMIPTERIDAE	*Scolopsis ghanam*	Silverflash/ Arabian spinecheek
		Scolopsis vosmeri	Paleband spinecheek/ White monocle bream
187	CORACINIDAE/ DICHISTIIDAE	*Coracinus/ Dichistius capensis*	Galjoen/ Damba
		Coracinus/ Dichistius multifasciatus	Banded Galjoen
189	KYPHOSIDAE	*Khyphosus bigibbus*	Grey chub
		Kyphosus cinerascens	Blue chub/ Highfin rudderfish
		Kyphosus vaigiensis	Brassy chub/ Lowfin rudderfish
190	SCORPIDIDAE	*Neoscorpis lithophilus*	Stonebream
192	EPHIPPIDAE	*Platax orbicularis*	Orbicular batfish/ Circular spadefish
		Platax pinnatus	Dusky batfish
192	EPHIPPIDAE (Cont.)	*Platax teira*	Longfin batfish
		Tripterodon orbis	Spadefish
193	MONODACTYLIDAE	*Monodactylus argenteus*	Natal moony
194	GERREIDAE	*Gerres longirostris*	Smallscale pursemouth/ Pouter
		Gerres macracanthus	Longspine pursemouht/ Pouter
195	DREPANIDAE	*Drepane longimana*	Concertina fish
196	MULLIDAE	*Mulloidichthys flavolineatus*	Yellowstripe goatfish
		Mulloidichthys vanicolensis	Yellowfin/ Flame goatfish
		Parupeneus bifasciatus	Twosaddle/ Two-barred goatfish
		Parupeneus cinnabarinus	Redspot goatfish
		Parupeneus cyclostomus	Goldsaddle/yellowsaddle
		Parupeneus indicus	Indian goatfish
		Parupeneus macronemus	Banddot goatfish/ Longbarbel
		Parupeneus pleurostigma	Blackspot/sidespot goatfish
		Parupeneus rubescens	Blacksaddle/ Rosy Goatfish
197	MALACANTHIDAE	*Malacanthus brevirostris*	Stripetail tilefish/ Quakerfish
		Malacanthus latovittatus	Sand tilefish/ Striped blanquillo
198	SILLAGANIDAE	*Sillago sihama*	Silver sillago/Smelt
199	SCIAENIDAE	*Umbrina ronchus/robinsoni*	Slender Baardman/ Tasslefish
204	POMACANTHIDAE	*Apolemichthys kingi*	Tiger angelfish
		Apolemichthys trimaculatus	Threespot angelfish
		Centropyge acanthops	Jumping bean/ African pygmy angelfish
		Centropyge bispinosus	Coral beauty/ Two-spined angelfish
		Centropyge multispinis	Dusky cherub/ Many-spined angelfish
		Genicanthus caudovittatus	Swallowtail/ zebra angelfish
		Pomacanthus chrysurus	Goldtail/African angelfish

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SSF	FAMILY	Species	Common name
204	POMACANTHIDAE	*Pomacanthus imperator*	Emperor angelfish
	(Cont.)	*Pomacanthus maculosus*	Yellowbar/ Arabian angelfish
		Pomacanthus rhomboides	Old woman angelfish
		Pomacanthus semicirculatus	Semicircle angelfish
		Pygoplites diacanthus	Royal angelfish/ Regal angelfish
205	CHAETODONTIDAE	*Chaetodon auriga*	Threadfin butterflyfish
		Chaetodon bennetti	Archer/Bennetts butterflyfish
		Chaetodon blackburnii	Brownburnie/ Chocolate butterflyfish
		Chaetodon dolosus	Black-edged/African butterflyfish
		Chaetodon falcula	Saddled/Saddleback butterflyfish
		Chaetodon guttatissumus	Gorgeous gussy/ Spotted butterflyfish
		Chaetodon kleini	Whitespotted/Klein's butterflyfish
		Chaetodon lineolatus	Lined butterflyfish
		Chaetodon lunula	Raccoon/ Halfmoon butterflyfish
		Chaetodon madagascariensis	Pearly/Chevron butterflyfish
		Chaetodon marleyi	Doublesash butterflyfish
		Chaetodon melannotus	Blackback butterflyfish
		Chaetodon meyeri	Maypole/ Meyer's butterflyfish
		Chaetodon trifascialis	Rightangle/ Chevroned butterflyfish
		Chaetodon trifasciatus	Purple/ Redfin butterflyfish
		Chaetodon unimaculatus	Limespot/Teardrop butterflyfish
		Chaetodon vagabundus	Vagabond butterflyfish
		Chaetodon xanthocephalus	Yellowhead butterflyfish
		Chaetodon zanzibarensis	Zanzibar butterflyfish
		Forcipiger flavissimus	Longnose butterflyfish
		Hemitaurichthys zoster	Brushtooth/black pyramid butterflyfish
		Heniochus acuminatus	Coachman/ Threadback/ Longfin bannerfish
		Heniochus diphreutes	Schooling coachman/Schooling bannerfish
		Heniochus monoceros	Masked coachman/ Masked bannerfish
206	OPLEGNATHIDAE	*Oplegnathus robinsoni*	Natal knifejaw/ Cuckoo bass
210	CARANGIDAE	*Alectis indicus*	Indian mirrorfish/ Threadfin
		Alepes djedaba	Shrimp scad
		Carangoides armatus	Longfin kingfish
		Carangoides caeruleopinnatus	Coastal kingfish
		Carangoides crysophrys	Longnose kingfish
		Carangoides ferdau	Blue kingfish
		Carangoides fulvoguttatus	Yellowspotted kingfish
		Carangoides gymnostethus	Bludger kingfish
		Carangoides malabaricus	Malabar kingfish
		Caranx ignobilis	Giant kingsih
		Caranx melampygus	Bluefin kingfish
		Caranx papuensis	Brassy kingfish
		Caranx sem/ heberi	Blacktip kingfish
		Caranx sexfaciatus	Bigeye kingfish
		Decapturus russelli	Indian scad
		Elangatis bipinnulata	Rainbow runner
		Gnathonodon speciosus	Golden kingfish
		Megalaspis cordyla	Torpedo scad
		Naucrates ductor	Pilotfish
		Pseudocaranx dentex	White kingfish/ Silver trevally
		Scomberoides commersonianus	Largemouth queenfish/ Talang

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210	CARANGIDAE	*Scomberoides tol*	Needlescale queenfish
		Seriola rivoliana	Longfin yellowtail
		Seriolina negrofasciata	Blackbanded kingfish
		Trachinotus africanus	African/Southern pompano
		Trachinotus blochii	Snubnose/Silver pompano
		Trachinotus botla	Largespot popano/ Wave garrick
		Trachurus trachurus	Maasbanker/Jack/Horse Mackerel
211	CORYPHAENIDAE	*Coryphaena hippuris*	Dorado/ Dolphinfish
212	RACHYCENTRIDAE	*Rachycentron canadium*	Cobia/Prodigal son
213	ECHENEIDAE	*Echeneis naucrates*	Shark remora
214	CIRRHITIDAE	*Cirrhitichthys oxycephalus*	Spotted hawkfish
		Cirrhitus pinnulatus	Marbled/Stocky hawkfish
		Cyprinocirrhites polyactis	Swallowtail hawkfish
		Oxycirrhites typus	Longnose Hawkfish
		Paracirrhites arcatus	Arc-eye/ Horseshoe hawkfish
		Paracirrhites forsteri	Freckled/ Pixy hawkfish
215	CHEILODACTYLIDAE	*Cheilodactylus pixi*	Barred fingerfin
		Chirodactylus brachydactylus	Twotone fingerfin
		Chirodactylus fasciatus	Redfingers
		Chirodactylus jessicalenorum	Natal fingerfin
216	PEMPHERIDAE	*Parapriacanthus ransonneti*	Slender sweepers
		Pempheris adusta	Dusky sweepers
219	POMACENTRIDAE	*Abudefduf notatus*	Dusky damsel/ Yellow-tail sergeant
		Abudefduf septemfasciatus	Banded sergeant/Sevenbar damsel
		Abudefduf sordidus	Spot damsel/ Black-spot sergeant
		Abudefduf sparoides	False-eye damsel
		Abudefduf vaigiensis	Sergeant major/ Indo-pacific sergeant
		Abundefduf natalensis	Fourbar damsel/ Natal sergeant
		Amphiprion akallopisos	Nosestripe anemonefish/ Skunk clown
		Amphiprion allardi	Twobar anemonefish/ Clownfish
		Chromis analis	Yellow chromis
		Chromis dasygenys	Bluespotted chromis
		Chromis dimidiata	Chocolate dip/ Twotone chromis
		Chromis lepidolepis	Brown/scaly chromis
		Chromis nigrura	Blacktail chromis
		Chromis opercularis	Doublebar chromis
		Chromis ternatensis	Golden chromis/ Ternate chromis
		Chromis viridis	Blue puller/ Blue-green chromis
		Chromis weberi	Darkbar damsel/ Weber's chromis
		Chrysiptera unimaculata	Onespot damsel/ Onespot chromis
		Dascyllus aruanus	Zebra humbug/ Humbug dascyllus
		Dascyllus carneus	Twobar humbug/ Indian dascyllys
		Dascyllus melanurus	Black-tailed dascyllus
		Dascyllus trimaculatus	Domino/ Threespot dascyllus
		Lepidozygus tapeinosoma	Redwing coral damsel/ Fusilier damsel
		Neopomacentrus cyanomos	Crescent damsel/ Regal demoiselle
		Plectroglyphidodon dickii	Narrowbar damsel
		Plectroglyphidodon johnstonianus	Widebar/ Johnston damsel
		Plectroglyphidodon lacrymatus	Jewel damsel
		Plectroglyphidodon leucozonus	Sash damsel
		Pomacentrus caeruleus	Blue pete/ Caerulean damsel

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219	POMACENTRIDAE	*Pomacentrus pavo*	Saphire/ blue/ azure damsel
		Pomacentrus sulfureus	Sulphur/ lemon damsel
		Pomacentrus trichourus	Yellowtail damsel
220	LABRIDAE	*Anampses caeruleopunctatus*	Bluespotted tamarin
		Anampses meleagrides	Yellowtail tamarin/ Yellowtail/ Spotted wrasse
		Anampses twistii	Yellowbreasted wrasse
		Anapses lineatus	Lined wrasse
		Anchichoerops natalensis	Natal wrasse
		Bodianus axillaris	Axilspot hogfish/ Turncoat
		Bodianus bilunulatus	Saddleback hogfish/ wrasse
		Bodianus bimaculatus	Twospot/Twospot-slender hogfish
		Bodianus diana	Diana's hogfish
		Bodianus leucosticus	Lined hogfish
		Bodianus perditio	Goldsaddle hogfish
		Bodianus trilineatus	Lined hogfish
		Bodinus anthioides	Lyretail hogfish
		Cheilinus/ Oxycheilinus bimaculatus	Twospot wrasse
		Cheilinus chlorourus	Floral wrasse
		Cheilinus/ Oxycheilinus digramma	Cheeklined wrasse
		Cheilinus fasciatus	Red-banded wrasse/ Red breasted splendour wrasse
		Cheilinus trilobatus	Tripletail wrasse
		Cheilinus undulatus	Humphead/ Napolean wrasse
		Chelio inermis	Cigar wrasse
		Cirrhilabrus exquisitus	Exquisite wrasse
		Coris aygula	Clown coris/ Clown Wrasse
		Coris caudimacula	Spottail coris
		Coris formosa/ ferei	Queen coris
		Coris giamard africana/ cuvieri	African coris
		Epibulus insidiator	Slingjaw wrasse
		Gomphosus caeruleus	Birdfish/ Bird wrasse
		Halichoeres cosmetus	Adorned wrasse
		Halichoeres hortulanus	Checkboard wrasse
		Halichoeres iridis	Rainbow wrasse
		Halichoeres nebulosus	Picture wrasse/ Nebulous wrasse
		Halichoeres scalpularis	Zigzag sandwrasse
		Hemigymnus fasciatus	Barred thicklip wrasse
		Hologymnosus annulatus	Barred ringwrasse
		Hologymnosus doliatus	Pastel-ring/ Ringed/ longface wrasse
		Labroides bicolor	Bicolour cleaner wrasse
		Labroides dimidiatus	Bluestreak cleaner wrasse/ Cleaner wrasse
		Labroides xanthonata	V-tail wrasse
		Macropharyngodon bipartitua	Divided/Vermiculate wrasse
		Macropharyngodon cyanoguttatus	Bluespotted wrasse
		Macropharyngodon vivienae	Madagascar wrasse
		Novaculichthys taeniourus	Rockmover wrasse
		Psedodax moluccanus	Chiseltooth wrasse
		Pseudocheilinus evanidus	Striated wrasse
		Pseudocheilinus hexataenia	Sixstripe wrasse
		Pseudojuloides cersinus	Smalltail wrasse
		Stethojulis albovittata	Bluelined wrasse
		Stethojulis interrupta	Cutribbon wrasse

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220	LABRIDAE (Cont.)	*Stethojulis strigiventer*	Threeribbon wrasse
		Thalassoma amblycephalum	Twotone wrasse
		Thalassoma genivittatum	Redcheek/ Blueneck wrasse
		Thalassoma hardwicke	Sixbar wrasse
		Thalassoma hebraicum	Goldbar wrasse
		Thalassoma lunare	Crescent-tail wrasse
		Thalassoma pupureum	Surge wrasse
		Thalassoma trilobatum	Ladder wrasse
		Xyrichthys pavo	Peacock wrasse
		Xyrichthys pentadactylus	Fivefinger wrasse
221	SCARIDAE	*Calotomus carolinus*	Christmas parrotfish/ Star-eye
		Cetoscarus bicolor	Bicolour parrotfish
		Hipposcarus harid	Longnose parrotfish
		Scarus atrilunula	Blue moon/ black crescent parrotfish
		Scarus cyanescens	Blue humphead/ saddled parrotfish
		Scarus frenatus	Bridled parrotfish
		Scarus ghobban	Bluebarred parrotfish
		Scarus psittacus	Palenose parrotfish
		Scarus rubroviolaceus	Ember/redlip parrotfish
		Scarus scaber	Fivesaddle/ Dusky-capped parrotfish
		Scarus sordidus	Bullethead parrotfish
		Scarus tricolor	Tricolour parrothfish
224	SPHYRAENIDAE	*Sphyraena barracuda*	Great barracuda
		Sphyraena flavicauda	Yellowtail barracuda
		Sphyraena jello	Pickhandle barracuda
		Sphyraena putnamiae	Sawtooth barracuda/ Seapike
		Sphyraena qenie	Blackfin barracuda
225	OPISTOGNATHIDAE	*Opistognathus muscatenis*	Robust jawfish
234	PINGUIPEDIDAE/ MUGILOIDIDAE	*Parapercis hexophtalma*	Blacktail/speckled sandsmelt
		Parapercis punctulata	Spotted sandsmelt
		Parapercis robinsoni	Smallscale sandsmelt
		Parapercis schauinslandi	Rosy sandsmelt/ Redspotted sandperch
		Parapercis xanthozona	Blotchlip/ Yellowbar sandsmelt
235	BLENNIIDAE	*Aspidontus dussumieri*	Floating/ Lance blenny
		Aspidontus taeniatus tractus	Mimic blenny
		Cirripectes auritus	Blacktip/ Eared/ Earspot blenny
		Cirripectes stigmaticus	Redstreaked/ Scarlet-spotted blenny
		Ecsenius midas	Golden/ Midas blenny
		Ecsenius nalolo	Nalolo blenny
		Exallias brevis	Leopard rockskipper
		Parablennius pilicornis	Ringneck blenny
		Plagiotremus rhinorhynchos	Twostripe blenny/ Sabretooth blenny
		Plagiotremus tapeinosoma	Piano blenny/ Scale-eating fang blenny
		Scartella emarginatus	Maned blenny
236	TRYPTERYGIIDAE	*Helcogramma obtusirostre*	Hotlips tripplefin
240	GOBIIDAE	*Amblyeleotris steinizi*	Steinitz's shrimp goby
		Amblygobius semicinctus	White-barred reef goby
		Bathygobius coalitus	Whitespotted goby
		Bryaninops yongei	Seawhip goby
		Fusigobius inframaculatus	Inner-spot goby
		Istigobius decoratus	Decorated goby

Table A3.1 (Cont.): List of 573 species of fishes, from 96 families, known to occur, previously observed to occur, or which could possibly occur in the iSimangaliso Wetland Park. SSF = Family number assigned in Heemstra and Heemstra (2004). Families listed by SSF number while species within families are listed alphabetically.

SSF	FAMILY	Species	Common name
240	GOBIIDAE (Cont.)	*Nemateleotris magnifica*	Firegoby/Fire dartfish
		Ptereleostris heteroptera	Blacktail/Spot-tail goby
		Ptereleotris evides	Scissortail/Cocord/Blackfin dartfish
		Ptereleotris heteroptera	Blacktail goby
		Ptereleotris zebra	Zebra goby
		Valencienea sexguttata	Bluespotted sleeper goby/ Six-spot goby
		Valenciennea helsdingenii	Railway glider/Two stripe sleeper goby
		Valenciennea strigata	Pennant glider/ Golden headed jawfish
243	ACANTHURIDAE	*Acanthurus dussumieri*	Pencilled surgeon/ Eyestripe surgeonfish
		Acanthurus leucosteron	Powder-blue surgeonfish
		Acanthurus lineatus	Bluebanded surgeon/ Striped surgeonfish
		Acanthurus mata	Elongate surgeon
		Acanthurus nigrofuscus	Brown surgeon/ Dusky surgeonfish
		Acanthurus tennenti	Lieutenant/ double-band surgeonfish
		Acanthurus thompsoni	Chocolate/ Thompson's surgeon
		Acanthurus triostegus	Convict/banded surgeon
		Acanthurus xanthopterus	Yellowfin surgeon
		Ctenochaetus binotatus	Twospot bristletooth
		Ctenochaetus striatus	Striped/ lined bristletooth
		Ctenochaetus strigosus	Spotted bristletooth/ Goldring bristletooth
		Naso brevirostris	Spotted unicornfish
		Naso hexacanthus	Blacktounge unicornfish/ Sleek unicornfish
		Naso lituratus	Orange-spine unicorn
		Naso tuberosus	Humpnose unicorn
		Naso unicornis	Bluespine unicornfish
		Naso vlamingii	Bignose unicornfish
		Paracanthurus hepatus	Palette surgeonfish/ Blue tang
		Zebrasoma gemmatum	Spotted tang/ Gem surgeonfish
		Zebrasoma scopas	Twotone tang/Brushtail tang
		Zebrasoma veliferum	Sailfin tang
244	ZANCLIDAE	*Zanclus canescens*	Moorish idol
245	SIGANIDAE	*Siganus luridus*	Dusky rabitfish/ Squaretail
		Siganus stellatus	Starspotted/ Stellate rabbitfish
		Siganus sutor	Whitespotted/ Bluespotted rabbitfish
249	SCOMBRIDAE	*Acanthocybium solandri*	Wahoo
		Auxis thazard	Frigate Tuna
		Euthynnis affinis	Eastern little tuna
		Katsuwonus pelamis	Skipjack tuna
		Scomberomorus commerson	King mackeral
		Scomberomorus plurilineatus	Queen mackerl
		Thunnus albacares	Yellowfin Tuna
252	ISTIOPHORIDAE	*Istiophorus platypterus*	Sailfish
		Makaira indica	Black marlin
		Makaira nigricans	Blue marlin
		Tetrapturus audax	Striped marlin
259	BOTHIDAE	*Bothus mancus*	Tropical flounder
263	BALISTIDAE	*Balistapus undulatus*	Orangestriped triggerfish
		Balistoides conspicillum	Clown/ waistcoat triggerfish
		Balistoides viridescens	Titan/ moustache triggerfish
		Melichthys indicus	Indian triggerfish
		Odonus niger	Redfang/blue triggerfish

Table A3.1 (Cont.): List of 573 species of fishes, from 96 families, known to occur, previously observed to occur, or which could possibly occur in the iSimangaliso Wetland Park. SSF = Family number assigned in Heemstra and Heemstra (2004). Families listed by SSF number while species within families are listed alphabetically.

SSF	FAMILY	Species	Common name
263	BALISTIDAE (Cont.)	*Pseudobalistes fuscus*	Rippled/ Yellowspotted triggerfish
		Rhinecanthus aculeatus	Picasso/ Blackbar triggerfish
		Rhinecanthus rectangulus	Rectangular/ Patchy triggerfish
		Sufflamen bursa	Boomerang triggerfish
		Sufflamen chrysopterus	Halfmoon triggerfish
		Sufflamen fraenatus	Bridled triggerfish
		Xanthichthys auromarginatus	Gilded triggerfish
		Xanthichthys lineopunctatus	Striped/lined triggerfish
264	MONACANTHIDAE	*Aluteres monoceros*	Unicorn leatherjacket
		Aluteres scriptus	Scribbled filefish
		Cantherhines dumerillii	Barred filefish/ Yellow eye leatherjacket
		Cantherhines pardalis	Honeycomb filefish/Wire-net filefish
		Oxymoncanthus longirostris	Longnose filefish/ harlequin filefish
		Paraluteres prionurus	Blacksaddle mimic/ mimic filefish
		Pervagor janthinosoma	Redtail / Blackbar filefish
		Stephanolepis auratus	Porky
266	OSTRACIIDAE	*Lactoria cornuta*	Longhorn cowfish
		Lactoria fornasinis	Thornspine/ Backspine cowfish
		Ostracion cubiscus	Boxy
		Ostracion meleagris	Whitespotted boxfish
268	TETRAODONTIDAE	*Amblyrhynchotes honckenii*	Evileye pufferfish
		Arothron hispidus	Whitespotted blaasop
		Arothron immaculatus	Blackedged/ immaculate pufferfish
		Arothron mappa	Map blassop/ Mappa pufferfish
		Arothron meleagris	Guineafowl blaasop
		Arothron nigropunctatus	Blackspotted blaasop
		Arothron stellatus	Star blassop/ Star puffer
		Canthigaster amboinensis	Spotted/ambon toby
		Canthigaster bennetti	Exquisite/ bennett's toby
		Canthigaster janthinoptera	Honeycomb tony
		Canthigaster rivulata	Doublelined toby (Rivulated toby)
		Canthigaster smithae	Bi-coloured toby/ puffer
		Canthigaster solandri	False-eye toby
		Canthigaster valentini	Model toby/ black-saddled toby
269	DIODONTIDAE	*Diodon holocanthus*	Balloon/ Longspined porcupinefish
		Diodon hystrix	Blackspotted/ common porcupinefish
		Diodon liturosus	Shortspine/black blotched porcupinefish

The predetermined species groups displayed in Table A2.3 were assigned to those species which were too morphologically similar to distinguish to species level using slate, photographic and videographic techniques.

Table A3.2: List of 24 species of fishes assigned to 8 species groups due to difficult species-level identifications during visual, photographic, or videographic transects. Species groups were treated as species in subsequent analyses. Species listed in alphabetical order.

Species	Common name	Group Name	Other species in group
Aspidontus taeniatus tractus	Mimic blenny	Bluestreak cleaners	*Labroides dimidiatus*
Bodianus leucosticus	Lined hogfish	Lined hogfishes	*Bodianus trilineatus*
Bodianus trilineatus	Lined hogfish	Lined hogfishes	*Bodianus leucosticus*
Labroides dimidiatus	Bluestreak cleaner wrasse/ Cleaner wrasse	Bluestreak cleaners	*Aspidontus taeniatus tractus*
Lutjanus fulviflamma	Dory/ Blackspot snapper	Dory/Russels Snappers	*Lutjanus russelli*
Lutjanus kasmira	Bluestriped/ Bluebanded/ Bluelined snapper	Bluestriped snappers	*Lutjanus bengalensis* and *L. notatus*
Lutjanus bengalensis	Bluestriped/ Bluebanded/ Bluelined snapper	Bluestriped snappers	*Lutjanus kasmira* and *L. notatus*
Lutjanus notatus	Blue striped snapper	Bluestriped snappers	*Lutjanus kasmira* and *L. bengalensis*
Lutjanus russelli	Russell's snapper	Dory/Russell's Snappers	*Lutjanus fulviflamma* and *L. russelli*
Myripristis kuntee	Epaulette/pearly soldierfish	Blotcheye Soldiers	*Myripristis murdjan*
Myripristis violacea	Lattice/violet soldier	Pale soldiers	*Myripristis melanostica*
Myripristis melanostica	Pale Soldier/ Finspot soldier	Pale soldiers	*Myripristis violacea*
Myripristis murdjan	Blotcheye soldier/ Red soldierfish	Blotcheye Soldiers	*Myripristis kuntee*
Mobula kuhlii	Devil ray	Devil rays	*Mobula kuhlii, M. eregoodootenkee, M. japonica, M. tarapacana, M. thurstoni*
Mobula eregoodootenkee	Longhorned mobula	Devil rays	*Mobula eregoodootenkee, M. japonica, M. tarapacana, M. thurstoni*
Mobula japanica	Spinetail mobula	Devil rays	*Mobula kuhlii, M. eregoodootenkee, M. tarapacana, M. thurstoni*
Mobula tarapacana	Spiny mobula	Devil rays	*Mobula kuhlii, M. eregoodootenkee, M. japonica, M. thurstoni*
Mobula thurstoni	Smoothtail mobula	Devil rays	*Mobula kuhlii, M. eregoodootenkee, M. japonica, M. tarapacana*
Nemanthias carberryi	Threadfin goldie/ Threadfin anthias	Sea Goldies	*Pseudanthias connelli, P. cooperi, P. evansi, P. fasciatus,* and *P. squamipinnis*
Pseudanthias connelli	Harlequin goldie	Sea Goldies	*Nemanthias carberryi, Pseudanthias cooperi, P. evansi, P. fasciatus,* and *P. squamipinnis*
Pseudanthias cooperi	Silverstreak goldie/Red-bar anthias	Sea Goldies	*Nemanthias carberryi, Pseudanthias connelli, P. evansi, P. fasciatus,* and *P. squamipinnis*
Pseudanthias evansi	Yellowtail goldie/ Yellowback anthias	Sea Goldies	*Nemanthias carberryi, Pseudanthias connelli, P. cooperi, P. fasciatus,* and *P. squamipinnis*
Pseudanthias fasciatus	Onestripe anthias	Sea Goldies	*Nemanthias carberryi, Pseudanthias connelli, P. cooperi, P. evansi,* and *P. squamipinnis*
Pseudanthias squamipinni	Sea goldie/ Goldie/ Lyretail anthias	Sea Goldies	*Nemanthias carberryi, Pseudanthias connelli, P. cooperi, P. evansi,* and *P. fasciatus*

Table A3.3: Area standardised abundance (individuals/100 m ^{2}) of species observed using (Slate), photographic (Photo) and videographic (Video) transects on Two-Mile Reef. Families and species within families are listed alphabetically. X = Species was not observed.

FAMILY	Species	Slate	Photo	Video
ACANTHURIDAE	*Acanthurus dussumierii*	0.07	0.03	X
	Acanthurus leucosteron	0.64	1.31	0.78
	Acanthurus lineatus	X	0.06	0.11
	Acanthurus mata	0.02	X	X
	Acanthurus nigrofuscus	1.16	0.28	1.28
	Acanthurus tennenti	0.67	0.71	1.36
	Acanthurus thompsoni	0.40	0.57	0.97
	Acanthurus xanthopterus	X	X	0.08
	Ctenochaetus binotatus	3.18	3.70	8.99
	Ctenochaetus striatus	0.04	0.03	0.19
	Ctenochaetus strigosus	4.51	0.34	1.25
	Naso brevirostris	0.07	X	X
	Naso hexacanthus	X	0.05	0.06
	Naso lituratus	0.18	0.43	0.50
	Naso unicornis	0.09	0.02	0.06
	Paracanthurus hepatus	X	0.02	X
	Zebrasoma scopas	0.40	0.12	0.11
APOGONIDAE	*Apogon kallopterus*	0.04	X	0.03
AULOSTOMIDAE	*Aulostomus chinensis*	X	0.02	X
BALISTIDAE	*Balistapus undulatus*	0.31	0.26	0.50
	Balistoides conspicillum	0.02	0.03	X
	Balistoides viridescens	X	0.02	X
	Melichthys indicus	0.22	0.31	0.39
	Odonus niger	2.67	1.49	1.00
	Sufflamen bursa	0.16	0.12	0.19
	Sufflamen chrysopterus	0.20	0.55	1.28
	Sufflamen fraenatus	0.13	0.08	0.11
BLENNIIDAE	*Aspidontus dussumieri*	X	0.62	0.39
	Ecsenius midas	X	0.02	X
	Exallias brevis	0.02	X	0.03
	Plagiotremus rhinorhynchos	0.02	0.02	0.06
	Plagiotremus tapeinosoma	0.02	0.09	0.03
CAESIONIDAE	*Caesio lunaris*	X	0.86	X
	Caesio xanthonota	1.78	2.11	1.67
	Paracaesio xanthura	X	X	0.14
	Pterocaesio tile	X	1.25	0.58
CARANGIDAE	*Carangoides fulvoguttatus*	0.33	X	X
	Caranx melampygus	0.04	0.11	0.06
CHAETODONTIDAE	*Chaetodon auriga*	0.02	0.08	0.28
	Chaetodon blackburnii	0.11	0.02	0.06
	Chaetodon guttatissumus	0.62	0.20	0.50
	Chaetodon kleini	1.22	0.69	1.34
	Chaetodon lunula	0.07	0.05	0.22
	Chaetodon madagascariensis	0.62	0.18	0.84
	Chaetodon meyeri	0.36	0.14	0.36
	Chaetodon unimaculatus	1.47	0.75	1.45
	Chaetodon vagabundus	0.04	X	0.28
	Forcipiger flavissimus	0.44	0.32	0.64
	Hemitaurichthys zoster	0.27	0.54	0.08
	Heniochus diphreutes	0.02	X	0.03
	Heniochus monoceros	0.02	0.03	X

Table A3.3 (Cont.) Area standardised abundance (individuals/100 m^2) of species observed using (Slate), photographic (Photo) and videographic (Video) transects on Two-Mile Reef. Families and species within families are listed alphabetically. X = Species was not observed.

FAMILY	Species	Slate	Photo	Video
CIRRHITIDAE	*Cirrhitichthys oxycephalus*	0.22	0.15	0.33
	Paracirrhites arcatus	0.27	0.23	0.42
	Paracirrhites forsteri	0.27	0.11	0.47
DINOPERCIDAE	*Dinoperca petersi*	0.07	X	X
DIODONTIDAE	*Diodon liturosus*	0.02	X	X
GOBIIDAE	*Nemateleotris magnifica*	0.09	0.02	0.06
	Ptereleotris evides	0.09	0.09	0.06
	Ptereleotris heteroptera	0.04	X	0.03
	Valenciennea strigata	0.16	0.06	0.11
HAEMULIDAE	*Plectorhinchus flavomaculatus*	0.02	0.02	0.08
	Plectorhinchus playfairi	X	0.14	X
HOLOCENTRIDAE	*Myripristis berndti*	0.13	X	0.03
	Myripristis kuntee	X	0.20	0.45
	Myripristis melanostica	0.09	0.05	0.03
	Myripristis murdjan	0.13	0.12	0.03
	Myripristis vittata	0.02	X	X
	Sargocentron caudimaculatum	X	0.05	0.06
	Sargocentron diadema	0.07	X	X
	Sargocentron spiniferum	0.11	X	X
KYPHOSIDAE	*Kyphosus cinerascens*	X	0.02	0.06
LABRIDAE	*Anampses caeruleopunctatus*	0.04	0.03	0.25
	Anampses lineatus	0.31	0.40	1.06
	Anampses meleagrides	0.11	0.14	0.33
	Anampses twistii	0.02	X	0.06
	Bodianus axillaris	0.02	0.02	X
	Bodianus bilunulatus	0.07	X	0.03
	Bodianus diana	0.20	0.06	0.19
	Bodianus perditio	0.09	X	X
	Cheilinus fasciatus	0.02	X	0.03
	Cheilinus trilobatus	0.02	0.02	X
	Cirrhilabrus exquisitus	X	X	0.03
	Coris aygula	X	0.06	0.06
	Coris caudimacula	0.09	0.35	1.06
	Coris formosa/ ferei	X	X	0.03
	Coris giamard africana/ cuvieri	X	0.02	0.11
	Gomphosus caeruleus	0.36	0.28	1.34
	Halichoeres hortulanus	0.31	0.28	0.42
	Halichoeres nebulosus	0.02	0.02	0.03
	Hemigymnus fasciatus	0.04	0.03	0.11
	Hologymnosus annulatus	X	X	0.03
	Hologymnosus doliatus	0.13	X	0.03
	Labroides bicolor	0.02	X	0.06
	Labroides dimidiatus	0.76	0.54	0.78
	Macropharyngodon bipartitus	0.11	0.02	0.11
	Macropharyngodon cyanoguttatus	0.02	X	0.17
	Pseudocheilinus evanidus	0.07	X	X
	Pseudocheilinus hexataenia	0.04	X	0.03
	Pseudodax moluccanus	0.04	0.02	0.03
	Pseudojuloides cersinus	X	X	0.03
	Stethojulis interrupta	0.04	X	0.03
	Stethojulis strigiventer	X	X	0.14

Table A3.3 (Cont.): Area standardised abundance (individuals/100 m ^{2}) of species observed using (Slate), photographic (Photo) and videographic (Video) transects on Two-Mile Reef. Families and species within families are listed alphabetically. X = Species was not observed.

FAMILY	Species	Slate	Photo	Video
LABRIDAE (Cont.)	*Thalassoma amblycephalum*	0.24	0.14	0.31
	Thalassoma genivittatum	0.02	X	0.03
	Thalassoma hardwicke	0.33	X	0.03
	Thalassoma hebraicum	1.00	0.60	2.73
	Thalassoma lunare	0.04	0.09	0.06
LETHRINIDAE	*Gymnocranius griseus*	0.02	0.03	0.08
	Lethrinus crocineus	X	0.03	X
	Lethrinus rubrioperculatus	X	X	0.06
	Monotaxis grandoculis	X	X	X
LUTJANIDAE	*Aphareus furca*	0.02	0.02	0.06
	Aprion virescens	X	0.02	X
	Lutjanus fulviflamma	0.89	1.11	0.72
	Lutjanus gibbus	0.27	0.06	0.81
	Lutjanus kasmira	0.58	0.42	0.25
	Lutjanus lemniscatus	X	X	0.03
MALACANTHIDAE	*Malacanthus brevirostris*	0.29	X	0.03
MONACANTHIDAE	*Cantherhines dumerillii*	0.04	0.11	0.11
	Cantherhines pardalis	0.07	0.03	0.06
	Paraluteres prionurus	0.04	X	X
MUGILOIDIDAE	*Parapercis punctulata*	0.04	X	X
	Parapercis schauinslandi	X	0.02	X
	Parapercis xanthozona	X	X	0.03
MULLIDAE	*Mulloidichthys vanicolensis*	X	4.33	0.06
	Parupeneus bifasciatus	X	X	0.06
	Parupeneus cyclostomus	0.16	0.06	0.22
	Parupeneus indicus	0.07	X	X
	Parupeneus macronema	0.60	0.26	1.25
	Parupeneus pleurostigma	0.04	X	0.08
	Parupeneus rubescens	0.29	X	0.47
MURAENIDAE	*Gymnothorax eurostus*	0.02	0.02	X
	Gymnothorax favagineus	0.02	X	X
	Siderea grisea	X	X	0.03
OPLEGNATHIDAE	*Oplegnathus robinsoni*	0.16	0.03	0.06
OSTRACIDAE	*Ostracion cubicus*	0.02	0.03	0.06
	Ostracion meleagris	0.04	X	X
POMACANTHIDAE	*Abudefduf natalensis*	0.44	1.23	1.50
	Abudefduf vaigiensis	X	X	0.11
	Amphiprion akallopisos	0.04	X	X
	Amphiprion allardi	0.11	0.12	0.17
	Apolemichthys trimaculatus	0.02	0.06	0.06
	Centropyge acanthops	0.36	0.02	0.17
	Centropyge bispinosus	0.02	X	X
	Centropyge multispinis	1.49	0.40	2.45
	Chromis dimidiata	23.02	16.74	28.20
	Chromis lepidolepis	0.02	X	X
	Chromis nigrura	3.98	2.86	7.32
	Chromis opercularis	8.18	0.69	1.61
	Chromis weberi	2.20	9.82	22.41
	Dascyllus trimaculatus	X	0.02	0.11
	Neopomacentrus cyanomos	1.00	X	X
	Plectroglyphidodon dickii	1.31	0.20	0.92

Table A3.3 (Cont.): Area standardised abundance (individuals/100 m ^{2}) of species observed using (Slate), photographic (Photo) and videographic (Video) transects on Two-Mile Reef. Families and species within families are listed alphabetically. X = Species was not observed.

FAMILY	Species	Slate	Photo	Video
POMACANTHIDAE (Cont.)	*Plectroglyphidodon johnstonianus*	0.62	0.06	0.14
	Pomacanthus imperator	0.04	0.09	0.08
	Pomacanthus rhomboides	0.16	X	0.08
	Pomacentrus caeruleus	0.49	0.22	0.72
	Pomacentrus trichourus	0.13	0.02	0.08
	Pseudochromis dutoiti	0.11	X	X
PRIACANTHIDAE	*Priacanthus hamrur*	0.36	X	X
SCARIDAE	*Calotomus carolinus*	0.02	0.02	0.06
	Scarus atrilunula	X	0.02	X
	Scarus cyanescens	X	0.02	X
	Scarus frenatus	X	0.05	0.06
	Scarus ghobban	X	X	0.08
	Scarus rubroviolaceus	0.93	0.49	0.95
	Scarus tricolor	X	0.02	X
SERRANIDAE	*Aethaloperca rogaa*	0.07	X	X
	Anthiinae	7.04	5.88	7.71
	Cephalopholis argus	X	X	0.06
	Cephalopholis miniata	0.04	0.02	0.06
	Cephalopholis sonnerati	X	0.03	X
	Cephalopholis urodeta/ nigripinnis	X	X	0.03
	Epinephelus rivulatus	0.04	X	0.06
	Epinephelus tukula	0.02	X	X
	Variola louti	0.04	0.06	0.08
SIGANIDAE	*Siganus sutor*	X	0.09	0.19
SPARIDAE	*Chrysoblephus puniceus*	0.02	X	X
	Diplodus cervinus hottentus	X	0.02	0.03
TETRAODONTIDAE	*Arothron mappa*	0.04	X	X
	Arothron nigropunctatus	0.02	X	0.03
	Arothron stellatus	0.04	X	X
	Canthigaster valentini	0.02	X	X
ZANCLIDAE	*Zanclus canescens*	0.33	0.42	0.50

Table A3.4: Area standardised abundance (individuals/100 m^2) of species observed at shallow (6–14 m), intermediate (14–22 m) and deep (22–30 m) depths during winter and summer using videographic transects on Two-Mile Reef. Families and species within families are listed alphabetically. X = Species was not observed.

		Winter			Summer		
FAMILY	Species	Shallow	Inter-mediate	Deep	Shallow	Inter-mediate	Deep
ACANTHURIDAE	*Ctenochaetus binotatus*	4.62	4.62	4.23	7.54	7.54	3.84
	Zebrasoma scopas	0.08	X	X	0.06	0.17	X
	Naso hexacanthus	0.06	0.84	X	X	0.89	0.06
	Acanthurus lineatus	0.06	X	X	0.28	X	X
	Naso unicornis	X	X	X	0.08	0.03	0.08
	Acanthurus nigrofuscus	0.22	0.53	0.28	0.64	0.78	0.19
	Acanthurus thompsoni	0.22	1.09	0.89	0.50	0.45	2.56
	Acanthurus tennenti	1.03	1.89	1.45	2.03	2.20	2.89
	Naso lituratus	0.28	0.39	X	0.47	0.19	0.17

Table A3.4: Area standardised abundance (individuals/100 m^2) of species observed at shallow (6–14 m), intermediate (14–22 m) and deep (22–30 m) depths during winter and summer using videographic transects on Two-Mile Reef. Families and species within families are listed alphabetically. X = Species was not observed.

		Winter			Summer		
FAMILY	Species	Shallow	Inter-mediate	Deep	Shallow	Inter-mediate	Deep
ACANTHURIDAE (Cont.)	*Paracanthurus hepatus*	X	X	X	0.11	X	0.14
	Acanthurus dussumierii	X	0.11	0.06	0.06	X	2.12
	Acanthurus leucosteron	1.03	0.25	X	2.09	0.17	0.03
	Ctenochaetus strigosus	0.95	0.64	0.61	2.25	1.45	0.75
	Zebrasoma gemmatum	X	X	X	0.03	0.08	0.14
	Naso brevirostris	X	X	0.03	X	0.03	0.06
	Ctenochaetus striatus	0.33	0.22	0.33	1.20	0.67	0.33
	Acanthurus xanthopterus	X	X	0.06	0.03	X	X
APOGONIDAE	*Apogon angustatus*	X	X	0.06	X	0.03	0.03
	Cheilodipterus quinquelineatus	X	X	X	X	X	0.03
	Apogon kallopterus	X	X	0.08	X	X	X
	Cheilodipterus artus	X	X	0.03	X	X	0.06
BALISTIDAE	*Sufflamen bursa*	X	X	0.11	0.19	0.11	0.14
	Sufflamen fraenatus	X	0.22	0.17	0.08	0.06	0.19
	Balistoides conspicillum	0.08	0.06	X	0.03	0.03	0.03
	Sufflamen chrysopterus	0.78	1.17	1.20	1.45	1.11	1.45
	Melichthys indicus	0.22	0.14	0.17	0.28	0.25	0.11
	Balistapus undulatus	0.08	0.08	0.06	0.33	0.25	0.42
	Odonus niger	0.61	1.98	0.47	0.22	1.53	0.78
	Pseudobalistes fuscus	X	X	X	X	0.03	0.06
	Balistoides viridescens	0.03	X	0.06	X	0.03	0.08
BLENNIIDAE	*Cirripectes auritus*	X	X	X	0.03	X	X
	Aspidontus dussumieri	0.06	0.08	0.03	0.45	X	0.03
	Ecsenius midas	X	X	X	0.03	X	X
	Exallias brevis	X	X	X	0.06	X	X
	Plagiotremus tapeinosoma	X	0.11	0.11	0.25	0.11	0.06
	Cirripectes stigmaticus	X	X	X	0.08	X	X
	Plagiotremus rhinorhynchos	X	X	X	0.03	0.06	0.08
CAESIONIDAE	*Pterocaesio tile*	X	X	X	0.33	X	X
	Pterocaesio capricornis	X	X	X	X	X	1.48
	Caesio xanthonota	0.50	4.04	4.62	0.50	1.98	3.51
	Paracaesio xanthura	X	X	X	0.03	X	0.06
CARANGIDAE	*Caranx melampygus*	0.11	0.06	0.08	0.19	0.14	X
	Carangoides caeruleopinnatus	X	0.08	X	X	X	0.03
	Decapterus macrosoma	0.61	X	X	X	X	X
	Carangoides fulvoguttatus	0.03	X	X	X	X	X
CHAETODONTIDAE	*Chaetodon dolosus*	X	X	0.19	X	X	0.06
	Chaetodon blackburnii	X	0.03	0.14	0.03	0.03	0.06
	Hemitaurichthys zoster	0.25	0.22	X	0.78	0.14	0.06
	Heniochus acuminatus	X	0.06	X	X	X	X
	Chaetodon guttatissumus	0.36	0.86	0.28	0.19	0.64	0.47
	Chaetodon unimaculatus	0.75	0.92	0.42	0.84	0.70	0.33
	Forcipiger flavissimus	0.17	0.31	0.06	0.50	0.25	0.14
	Heniochus monoceros	X	X	0.06	0.03	0.06	0.06
	Chaetodon meyeri	0.22	0.11	0.14	0.31	0.17	0.03
	Chaetodon madagascariensis	0.33	0.86	0.95	0.72	0.75	1.06
	Chaetodon lunula	0.17	X	0.03	0.06	X	X
	Chaetodon trifascialis	X	X	X	0.03	0.03	X
	Heniochus diphreutes	X	2.39	0.50	X	X	1.00
	Chaetodon auriga	X	X	0.06	0.28	0.08	0.19
	Chaetodon vagabundus	0.06	0.22	0.17	X	0.06	0.06

Table A3.4: Area standardised abundance (individuals/100 m^2) of species observed at shallow (6–14 m), intermediate (14–22 m) and deep (22–30 m) depths during winter and summer using videographic transects on Two-Mile Reef. Families and species within families are listed alphabetically. X = Species was not observed.

		Winter			Summer		
FAMILY	Species	Shallow	Inter-mediate	Deep	Shallow	Inter-mediate	Deep
CHAETODONTIDAE	*Chaetodon kleini*	0.64	2.17	2.14	0.84	1.11	1.34
(Cont.)	*Chaetodon xanthocephalus*	X	X	X	X	0.06	X
CHARCHARHINIDAE	*Carcharhinus sealei*	X	X	X	X	X	0.06
CIRRHITIDAE	*Paracirrhites arcatus*	0.08	0.19	X	0.47	0.22	0.11
	Paracirrhites forsteri	0.19	0.06	0.06	0.17	0.08	0.03
	Cirrhitus pinnulatus	X	X	X	0.03	X	X
	Cirrhitichthys oxycephalus	0.11	0.42	0.39	0.45	0.61	0.39
	Cyprinocirrhites polyactis	X	X	X	X	X	0.08
DINOPERCIDAE	*Dinoperca petersi*	X	X	0.03	X	X	X
DIODONTIDAE	*Diodon hystix*	0.03	X	X	X	X	X
	Diodon liturosus	X	X	X	X	X	0.03
ECHENEIDAE	*Echenis naucrates*	X	X	X	0.03	X	X
EPHIPPIDAE	*Tripterodon orbis*	0.03	X	X	X	X	X
FISTULARIIDAE	*Fistularia commersonii*	0.03	X	X	0.03	X	X
GOBIIDAE	*Ptereleotris heteroptera*	X	0.03	0.56	X	0.39	0.31
	Nemateleotris magnifica	X	0.11	0.50	0.11	0.58	1.20
	Valenciennea strigata	X	0.06	X	0.11	X	0.17
	Ptereleotris evides	X	X	X	0.11	0.22	0.03
GRAMMISTIDAE	*Grammistes sexlineatus*	X	X	X	0.03	X	X
HAEMULIDAE	*Plectorhinchus chubbi*	X	0.03	0.17	X	0.06	0.06
	Plectorhinchus flavomaculatus	X	X	X	X	0.03	0.03
	Plectorhinchus playfairi	0.06	0.03	0.06	0.03	0.17	0.03
HOLOCENTRIDAE	*Myripristis berndti*	X	X	0.11	X	X	X
	Myripristis murdjan	X	X	0.56	0.45	X	0.81
	Sargocentron diadema	X	0.03	0.67	0.14	0.03	0.17
	Myripristis melanostica	X	X	0.06	X	X	0.36
	Sargocentron spiniferum	X	X	0.08	X	X	X
	Sargocentron caudimaculatum	X	0.03	0.42	0.06	X	0.19
LABRIDAE	*Coris giamard africana/ cuvieri*	0.03	0.03	0.06	X	X	X
	Bodianus axillaris	0.03	0.08	X	0.03	0.03	0.03
	Hologymnosus annulatus	0.03	X	0.03	X	X	0.03
	Hemigymnus fasciatus	0.03	X	X	X	X	X
	Hemigymnus fasciatus	X	X	X	0.03	X	X
	Labroides bicolor	X	0.06	0.11	0.08	0.17	0.08
	Gomphosus caeruleus	0.56	0.22	0.28	0.86	0.58	0.33
	Anampses caeruleopunctatus	0.06	0.06	0.03	0.25	0.06	0.06
	Macropharyngodon cyanoguttatus	0.03	0.03	0.03	X	0.03	0.03
	Labroides dimidiatus	0.53	0.72	1.86	1.25	1.36	1.84
	Halichoeres hortulanus	0.33	0.45	0.28	0.92	0.86	0.11
	Pseudodax moluccanus	0.22	0.14	0.08	0.31	0.19	0.06
	Cheilei inermis	X	X	X	0.06	X	X
	Coris aygula	0.06	X	0.03	0.06	0.03	X
	Thalassoma lunare	0.08	0.19	0.06	0.14	0.14	0.06
	Stethojulis interrupta	X	X	0.03	X	X	X
	Bodianus diana	0.06	0.22	0.22	0.11	0.36	0.14
	Macropharyngodon bipartitus	0.14	0.11	0.19	0.08	0.17	0.17
	Cirrhilabrus exquisitus	X	X	X	0.14	X	X
	Thalassoma hebraicum	1.95	2.51	1.84	2.78	3.42	1.95
	Bodianus perditio	0.03	X	X	0.06	0.03	X
	Anampses lineatus	0.53	0.84	1.14	1.53	0.81	0.81
	Anchichoerops natalensis	X	X	X	X	X	0.03

Table A3.4: Area standardised abundance (individuals/100 m^2) of species observed at shallow (6–14 m), intermediate (14–22 m) and deep (22–30 m) depths during winter and summer using videographic transects on Two-Mile Reef. Families and species within families are listed alphabetically. X = Species was not observed.

		Winter			Summer		
FAMILY	Species	Shallow	Inter-mediate	Deep	Shallow	Inter-mediate	Deep
LABRIDAE (Cont.)	*Hologymnosus doliatus*	X	0.06	0.03	0.06	0.03	0.03
	Halichoeres nebulosus	0.11	X	X	0.45	0.03	0.08
	Halichoeres iridis	X	X	0.39	X	X	0.47
	Cheilinus fasciatus	X	X	X	0.03	0.03	X
	Thalassoma genivittatum	X	X	X	0.08	0.06	X
	Bodianus bilunulatus	0.08	0.19	0.31	0.03	0.03	0.19
	Thalassoma hardwicke	0.03	X	X	0.03	X	X
	Pseudocheilinus hexataenia	0.03	X	0.03	0.03	0.03	X
	Pseudojuloides cersinus	X	X	0.03	X	X	X
	Coris caudimacula	0.39	0.78	1.17	1.34	1.78	1.78
	Pseudocheilinus evanidus	X	X	0.08	0.31	0.33	0.61
	Thalassoma amblycephalum	0.03	X	X	1.20	X	0.17
	Anampses twistii	X	X	X	0.03	X	0.03
	Anampses meleagrides	0.22	0.19	0.03	0.19	0.47	0.17
	Halichoeres scalpularis	0.03	X	X	X	X	X
LETHRINIDAE	*Monotaxis grandoculis*	X	0.03	0.08	X	X	0.06
	Gymnocranius griseus	X	X	X	X	X	0.03
	Lethrinus lentjan	X	X	X	X	X	0.03
	Lethrinus rubrioperculatus	0.08	0.08	0.08	X	0.03	0.39
	Lethrinus crocineus	0.03	0.06	0.03	X	0.03	0.08
LUTJANIDAE	*Aphareus furca*	0.03	0.03	0.06	0.08	X	0.03
	Lutjanus kasmira	X	X	0.03	1.22	1.73	X
	Lutjanus fulviflamma	X	X	X	1.45	0.06	X
	Paracaesio sordidus	X	X	0.72	0.03	0.03	X
	Aprion virescens	0.03	0.03	0.03	X	0.08	0.19
	Lutjanus gibbus	0.03	0.03	0.17	0.42	0.17	X
	Lutjanus rivulatus	X	0.03	0.03	X	0.03	X
	Lutjanus bohar	X	X	X	0.06	X	X
MALACANTHIDAE	*Malacanthus brevirostris*	X	0.08	0.06	0.11	X	0.08
MONACANTHIDAE	*Cantherhines dumerillii*	0.06	0.06	0.06	0.11	0.03	X
	Cantherhines pardalis	0.08	0.03	0.06	0.06	0.06	0.08
MUGILOIDIDAE	*Parapercis xanthozona*	0.03	X	0.11	X	0.06	0.08
	Parapercis schauinslandi	X	X	X	X	X	0.03
	Parapercis punctulata	X	0.03	0.11	0.08	0.08	0.25
MULLIDAE	*Parupeneus macronema*	0.53	0.84	1.11	0.39	0.97	0.84
	Parupeneus rubescens	0.03	0.36	0.17	0.06	0.14	0.33
	Parupeneus pleurostigma	0.06	0.08	0.17	0.06	X	0.25
	Parupeneus cyclostomus	0.06	0.31	0.11	0.14	0.17	0.42
	Parupeneus bifasciatus	X	X	X	0.06	X	X
	Mulloidichthys vanicolensis	X	X	0.89	1.73	X	X
MURAENIDAE	*Siderea grisea*	X	X	0.03	X	X	X
	Gymnothorax favagineus	X	X	X	X	0.03	X
	Gymnothorax eurostus	X	X	X	X	0.03	X
	Gymnothorax nudivomer	X	X	0.03	X	X	X
MYLIOBATIDAE	*Manta birostris*	X	0.03	X	X	X	X
OPLEGNATHIDAE	*Oplegnathus robinsoni*	0.08	0.03	0.03	0.03	0.08	0.06
OSTRACIIDAE	*Ostracion cubicus*	X	X	0.03	X	0.08	0.03
PEMPHERIDAE	*Pempheris adusta*	X	X	X	0.14	X	X
POMACANTHIDAE	*Centropyge multispinis*	0.97	0.92	1.75	1.86	2.14	2.95
	Centropyge acanthops	0.06	0.28	0.89	0.08	0.70	1.11
	Pomacanthus semicirculatus	X	X	X	0.03	X	X

Table A3.4: Area standardised abundance (individuals/100 m^2) of species observed at shallow (6–14 m), intermediate (14–22 m) and deep (22–30 m) depths during winter and summer using videographic transects on Two-Mile Reef. Families and species within families are listed alphabetically. X = Species was not observed.

		Winter			Summer		
FAMILY	Species	Shallow	Inter-mediate	Deep	Shallow	Inter-mediate	Deep
POMACANTHIDAE (Cont.)	*Apolemichthys trimaculatus*	0.06	0.06	0.17	0.08	0.08	0.08
	Chromis nigrura	1.98	0.56	X	26.11	2.70	1.89
	Pomacentrus caeruleus	0.11	0.61	0.17	0.45	0.14	0.22
	Chromis dasygenys	X	X	X	X	X	0.58
	Chromis dimidiata	7.01	10.69	4.62	16.06	20.63	9.91
	Chromis weberi	13.70	13.14	7.46	18.57	14.84	6.57
	Dascyllus trimaculatus	X	0.47	2.12	0.31	0.75	0.64
	Pseudochromis dutoiti	X	0.06	0.06	0.06	0.06	0.08
	Pomacanthus imperator	0.06	X	0.14	0.03	0.17	0.08
	Plectroglyphidodon dickii	0.31	0.08	X	1.00	0.08	0.03
	Amphiprion akallopisos	X	0.06	0.17	0.11	0.14	0.17
	Pomacanthus rhomboides	X	0.28	0.17	0.17	0.14	0.14
	Amphiprion allardi	0.08	X	X	0.03	0.08	0.06
	Dascyllus carneus	X	X	0.53	0.03	0.17	0.11
	Plectroglyphidodon johnstonianus	X	X	X	0.14	0.03	X
	Pomacentrus trichourus	0.06	0.08	0.03	0.33	0.14	0.08
PRIACANTHIDAE	*Priacanthus hamrur*	X	0.08	X	X	0.56	0.03
PSEUDOCHROMIDAE	*Pseudochromis melas*	X	0.06	0.03	0.31	0.06	0.22
	Pseudochromis natalensis	X	X	0.06	0.08	0.06	0.14
SCARIDAE	*Scarus ghobban*	0.03	X	X	X	X	X
	Scarus frenatus	0.03	X	X	0.03	0.06	X
	Calotomus carolinus	X	X	0.03	X	X	0.03
	Scarus rubroviolaceus	0.81	1.00	0.56	1.06	0.72	0.45
	Scarus scaber	X	X	0.03	X	X	X
	Scarus tricolor	X	0.03	X	X	0.03	X
SERRANIDAE	*Epinephelus macrospilos*	X	X	X	X	0.03	X
	Cephalopholis miniata	X	X	0.06	X	0.22	0.19
	Cephalopholis urodeta/nigripinnis	X	0.06	0.08	X	X	0.06
	Anthiinae	4.12	9.97	15.48	8.63	13.03	11.22
	Epinephelus rivulatus	0.03	0.03	0.11	0.08	0.14	0.03
	Cephalopholis argus	X	X	X	0.03	0.03	0.03
	Epinephelus tukula	X	0.11	X	X	0.03	0.03
	Aethaloperca rogaa	X	X	X	0.06	X	X
	Variola louti	0.11	0.08	0.17	0.22	0.14	0.28
	Cephalopholis sonnerati	0.03	0.06	X	0.06	X	0.06
SIGANIDAE	*Siganus sutor*	0.14	0.03	0.11	0.06	0.03	0.03
SPARIDAE	*Chrysoblephus lophus*	X	X	X	X	X	0.03
	Polysteganus praeorbitalus	X	X	X	X	X	0.06
SPHYRAENIDAE	*Sphyraena jello*	X	X	X	X	0.06	X
TETRAODONTIDAE	*Canthigaster valentini*	X	X	X	0.03	X	X
	Arothron immaculatus	X	X	X	X	0.03	0.06
	Arothron nigropunctatus	0.03	X	X	X	X	X
	Canthigaster rivulata	X	X	X	0.03	X	X
	Canthigaster valentini	0.03	0.06	0.06	X	0.03	0.03
	Canthigaster amboinensis	X	X	X	0.03	X	X
	Arothron hispidus	X	X	X	0.03	X	X
ZANCLIDAE	*Zanclus canescens*	0.33	0.47	0.19	0.42	0.56	0.50

Appendix 4 -

A preliminary protocol for the survey of high-latitude, epibenthic coral reef fishes in the Western Indian Ocean by videographic transecting

The following protocol is specifically designed for the non-destructive underwater visual census of high latitude coral reef fishes in the Western Indian Ocean using videographic transecting by SCUBA divers. This protocol has been formulated based on the conclusions of Wartenberg (2012). The protocol is presented in point-form, and without motivation, to avoid redundant reviews of the information presented at length in Wartenberg (2012), and outlines the most appropriate approaches to experimental design, the selection of survey technique parameters, the quantification of supplementary variables, observer-related standardisation, and standardised videographic transect deployment.

By way of example, the protocol outlined here is specific to the research aims of Wartenberg (2012) that include between season- and depth-category comparisons of ichthyofaunal community structure on a single, large, high latitude patch reef. Simple adjustments to this protocol will allow for alternative comparisons, such as between reefs or protection regimes.

If this protocol is to be used as it is intended, then it is to be used in conjunction with Appendices 1 and 2 from Wartenberg (2012) which provide necessary additional information. The species lists presented in Wartenberg (2012) Appendix 3 will prove useful. For further information, thesis meta-data, or digital examples of videographic transects and associated fish counts contact the South African Environmental Observation Network, Elwandle Node, South Africa.

Experimental design

Sampling stratification

- Select an appropriate, representative patch-reef based on the initial research questions of interest. In this case: Do coral reef ichthyofaunal community structures change between seasons or depths?

- Design sampling such that balanced samples can be achieved for each category/stratum investigated. For example, equal numbers of replicates in mid-winter and mid-summer.
- Restrict sampling to the leeward-reef slope, unless a comparison between fore- and leeward-reef fish communities is required.
- Conduct transects on substrate known to consist of reef. Avoid, or exclude from analysis, transects conducted on sandy areas.
- Structure sampling such that the maximum number of replicates can be achieved in the least number of days.
- Restrict sampling to a narrow time-window, e.g. 09h00 – 13h00, and ensure that the selected time-window remains constant throughout the study or monitoring program.
- Those factors which cannot be stratified for must be measured and/or estimated as supplementary variables.

Replication

- If a study is being conducted in a new locality, the maximum number of replicates logistically feasible is recommended for each stratum. There should be no less than 10 and no more than 25 replicates at this initial stage. From the resulting data, the optimal number of replicates specific to a study area can be determined using the methods outlined in Chapter 4 of Wartenberg (2012).
- It was shown that, for Two-Mile Reef, South Africa, the optimal number of replicates was a minimum of 17, but more favourably 20, for each stratum.

Randomisation

- Conduct sampling using a stratified-random sampling design without replacement.
- Using a geographical information systems software package, such as ArcGIS 10 (Environmental Systems Research Institute, Inc ©) divide a georeferenced reef map into potential sampling sites based on required strata characteristics and the size of the survey technique boundaries i.e. 50 m based on transect length.
- Of the appropriate sites, use a random number generator to select which sites within each stratum are to be sampled.

- Generate co-ordinates for each site to serve as transect start points.
- Generate additional co-ordinates as a contingency to disrupted or failed transects.
- Locate selected sites in the field using a handheld GPS (e.g. etrex, Garmin ©).
- Ground-truth sample site depths using a boat based echo sounder prior to diver deployment.

Supplementary variables

The following supplementary variables must be measured and/or quantified for each sample site.

Topographic complexity

- Record depths at the start of each transect and at each 10 m interval. Topographic complexity in terms of profile and rugosity can then be calculated as per Chapter 3 of Wartenberg (2012).
- In addition, use the independent, qualitative metric habitat complexity, which quantifies the reef architecture for each sample according to five predetermined categories: highly complex, semi-complex, complex, homogenous, highly homogenous. For visual examples of each habitat complexity category see Appendix 1 of Wartenberg (2012).

Substrate

- Quantification of substrate for each sample site was according to five predetermined categories: reef, reef-sand, reef/sand, sand-reef, or sand. For visual examples of each substrate category see Appendix 1 of Wartenberg (2012).

Current

- Both divers estimate the current speed at each sampling site and the mean obtained. Record current direction at each sampling site.

Visibility

- Measure horizontal visibility at the sampling depth for each sampling site. Post transect completion diver I must swim away from diver II along the extended transect line. The point at which divers can no longer distinguish one another is recorded as the visibility for that site.

Water temperature

- Record water temperature at the sampling depth using dive computers such as the Gecko (Suunto ©).

Turbidity

- Collect water samples at the sampling depth, place on ice, and freeze as soon as possible for analysis in a laboratory.

Survey technique and fish assemblage parameters

Survey technique parameters

- Record the time of day that each transect commences.
- Transect length = 50 m.
- Keep swimming speed constant at approximately 6.25 $m.min^{-1}$ = 8 $min.transect^{-1}$.
- If unconfounded comparability to the original thesis is desired, conduct video transects using a DX 1200 HD Camera (© SEA&SEA SUNPAK Co., Ltd.) set to underwater video mode. If this camera cannot be attained ensure that, at a minimum, a camera with precisely the same field of view is employed.
- If a direct comparison to Wartenberg (2012) is undesirable, then ensure that a sufficiently high-resolution video camera is employed such that ease of fish identification from digital footage is facilitated.

Fish assemblage parameters

- Exclude cryptic species from counts.
- Only fish larger than 4 cm may be included in counts.
- Those species which are particularly morphologically similar must be assigned to species groups. For a practical example see Appendix 3 (Table A3.2) of Wartenberg (2012).
- Fishes passing from behind divers are to be excluded from counts.

- Once an individual is counted, it is not recounted.
- Individuals clearly attracted to the area due to diver activity are not counted.

Observer-related standardisation

Observer and diver selection

- Observers must possess sufficient diving experience and skill to conduct replicated transects with negligible variability. One prerequisite is that all divers are Class IV commercial divers or higher.

Reconnaissance diving

- In areas which have not been previously studied, reconnaissance diving prior to study commencement is strongly recommended to facilitate the collation of necessary preliminary study site information. A number of SCUBA diver deployments using the roving diver technique, as carried out in the original thesis, are recommended for this initial stage.

Diver equipment

- Aside from standard SCUBA diving equipment, the equipment required to conduct effective, holistic underwater videographic transects is outline in Appendix 1 (Table A1.2) of Wartenberg (2012).

Diver training

- To reduce observer-related bias and standardise the methods used to conduct videographic transects, observer training is recommended.
- Holistic observer training should be conducted in terms of fish identification, transect width estimation, swimming speed, and the quantification of supplementary variables. Appendix 2 of Wartenberg (2012) outlines the most appropriate methods for this.

Standardised videographic transect deployment

- An effective diving vessel such as a rigid-inflatable boat (RIB) should be employed for deploying divers.
- All dives should be conducted according the dive plan which is appended to Wartenberg (2012) (Appendix 1, Table A1.1) and should not last longer than

21 minutes regardless of the operational depth up to 30 m.

- Upon arrival at the preselected sample site coordinates a dive team, deploy a dive team consisting of a pair of observers using the negative entry technique.
- Divers must maintain neutral buoyancy 1 m above the reef substrate throughout each transect.
- Once arriving at the sampling depth, divers must swim into the current for 10 m along the depth contour before establishing the transect start point.
- Each diver in a dive team, diver I and II, must conduct specific duties, outlined in Appendix 1 (Table A1.2) of Wartenberg (2012), which must remain constant throughout the study.
- Capture video footage at a 1 m elevation on a plane parallel to the slope of the reef.
- The distance and angle of the observable field must be kept constant.
- Fish assemblage data from transects must be obtained by analysing videographic transects using the *Standard-count* technique.

For an example of the effective implementation of this protocol in surveying high-latitude epibenthic reef fishes in the WIO see Chapter 5 of Wartenberg (2012).