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Parrotfish Fisheries and Population Dynamics;
a case-study from Solomon Islands

Thesis submitted by

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In the

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STATEMENT OF CONTRIBUTIONS

The collaborative nature of this work has meant that several people need to be acknowledged for contributing to the success of this work. As part of a larger investigation on parrotfish Biology, Professor J H Choat has been the main contributor of demographic field samples. Alec Hughes, as part of this investigation with professor Choat, has provided me with demographic and life-history data which allowed for comparative analysis with data collected as part of this work. Collaboration with Simon Albert on the Roviana and Vonavona Marine Protected Areas project has meant that an extra dimension has been added to this research project and I thank him for his complimentary data. Finally, and most importantly, funding acquired from Professor S Aswani as part of the Roviana and Vonavona Marine Protected Areas project must be acknowledged as significantly contributing to the overall success of this work.

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Armagan Sabetian Date

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Equally important has been the role of my wife Jillian as a supporter, motivator, and cornerstone of our family unit. I would not have been able to do this without her and my two children Liana and Ona.

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ABSTRACT

There is unanimous agreement that the functional role of parrotfish as lower trophic level herbivores is a key component in the structural integrity and resiliency of shallow water coral reef ecosystems. The premise for carrying out a comprehensive fishery, biological and ecological investigation of parrotfishes here was based on the recognition that more concerted research is needed in order to help address management and conservation challenges related to parrotfish fisheries. More importantly, it was decided from the outset that urbanized fisheries posed the greatest threat to coral reef fish assemblages, which prompted me to place the investigation of parrotfish ecology within the context of a typical artisanal fishery case study in the South Pacific (i.e. Solomon Islands).

I feel some vindication for the research theme adopted for this PhD given the recognition of both parrotfish assemblage decline and open-access urban fisheries by the 11th International Coral Reef Symposium, held in the international year of the reef (2008), as critical factors to the survival of coral reefs around the globe.

The fishery component of this research looked into artisanal catch landings, supported by independent creel and abundance surveys. The various types of data collected provided an effective form of triangulation. This study has described the first comprehensive investigation of an urbanised parrotfish artisanal fishery in Western Solomon Islands. Records of catch landings over a two-year period showed a transition from larger excavating parrotfish to smaller scrapping parrotfish. Although creel data did not validate the exact species-specific decline depicted in catch landings, the overall transitional shift of larger to smaller parrotfish was the same. Triangulation of catch, creel and abundance data showed that parrotfish density, large bodied fish numbers, and mean body size had all declined.

Having quantified fishing effort and assessed possible reactionary declines in density and size structure, I proceeded to investigate age-based demographic and reproductive parameters in order to discuss parrotfish resiliency to fishing pressure, but also to gain a comparative perspective with regards to age and growth plasticity from other ocean systems. There was considerable variation in demographic parameters, even within same ocean systems. Parrotfish from Solomon Islands have shorter life span, grow fast, and are very young at first female maturity. However, their regenerative capacity from fast growth and high fecundity appears to provide insufficient resiliency against quantified fishing pressure.

The issue of parrotfish conservation was studied through the analysis of abundance data and herbivory rates within and outside of a marine protected area. Algal tile and fish surveys demonstrated higher grazing activity and parrotfish abundance within the reserve, which appear to suggest that parrotfish do benefit from protection.

This study provides important biological, ecological and fishery information of an urbanized parrotfish fishery, while also conducting a comparative demographic and life-history analysis. Information gathered as part of this research program not only contribute to the knowledge-base of parrotfish fishery management and conservation, but can also be used a base-line for future research initiatives.

TABLE OF CONTENTS

Chapter Five: Parrotfish Reproductive Biology; a comparative analysis....... 144

LIST OF FIGURES

LIST OF TABLES

Chapter One

Management of tropical reef fisheries; a dynamic challenge

1.1 Introduction

Fishing pressure is the greatest threat imposed on coral reefs (Dalzell et al., 1997; Mclanahan 2002). Although millions of people are directly and indirectly dependant on coral reef fishes for social and economical wellbeing, the fishery itself is afforded little attention in terms of funding and research compared to industrialised fisheries (Sadovy 2005). Further to this, a large percentage of people dependant coral reefs live in countries that are severely underdeveloped and which will see doubling in population in less than 50 years (FAO 1999). The distribution of this increasing population is of more concern as the movement of people looking for employment will invariably lead to the concentration and expansion of urbanised centres, thus leading to intensified fishing pressure (Adams et al., 1996, Munday 2000). In fact, the United Nations has stated that by the end of 2008 more than 50% of our planet's population will be living in urban centres as survival in rural areas has become increasingly difficult.

Sadovy (2005) lists a number of factors that are impeding effective management of coral reef fisheries, most important of which include:

- Perceptions of low economic value,
- misdirected non-governmental organisation (NGO) initiatives,
- lack of effective local, national or international management measures,
- poor understanding of biological and ecological systems, and
- little knowledge of species resilience to local fishing pressure

I believe that these problems are further compounded by

1) Lack of consensus among reef scientists in relation to prioritising problems and proposing effective solutions,

2) regional variation in terms of socio-economic and ecological parameters, and

3) continually changing dynamics in terms of global economical and ecological factors.

From a biological perspective, I have highlighted four key issues that are central to the design of effective management and conservation measures in coral reef fisheries:

1) *The capacity of exploited species to respond to non-subsistence levels of fishing pressure.*

Although the consequences of industrial fishing have been well documented and publicised in the past, much less attention, in terms of research, has been directed to the effects of subsistence or artisanal fishing. Hawkins and Roberts (2004) attribute this to the public perception that artisanal fisheries are pursued by small-scale fishers using traditional methods, and as such are viewed as environmentally benign. This perception is misguided and inaccurate given the increasing use of mechanised fishing gear, motorized boats, and SCUBA by artisanal fishers (Sabetian and Foale 2006).

Coral reef fisheries may not be commercially viable in terms of international export for many developing tropical countries, but significant local trade is still conducted (Adams et al. 1996; Gillett and Lightfoot 2001; Gillett and Lightfoot 2002). For example, in Solomon Islands, Fiji, and Papua New Guinea, significant inter-provincial trade in fresh marine organisms does take place (Pers Com. Bruno Manele, WWF Solomon Islands; Dan Afzal, WWF Fiji, Roger James, Conservation International). In reality, the demands of growing urban populations are cost-effectively being met through local or inter-regional export. This is certainly the case in Solomon Islands, where I have personally witnessed trade in large quantities of freshly caught fish from

Gizo, the capital of Western Province, to the national capital Honiara. However, given the cash economy in many of these nations, there is very little official historical data in term of species distribution or catch quantities. As a result, there is a dearth of quantitative information on the resilience of marine species to increasing artisanal fishing pressure.

The problem is further compounded by the fact that the multi-species nature of coral reef fisheries can disguise overfishing of some individual species (Adams et al., 1996). Hamilton (2004), for example, has shown that artisanal fishers have overfished Bumphead parrotfish stocks in Roviana Lagoon of the Western Solomon Islands. However, the status of other scarids within the same area are still unknown.

2) *Lack of detailed comparative biological and ecological information*

Understanding the processes that drive and regulate the local population dynamics of coral reef fishes is central to effective management of fisheries (Hixon 1998). Information on life-history, age and growth-based demography is of fundamental importance. The reason for this is two-fold; not only such data are primary requirements in any quantitative fisheries stock assessment (Hilborn and Walters 1992), they also provide the most appropriate metric for comparing populations of reef fishes by habitat, bio-geographic region and taxonomic groupings (Choat et al. 2006). Although life-history and demographic data on temperate, open-ocean, and deep-water exploited fish stocks are readily available in the literature, the same is not comparatively true with regards to coral reef fishes (Choat and Robertson 2002).

Accurate demographic and biological data are critical in assessing the scale of exploitation problems and consequent management designs (Choat et al. 2006). However, management is further complicated as coral reef fishes display high complexity in demographic and life-history parameters, sometime within very short distances (Gust 2000, Choat and Robertson 2002). A number of possible triggers have been proposed, ranging from water temperature, nutrient level, and wave exposure, to other specific latitudinal environmental characteristics. Current research form the

University of Auckland (New Zealand) is focusing on the role of cellular oxidative stress in spatial growth plasticity of temperate fish species.

Hence, access to comparative growth and reproductive information on fish populations from different areas is highly beneficial to localised management and conservation designs. More importantly we also need to know if sustained fishing pressure can alter demographic and life-history patterns.

3) Over-exploitation and possible links with life history traits.

Recent studies (Jennings et al., 1999, Dulvy et al., 2003) have promoted the idea that vulnerability to over-fishing may be accentuated by large body size, which is associated with conservative life-history patterns such as slow growth and low fecundity. In particular, the notion that large body size can be used as a vulnerability indicator has been proposed. It's worth considering this in a bit more detail as large body size is often associated with long-life and slow turnover rate. However, more recent studies (e.g. Choat et al., 2006) have shown that some large species can have relatively rapid growth rates. The fact that we do know if there is a strong negative relationship between density and size-rarity of large species is a real problem for management.

In general, small to medium sized fish are characterized by fast growth, shorter life span, young age-at-maturity, and serial spawning periodicity, which are characteristic of r-selected species. As a result, these species are able to display faster turnover rates. On the other hand, K-selected species display slower growth, older age-at-maturity, large size, longer life-span, and infrequent spawning periodicity. The conservative life history characteristics of these species, make them more vulnerable to overfishing.

Polunin and Graham (2003), reiterate the above points by stating that "species with 'fast' life histories are generally expected to be less vulnerable to exploitation and have higher rebound potential than species with 'slow' life histories". In fact, Polunin and Graham (2003) propose to address the paucity of data on tropical reef fish by using ‘surrogates’ of life-history traits as indicators of vulnerability to exploitation, thus exercising precautionary or as they state ‘data-unintensive’ coral reef fisheries management.

However, in a recent study on one of the largest teleost fishes associated with coral reefs (*Cheilinus undulatus)* Choat *et al*., (2006) appeal for careful re-examination of the link between large size and vulnerability to over-fishing of this species, as proposed by Sadovy et al., (2003a). The fact remains that coral reef fishes display dynamic growth rates and strong phylogenetic patterns in life history parameters (Gust 2001; Choat et al., 2003), thus the extent to which adverse effects of fishing can “in part” be attributable to conservative life-history features must be carefully examined.

4) The capacity of coral reef ecosystems to respond to protection.

The principle motives behind the establishment of Marine Protected Areas (MPA) are several, but the overwhelming majority of managers and researchers advocate it as a precautionary and data-less conservation and/or management tool (Roberts and Polunin, 1991; Callum et al. 2001, Russ and Zeller 2003, Aswani et al. 2007).

The current debate on MPA surrounds the issue of effectiveness. A study by Halpern and Warner (2002) reviewed 112 independent measurements of 80 reserves to show that “the average values of density, biomass, average organism size and diversity inside reserves reach much higher levels (relative to control groups) within a short period of time, usually one to three years”. Although the majority of the reviewed studies did report positively with robust conservation benefits, fishery benefits of MPA remain controversial (Callum et al., 2001).

In practical terms, the expectations behind the establishment of MPA are centred around the belief that they have the ability to manage coral reef fisheries by protecting the habitat of a critical spawning stock biomass which in turn could sustain or rehabilitate adjacent fisheries via net exportation of adults through the ‘spillover effect' and through recruitment supply via larval dispersal (Russ 2002). This expectation has been shown to be reasonable "given enough time" (Alcala et al., 2005). Russ and Alcala (Russ et al., 2003) emphasised this long-term approach by pointing out that differing life-history characteristics and demographic plasticity of most reef fish complicated matters, suggesting the time required for population recovery from spillover, a density driven process, may involve decades.

There is still a lack of empirical data on rates and pattens of increase in density and biomass of target species. This is further complicated by the fact that the capacity of protected species to respond to removal of fishing pressure is both taxon and demography specific (Russ and Alcala 2004).

Although marine reserves offer a precautionary approach by allowing management or conservation to occur without detailed technical and scientific capacity, their effectiveness is also dependent on education, awareness, and as Russ and Alcala (1996, 2004) point out, local fishing community support and the response of fish populations to reduced exploitation. Hence, the integral role stakeholders must play in order to achieve a successful outcome. As such, the effectiveness of MPA systems as conservation and management tools remain controversial.

The four issues highlighted above are important ecological and biological issues that need to be addressed if coral reef fisheries are to be conserved and managed effectively. However, as stated by McClanahan (2002), coral reef conservation and management is further complicated by regional factors. Central to this thesis is the notion that urbanisation of coral reef fisheries has now become the key issue in management and conservation efforts in Oceania.

1.2 The emerging issue of urbanised fisheries in Oceania

A number of factors have contributed to fisheries mismanagement in Oceania, including lack of data and inadequate scientific models, ignorance about natural and customary tenure systems, weak governance (often related to political fragmentation), and the lack of focus on scientific, technical and logistical capacity (Sabetian and Afzal 2004). However, population growth and distribution is a serious threat to coastal coral reef resources in the Pacific (Zann 1994). Internal migration towards urban centres in seek of employment has ensured that population distribution is skewed and concentrated in specific regions. To compound this problem, no new infrastructure is being built, which means that existing Pacific Island towns are struggling to meet the demands of their growing population.

This issue was discussed by Rapaport (1999), highlighting the fact that limited resources and financial capital leads to the concentration of logistical capabilities around a handful number of urban centres, making these locations the point of intensified commercial agriculture and fishing trade. Furthermore, he also points out that most Pacific Island countries towns and cities are growing faster than their respective national population rate, leading to exceptionally high population densities comparable with those in the most highly populated Asian cities. For example, Solomon Islands' Western Province population growth rate is 3.2%, significantly above the national average of 2.8%.

Laroche et al., (1997), observed that in relation to artisanal fisheries of Madagascar the lack of transport infrastructure has led to a concentration of fishing effort close to urban centres. Although Rapaport (1999) and Zann (1994) points out that coral reefs in smaller Polynesian and Micronesian countries appear to be suffering more broadly from the effects of concentrated population growth associated with urbanisation, larger urban town and cities in Melanesian countries are not immune from this.

To highlight the above point, I will use Tuvalu and Papua New Guinea, as comparative examples. Tuvalu's capital, Funafuti atoll, where nearly half of Tuvalu's 12000 population live, is a narrow stretch of land between 20 and 400 meters wide in some areas, and totals 2.4 km², approximately 10% of Tuvalu's total landmass. As expected, its nearshore fisheries resources are intensively utilised to meet nutritional and artisanal demand both on Funafuti, and also for local export to other atolls such as Tarawa. Negative impact on local herbivorous fish stocks and subsequent ecosystem changes have been reported by Gillett and Moy (2006) around Funafuti. The situation is equally bad on Tarawa atoll, where the severe decline of nearshore fish resources has meant that the local population have had to supplement their diet by importing food and also by exploiting migratory Tuna species, the stocks of which are fortunately plentiful (Pers com., Brian Tufi, local resident: April 2009).

Papua New Guinea, on the other hand, is the biggest developing country in the South Pacific, boasting a total landmass of approximately 465,000km2, including nearly 600 islands, and a population of 5.9 million people (Official 2005 statistics). It has between 26 and 30 urban centres, depending on various references, the biggest of which are Port Moresby, the nation's capital, and Lae, the capital of Morobe province. Overall, the urban centres of Papua New Guinea are home to approximately 18% or 1 million of the total population.

Munday (2000), reported that the nearshore fisheries of Port Moresby, which had an estimated population of 250,000 at the time, appeared to be under threat from overfishing. More recent comprehensive quantitative small-scale artisanal fishing data from Alotau, the capital of Milne Bay province, which is home to approximately 10,000 inhabitants (2000 census), and Lae, the capital of Morobe province, which is home to over 120,000 inhabitants (2000 census) appears to confirm the notion that PNG's urban fish markets are showing signs of overfishing (Kaly and Preston 2006/2007).

However, as Adams et al., (1996) point out, the multi-species nature of coral reef fisheries is such that signs of overfishing on particular species, and consequent 'tip-off' points, are difficult to identify. This problem is compounded by the lack of base-line artisanal and subsistence data in peer-reviewed literature, which can be used to compare historical mode of fishing and catch rates, and also to identify warning signs and estimates of future trends. That is not to say that such data do not exist. Zeller et al., (2006) highlight the fact that some information may be hidden in grey literature reports, are unpublished, or form part of published studies with a primary focus other than catch reporting. I have personally observed the collection of artisanal market catch landings by NGO conservation staff, only later to be told that the data collection was ceased because either the project focus had changed or ongoing funding was no longer available. Gaps in subsistence and artisanal fisheries knowledge-base can be overcome if representative data are used as a point of reference for estimating future trends (e.g. Rhodes et al. 2008).

However, we must not underestimate the practical difficulty, not to mention the labour intensiveness of routinely recording species, catch, and size estimates from a multi-species fishery. From my own experience, such an exercise can be very time-consuming and expensive, not to mention inconvenient for fishers who have just returned from a day (or night) of fishing and are eager to sell their catch as soon as possible. That is not to say that other methods, such as creel surveys (see Rawlingson et al., 1996), cannot be used for this purpose. However, as highlighted, creel surveys on their own cannot provide an accurate picture of a multi-geared, multi-species fisheries. As a result, a more concerted effort concentrating on a limited number of families and fishing gears would be a good start.

I have so far identified specific biological and ecological issues that need to be addressed, and placed them in an urban fisheries context. Specifically, urban fisheries, for the purposes of this research, refers to the fishing activity of artisanal fishers from urban centres. The town of Gizo (on Ghizo island) in Solomon Islands was chosen as the case-study location, and parrotfish as the case-study clade for this study. Below I provide background information on both.

1.3 Solomon Islands

Solomon Islands is situated in the South-west Pacific, positioned at latitude 5°S to 11°55'S and longitude 155°30'E to 162°55'E. It consists of six large islands followed by over 900 other small and medium sized islands, many of which are uninhabited. Ghizo island (Figure 1.1), the capital of which is Gizo town (note different spelling between the island and township), serves as the provincial centre of the Western Province, which includes the island groups of Rannonga, Rendova, Vella Lavella, Kolombangara, Simbo, Shortlands, and the lagoonal systems of Vonavona, Roviana, Marovo.

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Figure 1.1. Western Province, Solomon Islands. (source: Google maps)

Gizo town (figure 1.2) is the second biggest urban centre in Solomon Islands, and focal point of trade for the island's near 8000 population (Otter 2002). Today, the inhabitants of Ghizo are a mixture nearly all ethnic groups from around the Solomons. Records show that areas around Ghizo have been subject to artisanal and small-scale commercial fishing since the turn of the 20th century (Eginton and James 1979). However, the increasing urban population (around 4.5% per annum) has meant that the percentage of income generated from informal economic activities such as fishing has been steadily increasing (Otter 2002).

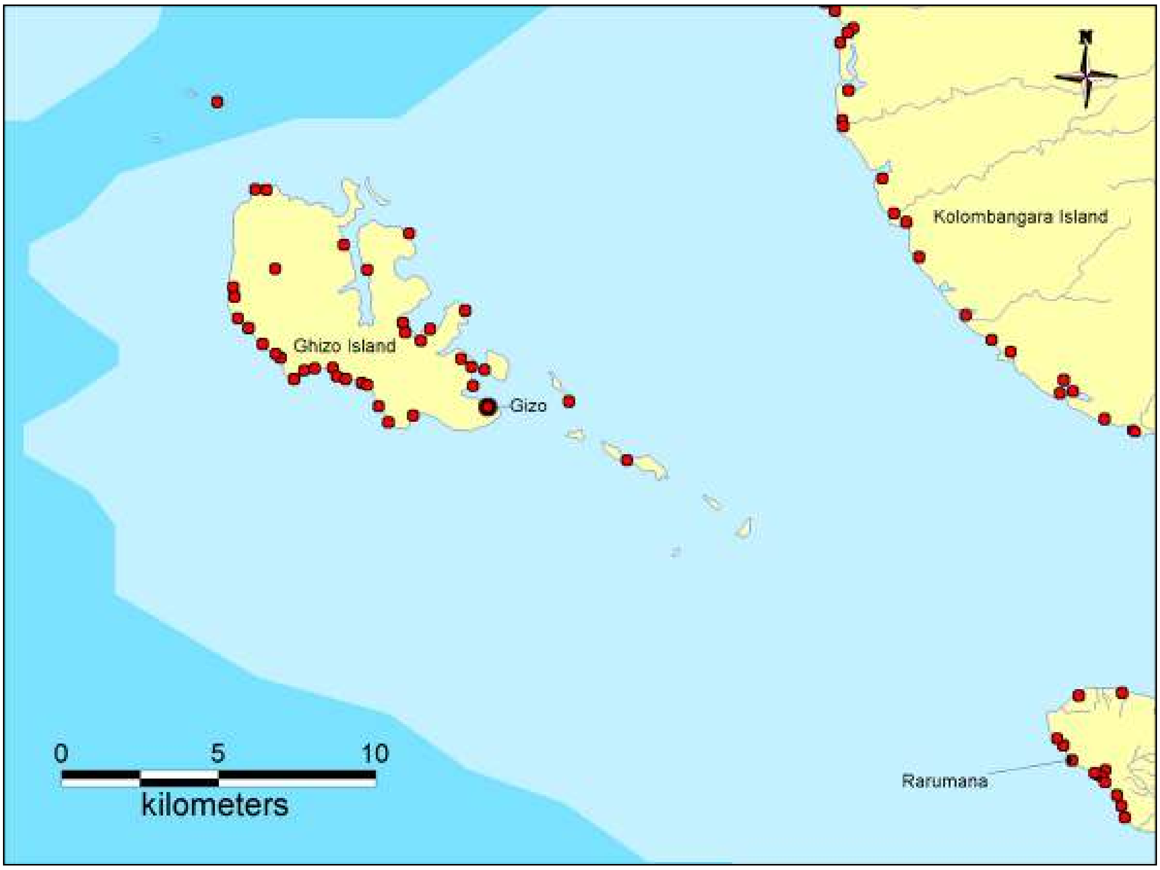

Figure 1.2. Map of Ghizo island showing location of Gizo town and proximity to the island of Kolombangara and the area of Rarumana on Kohingo island. (Source: Sabetian & Foale, 2006)

Gizo town is by far the biggest fish and food market in western Solomons. It is a bustling hub for marine and agricultural produce which are brought for sale from all around the region. The local economy revolves around public services and tourism, but the food market which until recently (for security and loitering reasons) was open 24 hrs seven days a week, attracts people from all around the province including fishers and agricultural produce sellers from other provinces such as Choiseul and Santa Isabel. In simple economic terms Gizo provides the biggest concentrated population of potential customers for local fishers and farmers.

Sabetian & Foale (2006) point out that reef tenure around Ghizo island is complicated and ambiguous as most of the reefs targeted by artisanal fishers are adjacent to ‘alienated’ land which was purchased from its former traditional custodians during the colonial era, and was converted to leasehold land presently held by the government. This basically means that access to reefs around Ghizo is open with no customary mechanism for enforcing restrictions.

The customary link theory was put forward by Dalzell et al., (1996), describing a situation where increasing migration into urbanised centres by people seeking better incomes and who are disenfranchised from customary links with the surrounding reefs pose the greatest threat to local fish populations. The scenario described by these authors is identical to the situation on Ghizo, where an increasing multi-ethnic population is utilising a range of economic opportunities, such as fishing, to provide an income for themselves and their families.

Dalzell et al., (1996) also reported that concentrated fishing effort will pose the greatest threat for trophically important and vulnerable species that have a limited range, restricted to shallow water habitats and defenceless against certain fishing methods. They also pointed to the fact that multi-species reef fisheries can disguise overfishing of some individual species, pointing to anecdotal evidence of spearfishing increase in Fiji and Micronesia and subsequent herbivorous fish decline.

In 1998, a fishery, ecological and biological survey into parrotfishes in American Samoa (Page 1998) became one of the first comprehensive studies to draw attention to warning signs and highlight possible effects of localised herbivorous fish declines on the reef ecosystem. During a two-year fieldwork period, Page (1998) only sighted one bumphead parrotfish (*Bolbometapon muricatum*). Today, species such as the bumphead parrotfish and the humphead wrass (*Chelinus undulatus*) have also all but disappeared from places such as Fiji (Sadovy 2005).

In a more recent report, Gillett and Moy (2006) highlighted the issue of spearfishing and the increasing use of SCUBA in some Pacific Island countries. Although, the vulnerability of herbivorous fishes such as parrotfishes and surgeonfishes to spearfishing was initially highlighted by Page (1998), their localised decline has also been linked to ecosystem changes in Funafuti, the heavily populated capital of Tuvalu (Gillett and Moy 2006). The problem of urbanisation is compounded by the fact that many Polynesian countries, like Tuvalu, are land deficient, and in this case over 6000 of the total 12000 inhabitants of Tuvalu live on the narrow Funafuti atoll.

Given the functional importance of herbivorous fish to the structural integrity of coral reef ecosystems (Hughes 1994), parrotfish were chosen as a case-study group for this study.

1.4 Parrotfish fisheries

1.4.1 Scarine labrids

Parrotfishes (also referred to as scarids) are a subgroup from the Labridae family and one of the most important groups of herbivorous fishes found on coral reefs with about 90 known species to date (Streelman, Alfaro et al. 2002). Although phylogenetic and evolutionary analysis of parrotfishes is still ongoing, it is now accepted that they are a clade in the tribe Cheilini, and more commonly now referred to as scarine labrids (Westneat and Alfaro 2005). The majority of scarids are known to reach maximum Standard Lengths (SL) of between the 200 and 500 mm. However, some species such as the bumphead parrotfish (*Bolbometopon muricatum*) do regularly reach 1000 mm (SL) (Choat et al. 1996) and have more recently been shown to attain maximum length in excess of this estimate in remote regions of Solomon Islands (Hamilton 2004). Unlike many other genera of reef fish, parrotfish are of crucial functional importance within the reef ecosystem. With their beak-like fused jaws they graze continually over reef surfaces either by scraping up algae and detritus or by excavating the reef surface itself (Bellwood 1995 a; Bellwood 1995 b; Crossman et al. 2001; Choat et al. 2002; Hodgson and Liebeler 2002). According to these studies this continuous grazing has two major effects on the reef ecosystem; Intensive feeding especially by smaller scraping parrotfishes continually removes algal material from reef surfaces and prevents overgrowth of algae, scleractinian corals and other sessile animals, while the larger excavating parrotfishes have the ability to remove and redistribute as sediment up to 30 kg $^{m-2}$ of calcareous material per year (Bellwood 1995 a; Bellwood 1995 b). Both activities are of fundamental importance to the functioning of coral reefs where maintaining a healthy herbivorous fish population is a key element in the effort to control algal growth that otherwise threatens to overtake the coral reefs (Hughes 1994; TNC 2002).

Scarids are circumtropically distributed, although the majority of the taxa are confined to the Indo-Pacific region (Streelman, Alfaro et al. 2002). Their wide distribution, high abundance on coral reefs (Choat and Robertson 2002), and their pivotal functional role within coral reef ecosystem make them an ideal group for investigation.

The 11th International Coral Reef Symposium in 2008 highlighted a range of concerns which are currently undermining the health of coral reefs worldwide. Consensus was reached on 10 measures, which if carried out would alleviate stress on coral reef ecosystems and help ensure their long-term survival. Two recommendations, which are interest to this study, are cited below:

Protect coral reef herbivores, including parrotfish. Ban the harvesting of these species for sale and commercial consumption.

Eliminate open access fisheries in coral reef ecosystems; Establish and enforce regulations on user rights, total allowable catch, individual catch quotas, non-destructive gear and other sustainable fisheries regulations.

1.4.2 Historical fishing trends

Subsistence and artisanal fishers consume and extract a wide range of "non-imported" or "locally-sold" species (Adams et al. 1996), the stock status of which we know little about. Parrotfish, which have long been harvested by subsistence and artisanal fishers around the world (Munro 1983; Polunin and Roberts 1993; Jennings and Polunin 1995; Salad Sabrie 1995; Dalzell et al. 1996; DPU 1997; FAO 1997; Page 1998; CRTF 1999; Wing and Wing 2001; FAO 2002b; Narriman and Ohman 2002; NOAA 2002b; Hawkins and Roberts 2003; Hawkins and Roberts 2004; Robinson and Shroff 2004; Wilkinson 2004; Gillett and Moy 2006) are one such group.

Historically more intensively targeted in the Caribbean (Munro 1973; TNC 2002), parrotfishes have also been increasingly fished across other ocean systems such as the Indian and Pacific (Dalzell et al. 1996; Craig et al. 1997; Page 1998; Narriman and Ohman 2002; Hawkins and Roberts 2004). Historical and recent catch data highlight parrotfishes as an important component of subsistence and artisanal fishing in the Western Pacific (e.g. Connell et al. 1998; Mackay 2001; Munday 2000).

Some of the earliest information on exploited parrotfish populations was published in Wing & Wing (2001) where prehistoric analysis of fishing pressure and its effects on coastal reef fish populations in the Caribbean region was investigated through archaeological records. Munro (1983) collated a cross section of fishery, ecological, and biological studies from the Caribbean dating back to the 1960's. Parrotfishes were described as being of high economic value to the local economy making up a large component of trap catches from coastal areas (Munro 1973). Trapping fish has historically been the most preferred method of fishing in the Caribbean followed by hook and line, nets, and spearfishing (Munro 1973). At the time, Munro (1973) described spearfishing as benign and "tend to harvest the oldest, largest, least-productive members of the fish community, thus promoting conservation of targeted fish stocks". Today of course, spearfishing has become more widespread in the Caribbean, with the use of SCUBA a growing problem in some areas (Gillett and Moy 2006).

Catch figures from some Caribbean countries indicate that fishing pressure on scarids has remained high where netting, trapping, and increasingly spearfishing, are used effectively to a point where stocks have been severely deplete (TNC 2002; NOAA 2002b). For example, estimated landings from St.Croix (Figure 2.1 and 2.2) confirm that trap and net fishing methods are disproportionately more effective with parrotfishes compared to other families.

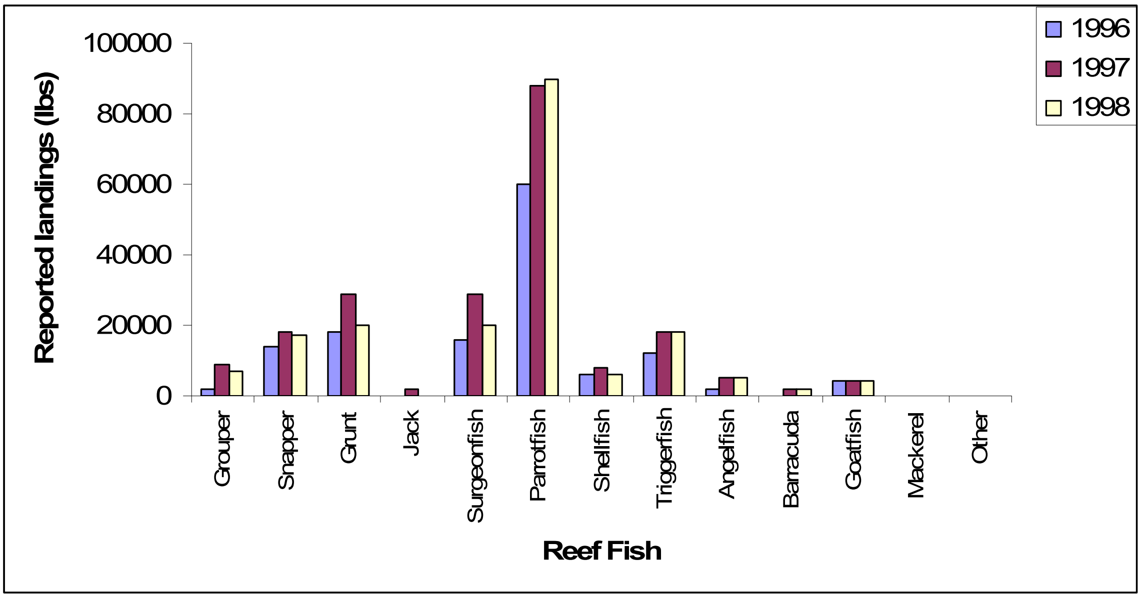

Figure 1.3. Estimated commercial catch composition for Trap Fishery in St. Croix. (NOAA 2002b)

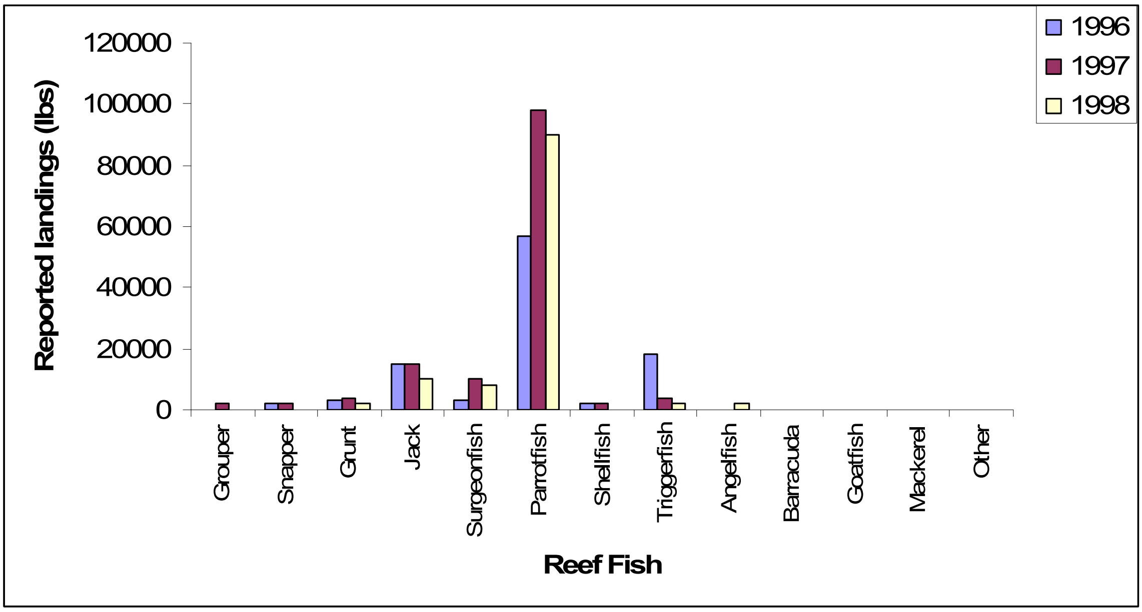

Figure 1.4. Estimated commercial catch composition for Net Fishery in St. Croix. (NOAA 2002b)

In Pacific Islands, Gillett and Moy (2006) report that spearfishing is by far the single most effective methods with which scarids are hunted. For example, house hold surveys conducted by Mackay (2001) revealed that in Eastern Samoa nearly 80% of households questioned owned underwater spears, with nearly 50% also owning underwater torches for night spearfishing. Hamilton (2004), looked at Bumphead parrotfishery in Solomon Islands and concluded that night spearfishing was in a larger part responsible for the decline of this species within western Solomons.

In a worldwide report on coral reef fisheries Wilkinson (2004) stated that the use of SCUBA in night spearfishing was notably increasing in the Pacific and that it was only a matter of time before such fishing techniques are commonly used across the entires region. Page (1998) had previously highlighted this issue in American Samoa where the use of SCUBA had become the primary method of conducting night spearfishing. Figure 1.5 shows historical harvest data of parrotfish on the main island of Tutuila, where by 1997 SCUBA had become the single most preferred method to target parrotfish.

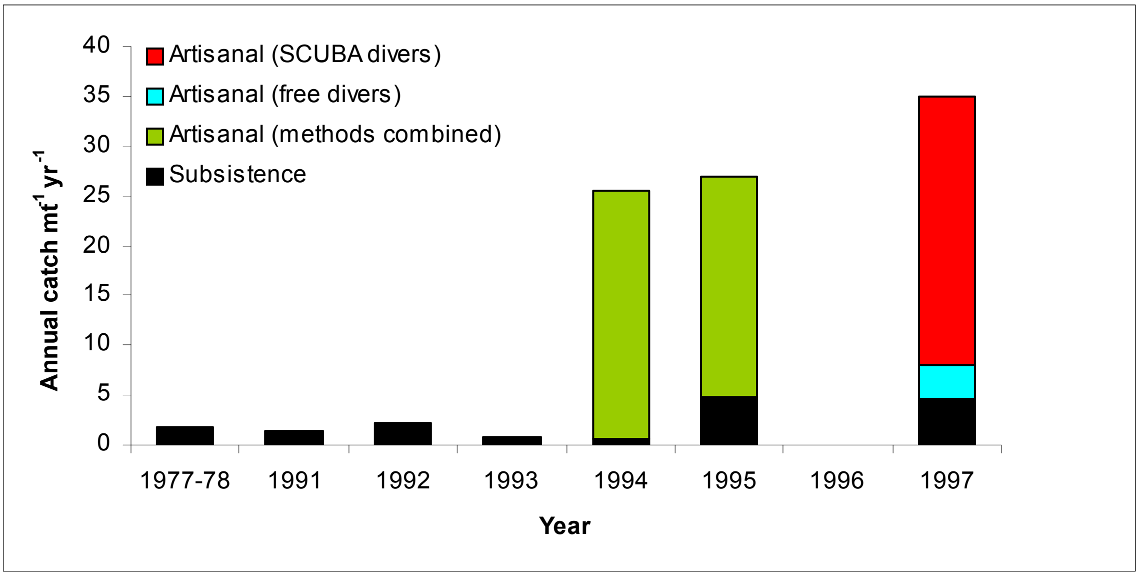

Figure 1.5. Estimated annual harvest of parrotfish on Tutuila island (Page, 1998).

In terms of subsistence catch trends, evidence points to good parrotfish stocks, particularly in larger countries such as Solomon Islands (Samoilys et al. 1995; Sabetian 2002). However, parrotfishes make up a more substantial component of the inshore fisheries in smaller Polynesian countries such as Samoa (Mackay 2001) and Tonga (FAO 2002a), and have been severely depleted around Tuvalu's capital Funafuti (Gillett and Moy 2006).

More recently, comprehensive small-scale fisheries and socio-economic surveys have been conducted in Papua New Guinea (Kaly 2006; Kaly and Preston 2006/2007). As part of these investigations, detailed catch landing data in combination with the artisanal component of two large provincial fish markets were routinely recorded for; 1) Alotau, the capital of Milne Bay province, which is home to over 200,000 inhabitants, and 2) Lae, the capital of Morobe province, which is home to over 500,000 inhabitants. Interestingly, no fish above 50cm (FL) were landed within both markets, which may be an indication of heavy fishing pressure overall in both regions. This notion seem to be supported by the fact that although Trevallies, Snappers, Groupers, Tunas and Mackerels made up a large percentage of total numbers landed, their overall size-distribution was small with only 7%>30cm(FL) in Alotau and 8%>50cm(FL) in Lae.

Munday (2000) stated that unsustainable fishing in Papua New Guinea, including both overexploitation and destructive practices, will inevitably lead to localised crashes near major population centres, advocating that abundances and size frequencies of reef fish must be monitored around growing urban areas, "as these are the areas at most risk and where well designed monitoring programs can have useful and applied outcomes".

1.4.3 Conservation and management

A growing body of evidence is showing that vulnerability of parrotfishes to fishing pressure is accentuated by their schooling and shallow water sleeping characteristics, which makes them particularly vulnerable to night spear fishers (TNC 2002; Hamilton 2004). Gillett and Moy (2006) have highlighted the issue of spearfishing and the increasing use of SCUBA in some Pacific Island countries. The vulnerability of low-trophic level herbivorous fishes such as scarids and acanthurids to spearfishing was highlighted by Page (1998) in American Samoa where they made up a significant proportion of artisanal and subsistence catches by weight. In some Pacific Island countries they are almost exclusively hunted via spearfishing, while their localised decline has been linked to ecosystem changes through increased filamentous algal growth and trophic vacuums being filled by ciguatera organisms (Gillett and Moy 2006).

In many parts of the Pacific herbivorous fishes are highly prized food fish and make up the majority of subsistence and artisanal spearfishing catches (Craig et al. 1997; Page 1998; Gillett and Moy 2006). Given this fact, herbivorous fishes appear to bear the brunt of increased fishing pressure (McClanahan 2002) around developing urban centres. However, conservation and management initiatives are rarely exercised in growing urban areas for a variety of reasons.

In 2005, the Virgin Islands department of planning and natural resources, with the assistance of The Nature Conservancy (TNC), tabled the U.S. Virgin Islands Marine Resources and Fisheries Strategic and Comprehensive Conservation Plan. The report highlighted conservation of herbivorous fishes as the main priority, stating that not only it was an important market sought by local fishers, but that herbivorous fish have an integral role in maintaining the structure of shallow-water reef communities. However, as pointed out in the report, the design of a long-term effective management plan requires an accurate understanding of their ecology and life-history.

Some of the earliest basic studies on parrotfish were summarised in Reeson (1975). However, over the past decade studies investigating various aspects of scarid phylogeny, ecology, demography, life-history, herbivory and bioerosion have continued to gain momentum (Harmelin-Vivien et al. 1992; van Rooij et al. 1995; Bellwood 1995 a; Bellwood 1995 b; Marconato and Shapiro 1996; McAfee and Morgan 1996; van Rooij et al. 1996; Crossman et al. 2001; Sluka and Miller 2001; Williams and Polunin 2001; Mumby and Wabnitz 2002; Streelman et al. 2002; Gust 2002a; Gust et al. 2002b; Choat et al. 2003; Bay et al. 2004).

Gust's (2001) research on the spatial variability in demographic and life-history traits of scarids has shed some interesting light on growth plasticity. His research has found that scarids can display plasticity in somatic growth within small spatial scales. Such information, for example, is vital in the development of a regional perspective with regards to management. Hamilton (2004) has built on the biological knowledge-base of scarids within the Western Pacific by also examining the fisheries aspects of the Bumphead parrotfish in Solomon Islands.

These studies have highlighted interesting issues important in management considerations. The purpose of my work is to further contribute to this knowledge-base by undertaking a research exercise which will investigate the four highlighted biological and ecological issues related to coral reef fisheries in order to address the emerging issue of urbanised fisheries in Solomon Islands. The specific aims and objectives of this plan are highlighted below.

1.5 Aims

The intention of this PhD program was to document biological, ecological and fishery information related to several parrotfish species. Utilising analysed information from these sources, I address the four key management and conservation issues highlighted in this chapter. Because Gizo was chosen as a case-study urban centre, a substantial amount of ecological, biological, and ethnographic parrotfish data were extracted from the town and adjacent reefs. However, supplementary demographic and reproductive data were also sourced from other locations and also through collaborations in order to allow for better comparative analysis.

Three specific aims were drafted:

1) ***To evaluate the impacts of local small-scale fishing on parrotfish assemblages in Gizo, Solomon Islands.***

2) ***To compare the demographic and life-history characteristics of parrotfish species from Solomon Islands with other locations.***

3) ***To investigate the effectiveness of Marine Protected Areas (MPA) as a parrotfish conservation tool.***

The following chapters have been designed in sequence to achieve the above aims. Chapters Two and Three provide information pertinent to the first aim by conducting fishery and ecological surveys, respectively, while chapter Four and Five form the basis of discussion surrounding the second aim by providing demographic and life-history information, respectively. Chapter Six addressees the third aim, while chapter

Seven forms a synthesis of all the major findings. Specific aims and objectives are highlighted within each data chapter, followed by the relevant methodologies.

In **Chapter Two** a comprehensive investigation of an urbanised parrotfish fishery is presented in order to assess if artisanal fishing pressure does impact on harvesting parameters such as total landings and size distribution. This study gathered spearfishing-specific catch data from Gizo artisanal fishers over a two year period, supplemented by data from independent creel surveys. A fisher knowledge survey was also carried out, following ethnographic methodologies outlined in Sabetian (2002). The fisher knowledge information was used to acquire a historical perspective of the local urbanized fishery and discuss possible future trends.

In **Chapter Three** the density and distribution patterns of parrotfishes from Gizo's fishing grounds are compared over a two year period. In particular, the effects of localised fishing pressure on the size, density and composition of parrotfish species are presented. In this chapter, I determine if fishing pressure has induced a shift in scarid distribution and density, and if so, is it reflected by the catch data patterns portrayed in Chapter Two.

In Chapters Four and Five scarid demography and life-history are investigated in detail in order to compare the plasticity of these parameters from Solomon Islands to other locations. Firstly, age and growth rate analysis are carried out on selected population samples from the Indian, Caribbean and Pacific Ocean systems in chapter four. This chapter quantifies the difference in growth rates and life-spans between several excavating and scrapping scarids. I use comparative analysis to highlight plasticity in demographic parameters. Reproductive parameters of two parrotfish species are closely examined in chapter five in order to highlight interesting natural or induced traits.

Chapter Six assess the effectiveness of marine reserves as a conservation tool by presenting baseline herbivory and ecological data from a MPA in Solomon Islands. The role of marine protection as possible management or conservation tool is discussed.

In Chapter Seven the findings from each chapter are summarised and discussed in relation to the four issues highlighted in the beginning of this thesis. Furthermore, the contribution of this thesis to the understanding reef fish ecology is also discussed and future research priorities suggested.

Chapter Two

The Gizo parrotfish fishery

The findings of this chapter have been published in:

1) Sabetian, A. Foale, S., (2006). Evolution of the Artisanal fisher; case-studies from Solomon Islands and Papua New Guinea. SPC Traditional Marine Knowledge and Resource Management Bulletin. (20) 3-10. Mr Foale was not a collaborator in data collection for this exercise.

2) Aswani, S. Sabetian, A., (2010). Implications of Urbanization for Artisanal Parrotfish Fisheries in the Western Solomon Islands. Conservation Biology, 24(2) pp520-530. Mr Aswani was not a collaborator for data collection in this exercise.

2.1 Introduction

Fishing is a relatively inexpensive supplemental or part-time employment option in urban centres across Oceania. Laroche et al., (1997), made the observation that lack of transport infrastructure in developing countries often leads to the intensification of fishing effort close to urban areas where such services are available. I can confirm this from personal experience as a number of my own family members are involved in artisanal fishing operations in Solomon Islands, where they depend entirely on bi-weekly ferry transport from the capital Honiara to neighbouring Malaita Island.

Assessments of coral reef ecosystems is important to conservation initiatives and measures. However, Rhodes et al., (2008) highlight the point that while such information "provide general insight into the composition of the coral reef community, detailed investigations of the commercial or subsistence sectors of the coral reef fishery could also be used to improve fisheries management".

In this chapter I first begin by reviewing common trends from artisanal fishery case-studies across the Pacific, followed by a brief discussion on fishing gear selectivity, before outlining the aim and specific objectives of this chapter. This is the first study to provide a comprehensive analysis of an urbanised parrotfish artisanal fishery in western Solomon Islands. Mackay (2001) has described catch composition of small-scale inshore artisanal fisheries from Samoa. In Table 2.1 selected distribution of catches collected through long-term surveys from *Apia* (Samoa's capital) fish markets, and an independent creel survey conducted on the same markets, are presented along with a one week long student survey carried out from another region. Interestingly, the general trends in species distribution of the one week long survey appear similar to the distribution of the longer surveys. Of particular interest, are the relatively higher composition of acanthurids (Surgeon/Unicornfishes), lethrinids (Emperors), scarids (Parrotfishes), and to a lesser extent serranids (Groupers), holocentrids (Squirrel/Soldierfishes) and siganids (Rabbitfishes). In particular, the high relative percentage of acanthurids and acarids indicate the importance of these families to artisanal inshore fisheries in Samoa.

Family/Group	**Common name**	**% Distribution**		
		Student census	Creel census	Apia market
Acanthuridae	Surgeon/Unicornfishes	22.7	22.2	29
Carangidae	Jacks&Trevallys	3.3	1.0	3.9
Chaetodontidae	Butterfly fishes	2.4	2.0	
Holocentridae	Squirrel/Soldierfishes	7.9	10.2	3.3
Labridae	Wrasses	3.3	6.2	1.2
Lethrinidae	Emperors	11.3	11.6	11.8
Lutjanidae	Snappers	0.5	3.2	1.4
Mullidae	Goatfishes	3.2	2.4	2.9
Mugilidae	Mullets	1.5		10.9
Priacanthidae	Bigeyes	0.9	0.1	
Scaridae	Parrotfishes	9.1	12.4	24.8
Serranidae	Groupers	10.2	4.1	3.8
Siganidae	Rabbitfishes	3.6	7.5	1.0
Scombridae	Tunas&Mackerels		0.2	
Theraponidae	Grunters	0.7	0.4	

Table 2.1. Representative catch composition of the small-scale inshore fishery in Samoa. (source: Mackay 2001)

According to Munday (2000), artisanal catch composition of Papua New Guinea's coral reef fishery is affected, among other factors, by fishing gear selectivity. He adds that artisanal catches are generally dominated by Carangids (Jacks&Trevallies), Lutjanids (snappers), lethrinids, belonids (Needlefishes), acanthrids, siganids, serranids, scarids, and labrids (Wrasses). Table 2.2 presents historical artisanal data from four different locations around Papua New Guinea. The Port Moresby dataset provides a more complete transitional picture in terms of species composition. In particular, several interesting shift have taken place over the 11-year period. The percentage of Scombrids (Tunas and Mackerels) has significantly increased, indicating the relative importance of off-shore migratory species to artisanal fish markets in Papua New Guinea. The marked decline in Lethrinids, and to a lesser extent Scarids and Mullids (Goatfish), appear to suggest signs of overfishing.

Family	**Percent (%) composition of catch**							
	Port Moresby			*Huon Gulf*			*Tigak Is.*	*New Ireland*
	1986	**1986**	**1997**	**1992**	**1992**	**1997**	**1980-81**	**1994**
Scombridae	10.1	4	36.9			34.8	2.7	
Carangidae	8.3	7.8	11.6	23		19.4	14	5.6
Lutjanidae	4.7	5.1	10.1	19	23.24	18.1	13.3	18.3
Lethrinidae	29.3	31.8	6.8	6.4	11.13	4.4	10.4	4.9
Belonidae	5.2	5.4	6.4			5.3	<1	
Acanthuridae	6.8	7.3	5.7			0.9	4.7	
Siganidae	5.5	6	3.6			1.2	1.3	
Serranidae	2.5	2.7	2.7	5.1	8	4.0	9.1	4.6
Scaridae	5.2	5.6	1.5	1.9		<1	8.1	6.5
Chanidae	<1	<1	<1			<1	2.3	
Haemulidae	3.9	4.2	<1	2.5		<1	3.3	3.2
Mugilidae	4	4.4	<1			1.4	21.2	
Mullidae	4.5	4.8	<1	3.4		<1	<1	
Sphyraenidae			<1			7.2		
Hemiramphidae			<1			<1		

Table 2.2. Percent catch composition of some artisanal reef fin-fish fisheries in Papua New Guinea. (Sources: Lock 1986, Anas & Federizon 1997, Hermes & Anas 2000, Wright & Richards 1985; Cited in Munday 2000)

Anas (1998, cited in Munday 2000), conducted a month long daily survey of a Port Moresby fish market in Papua New Guinea. One hundred and sixty-nine species were identified during the survey. The dataset was dominated by Groupers, followed closely by Jacks and Trevallys, Snappers, Parrotfishes, and Surgeon and Unicornfishes. However, closer examination revealed a handful number of over-represented species within each group. Of equal interest and relevance were the representative lack of, or total absence of, large coral reef fishes. In particular, the fact that no Humphead wrass (*C.undulatus*) and Bumphead parrotfish (*B.muricatum*) were sighted during the month long survey, in combination with the fact the 16 grouper species sighted contributed less than 100kg combined, is an indication of overfishing in areas targeted by the artisanal fishers.

Other recent case-studies of small-scale fisheries in Papua New Guinea were carried out by Kaly and Preston (2006 and 2007). A closer inspection of these comprehensive surveys carried out in two very large urban centres, namely Lae (Morobe province) and Alotau (Milne Bay) in Papua New Guinea, reveal some interesting trends, both in terms of overall landings and also in relation to the artisanal component (i.e. fish for sale). These surveys, which were collected periodically for 12 months, are the best indication of respective fishing ground productivity from Milne Bay and Morobe province. Analysis of catch landing data shows that Trevallies, Snappers, Groupers, and Tunas/Mackerels, which make the bulk of landings at Lae and Alotau, were small overall with only 7% of catches greater than 30cm (FL) in Alotau and 8% greater than 50cm(FL) in Lae. Surprisingly, Surgeon/Unicornfishes were conspicuously absent from the Lae landings, while only making up a small percentage of landings at Alotau.

The maximum size range distribution of parrotfishes in Lae were small at approximately 35cm(FL), while Alotau exhibited relatively better maximum size range at near 50cm(FL). Given the larger coastal area of Milne Bay province and its much smaller population, we could be forgiven for thinking that the local fishery is fine. However, the lack of large Scarids (>50cmFL) such as the Bumphead parrotfish (*B.muricatum*), or the high profile Humphead wrass (*C.undulatus*), in combination with a fishery dominated by small Trevallies, Tunas and Mackerels appear to indicate heavy fishing pressure. The maximum size-structure of Trevallies and Tunas/Mackerels were inflated by small numbers of larger individuals, with overall average size of 18cm(FL) and 23cm(FL) recorded, respectively. In fact, only 4% of Trevallies and 33% of Tunas/Mackerels landed at Alotau were above 25cm(FL). The overall trend in Lae appears similar to the Alotau market, with conspicuous absence of important large functional species and small overall average sizes across prominent predatory species such as Trevallies and Groupers.

In terms of fish brought to the market for sale (i.e. artisanal component), the two markets were distinctly different in species composition. Cooked fish (smoked/dried) represented 43% of total number of fish on sale in Lae, while the rest were variously processed (fresh/gilled/gutted). River fish such as Tilapias, Archerfish, and Carps dominated species on offer for sale, followed closely by Tunas, Mackerels, and Mullets. In fact, the four main types of fish sold at the Lae maket were river fishes (most common), estuarine fishes, pelagic fishes, and reef fishes (least common). The average size of fish for sale was 20.5cm(FL), while approximately 50% of fishes ranged in size from 10-20 cm, with very few fishes recorded at over 40 cm. The Alotau market exhibited markedly different species distribution lead by small Trevallies, Tunas and Mackerels, and followed closely by coral reef fishes including Snappers, Groupers, Surgeonfishes and Emperors. About half (51%) of all fish offered for sale were cooked (smoked/dried). However, average size of fish for sale was similarly small, calculated at 27cm(FL), while 61% of all fishes ranged in size from 15-30 cm, with few over 55 cm.

The contrasting artisanal fisheries for both Lae and Alotau lead to following conclusion: landing figures of coral reef fish in Lae are markedly different to artisanal sale figures, indicating that a substantial component of catches are used for personal consumption. The reason for this may me complicated, but one possibility could be that given the low productivity of Morobe province's fishing grounds, artisanal fishers are resorting to utilizing their coral reef fish catches for sustenance, while targeting other common freshwater species for sale. This may be a indication of artisanal behavioural transition in areas suffering from long-term overfishing trends.

In Micronesia, Rhodes et al., (2008) recently described the local fishery of Pohnpei, the second most populous of four states in the Federated States of Micronesia. They have reported that the commercial coral reef fisheries of Pohnpei extract approximately 1,521 kg of reef fish daily, or approximately 500 MT year^{-1}, from 152 km2 of surrounding reef. In terms of the local artisanal fishery, fishers kept an estimated 14.8% (144.2 kg of fish day-1) of catch, in addition to the total volume sold, for home consumption.

Catch distribution revealed that Acanthurids accounted for 28.6 ± 1.7% of the total catch weight, followed closely by scarids at 16.4 ± 1.7% and serranids at 14.3% of the total catch. Serranids are subjected to ban months as part of management measures, and overtake scarids in catch distribution when the ban is removed, which according to Rhodes et al., (2008) demonstrates the importance of serranids to Pohnpei fisheries. However, they report that serranid catches are dominated by juveniles and small adults, an indication of high fishing pressure. They also describe other major families, in decreasing order of overall weight, volume, and weight contribution included the Lutjanidae, Carangidae and Lethrinidae. They reported that overall CPUE was 3.4 ± 0.1 (SE) kg hr^{-1}, with an average of 13.8 ± 0.4 fisher h trip-1 among all gear types.

Rhodes et al., (2008) make a similar observations to suspected signs of overfishing in Alotau and Lae (in Papua New Guinea) by drawing attention to three general trends in Pohnpei:

1) Anecdotal reports of fisheries-induced size and abundance reductions in large-bodied species such as the humphead wrasse, (*C.undulatus*), and bumphead parrotfish, (*B.muricatum*),
2) Negative impacts on spawning aggregations and migrations of siganids and serranids (Established by Rhodes and Sadovy 2002:cited in Rhodes et al., 2008), and
3) Accounts of increasing crown-of-thorns starfish outbreaks (*Acanthaster planci*)

They also mention that the sale of bumphead parrotfish is banned in Pohnpei, based on “large-scale reductions in size and abundance from historic levels”.

The urban fishery case-studies cited above share similar trends and characteristics. Firstly, in terms of species distribution, herbivorous species including acanthurids and scarids, and predatory species including carangids, serranids, and lutjanids contributed significantly to overall catch figures. Secondly, species dominating catch figures in all case-studies are characterised by small average size (<30FL). Thirdly, large coral reef species (>50cmFL), both predatory and herbivorous, are conspicuously absent from all datasets. Finally, all fisheries are characterised by overall small or declining size structures.

Any commercial activity, whether small or large-scale, is dictated by cost-effectiveness, and artisanal fisheries are no different. Urbanized fisheries, are naturally characterised by fishers who are looking to conduct their trade with minimum effort and maximum profit. As a result, the mode of fishing which will give them the most advantage is usually preferred. Fishing technique, in relation to gears used and the susceptibility of specific groups of fish to certain fishing methods, is an important contributing factor to overall fishing intensity.

2.2 Fishing gear selectivity

Most parrotfish species display schooling behaviour which makes them more vulnerable to spearfishing, especially while sleeping at night. Hamilton (2004), highlighted the schooling behaviour of bumphead parrotfish, *B. muricatum,* as one of the primary reason why this species has been overfished in Roviana lagoon of Western Solomon Islands. In fact, Dulvy and Polunin (2004) state that the bumphead parrotfish’s “combination of intrinsic vulnerability, relatively high catchability and high cultural (and/or monetary) value combined with evidence for decline and local extinction suggests that fishing may have resulted in global rarity of this functionally important fish species”.

Rhodes et al., (2008), report that in Pohnpei, where night spearfishing is the predominant fishing method, stocks of siganids (rabbitfishes) and mugilids (mullets) have been “dramatically reduced from historic levels by targeting spawning migrations outside existing MPA". They highlight the fact that fishers not only target spawning aggregations but also known migratory routes pre- and post-aggregation. It is worth noting that in Pohnpei, seasonal bans and marine protected areas have been established to protect "reproductively active serranids at and away from spawning sites". However, it is evident from the actions of the relevant Pohnpei authorities that their management initiatives, which are directed towards the commercially important grouper species, are bypassing other less economically important families.

Other fishing methods, such as dynamite and poison (e.g. Cyanide) fishing, are extremely indiscriminate and destructive (Bryant et al. 1999). Not only do these methods kill most living things in the vicinity of the blast or chemicals, they also destroy the reef's habitat structure, which ultimately impacts on recovery. That is not to say that netting and spearfishing do not also contribute to habitat destruction. From personal experience during many fishing expeditions on coral reefs I regard spearfishing as the more destructive of the two. This is primarily because of its non-selective nature, but also because of extensive coral damage from trampling and having to break corals in trying to retrieve speared fish wedged in coral branches and crevices.

Anas and Federizon (1997) investigated the vulnerability of specific families to different gear types in Papua New Guinea and found that "Handline catches were dominated by predatory species, typically from the families Serranidae, Lutjanidae, Lethrinidae and Carangidae, Nets and Spears characteristically caught species from families such as Siganidae, Belonidae and Scaridae, while troll catches were mostly reef associated pelagics".

Gillett and Moy (2006) cited information from Fijian fishers to state that spearfishing was primarily being used to target parrotfishes, unicornfishes, surgeonfishes, and goatfishes, while gillnetting was only occasionally used. Interestingly, other sources cited by Gillett and Moy stated that in other countries such as Solomon Islands, Tonga, Tuvalu and Samoa, netting is also hardly every used to target acanthurids or scarids.

Rhodes et al., (2008), conducting a similar analysis of the Pohnpei artisanal fishery, described that catches of 148 species were attributed to spears, followed by net at 73 species, and line fishing as responsible for 69 species. In terms of spear ownership in Samoa (Mackay 2001), nearly 80% of households questioned owned underwater spears, with nearly 50% also owning underwater torches for night spearfishing. Both Rhodes et al., (2008) and Gillette and Moy (2006) drew attention to the non-selective nature of spearfishing, and state that current levels of spear pressure are unsustainable across urban areas, reflected by large numbers of small and juvenile species.

The analysis of Catch Per Unit Effort (CPUE) data can also be used to estimate the effectiveness of particular fishing gears. The productivity of spearfishing has been confirmed from different regions in Papua New Guinea (Kailola 1995: cited in Munday 2000). In Pohnpei, Rohdes et al., (2008) estimated the CPUE of spearfishing was highest at 3.6 ± 0.1 (SE) kg hr^{-1}, followed by line fishing at 2.6 ± 0.1 (SE) kg hr^{-1}, and netting at 3.9 ± 0.3 (SE) kg hr^{-1}.

Having established a historical perspective of small-scale artisanal fisheries and analysed the trends of more recent case-studies in consideration of predominant gear selection, the importance of documenting and monitoring ecological and fishery parameters are clearly evident. Rhodes et al., (2008), state that in order to manage effectively, information on the demographic and life-history parameters of locally and regionally important coral reef fish species, must be complimented by data on the commercial or subsistence sectors of the coral reef fishery.

2.5 Objectives

In order to evaluate the impacts of local small-scale fishing on parrotfish assemblages in Gizo, a fishery investigation was conducted. This was achieved by collecting catch data from Gizo's artisanal fishers, and also from creel surveys carried out to independently assess the productivity of the same fishing grounds. In order to incorporate the fishers into this investigation, an interview survey was also conducted to assess their local ecological knowledge and discuss matters of concern to the Gizo fishery and possible emerging issues.

The specific objectives of this chapter were to:

1) Collect representative artisanal CPUE data from targeted fishing grounds.
2) Conduct an independent Creel survey on the same fishing grounds.
3) Carry out an ethnographic knowledge survey of artisanal fishers.

Since establishment of the multi-national peace keeping force RAMSI (Regional Assistance Mission to Solomon Islands) in 2003 to deal with widespread ethnic violence, and subsequent return of peace to Ghizo, increasing anecdotal information has been pointing to a gradual decline in parrotfish numbers and size in market landings. Unofficial reports from WWF staff suggest that this has been due to the increased urban population and spearfishing prevalence (discussed in fisher survey).

Utilising catch and creel information I will determine if fishing pressure has in fact resulted in localised shifts in parrotfish catch numbers over time, and if so, can that be corroborated with patterns portrayed by the independent creel survey conducted on the same fishing grounds.

2.6 Methods and Materials

2.6.1 Catch Per Unit Effort

CPUE information was collected from the reefs of Gizo in 2004 and 2005. For the purpose of identification the 'Gizo' reefs will from now refer to the open access areas immediately East and South-eastern side of Gizo township. Having previously conducted CPUE surveys in the same region (Sabetian 2002), I approached the same informants who had assisted me in the past. Collection was conducted in a similar manner with pre-prepared survey sheets, which in addition to a scale and several watches, were provided to the informants. In order to facilitate recording of catch information, the sheets were labelled with all essential information such as name, sex, time went fishing, time spent fishing, ecological zone, species caught and their frequency. The fishers simply filled in the necessary information next to the labels and I made regular visits to check on progress.

A number of issues were discussed, ranging from positive identification of fish species using a combination of English and local names, to specific fishing methods, and the materials needed for data collection. We decided that given the popularity of spearfishing by Gizo fishers, a representative sample was required. As such, catches of the spearfishing crew were recorded for me before being sold at the market. Thus the collected information only represents CPUE rates based on night spearfishing expeditions.

Considering that these fishers would be physically exhausted at the end of their expeditions, it was decided to minimise effort of recording CPUE. Therefore, the crew were instructed to group most fishes (excluding parrotfish) into families and weigh them as such, thus no species-specific composition rates are available for non-parrotfish families. Furthermore, the crew indicated that they would be unwilling to record individual sizes of fish. I assumed that this was due to their perception of the laborious nature of this exercise and decided against insisting in order to maintain a productive collaboration.

In relation to the targeted parrotfishes, we agreed on a list of six species which I was interested in, namely; *Hipposcarus longiceps, Scarus ghobban, Scarus rubroviolaceus, Cetoscarus bicolor, Bolbometopon muricatum, Chlorurus microrhinos*, but also left room for other species if they became prominent in total catch estimates. *Scarus oviceps* was pointed out by some fishers as one such species. The reason for choosing the above six species was primarily based on their status as the largest (Maximum TL>50cm; Allen and Steene 1994) of all Scarids.

2.6.2 Creel survey

In order to gauge the productivity of Gizo's fishing grounds, and also have the opportunity to validate the parrotfish CPUE data from informants, I led several expeditions to conduct independent creel surveys. The Gizo reefs were targeted using night spearfishing trips harvesting only parrotfishes. Species were identified (where possible), partitioned by colour phase, and weighed to the nearest kilogram. Comparative size structure analysis was also carried out for two species, namely *S.ghobban* and *H.longiceps,* where after being weighed to the nearest kilogram, the length of each fish was also measured and divided into five size categories (mm FL). All fishing trips were conducted at night with similar gear to those of our informants, using handheld or slingshot spears, and underwater torches.

2.6.3 Ethnographic survey

An interview format was adopted for the fisher survey. Between 2004 and 2006, fifteen "expert" fishers were asked a range of questions (Appendix 1). The purpose of this survey was to engage the fishers in dialogue that covered topics ranging from their skills and knowledge, their perception of resource use and management, and speculation of future trends. Seven questions were asked in a semi-informal structured interview format and elaborated on where necessary.

Questions:

1. How would you rate your knowledge of parrotfish behaviour, and ecology? Give examples.
2. What is the best method with which to target parrotfish? Why?
3. Are parrotfish popular with customers? Which species?
4. Which fishing grounds have you targeted over the past year?
5. How would you rate the parrotfish stocks around Gizo?
6. What do you think of past, current, and future fishing trends around Gizo?
7. Do you intend to continue fishing as a main source of income? What would you like to do or have to increase your fishing capacity?

2.7 Results

2.7.1 CPUE data from Gizo

The following dataset represents catch rates, composition and CPUE estimates from the Gizo reefs. Between July and September 2004, twelve expeditions, representing 110 man-hours of fishing were recorded. A further six expeditions were conducted between May and June 2005 representing 94 man-hours of fishing. The decision to stop at six expeditions was based on advise from the fishers that it was no longer economically feasible for them to target the Gizo reefs as they perceived productivity to have declined below acceptable fishing effort. However, the data collected does allow us to make tentative comparative analysis.

2004

Family	Common Name	Total weight (kg)	Percentage
Scaridae	Parrotfishes	184	26.78
Acanthuridae	Surgeon and Unicornfishes	87	12.66
Labridae	Wrasses	87	12.66
Lutjanidae	Snappers	62	9.0
Serranidae	Groupers	59	8.59
Balistidae	Triggerfishes	41	5.97
Carangidae	Travellys	33	4.80
Holocentridae	Soldier and Squirrelfishes	24	3.49
Siganidae	Rabbitfishes	20	2.91
Mullidae	Goatfishes	17	2.74
Lethrinidae	Emperors	17	2.74
Haemulidae	Sweetlip and Grunts	15	2.81
Others	Fish, Turtles, Clam shells	41	5.97
Total		**687**	**100.0**

Table 2.3. Total catch weight partitioned by family (nearest Kg) for Gizo in 2004.

Species	Common Name	Total weight (kg)	Percentage
S.ghobban	Bluebarred parrotfish	30.5	16.58
B.muricatum	Bumphead parrotfish	29.5	16.03
S.rubroviolaceus	Redlip parrotfish	21	11.41
S.oviceps	Dark Capped parrotfish	20.5	11.14
H.longiceps	Pacific Longnose parrotfish	20.0	10.87
C.microrhinos	Pacific steephead parrotfish	19.5	10.60
C.bicolor	Bicolor parrotfish	9.0	4.89
Assorted		34.0	18.48
Total		**184**	**100**

Table 2.4. Total catch weight for parrotfish species (nearest Kg) caught at Gizo in 2004.

Species	Common Name	CPUE gm/hr/fisher
S.ghobban	Bluebarred parrotfish	277
B.muricatum	Bumphead parrotfish	268
S.rubroviolaceus	Redlip parrotfish	191
S.oviceps	Dark Capped parrotfish	186
H.longiceps	Pacific Longnose parrotfish	182
C.microrhinos	Pacific steephead parrotfish	177
C.bicolor	Bicolor parrotfish	82
Assorted		309
Overall		**1673**

Table 2.5. CPUE values for parrotfish species caught at Gizo in 2004.

2005

Family	Common Name	Total weight (kg)	Percentage
Scaridae	Parrotfishes	150	31.91
Acanthuridae	Surgeon and Unicornfishes	61	12.98
Serranidae	Groupers	43	9.15
Carangidae	Travellys	34	7.23
Holocentridae	Soldier and Squirrelfishes	28	5.96
Haemulidae	Sweetlip and Grunts	27	5.74
Lethrinidae	Emperors	18	3.83
Siganidae	Rabbitfishes	15	3.19
Labridae	Wrasses	13	2.77
Lutjanidae	Snappers	13	2.77
Balistidae	Triggerfishes	10	2.13
Mullidae	Goatfishes	7	1.49
Others	Fish, Turtles, Clam shells	51	10.85
Total:		**470**	**100.0**

Table 2.6. Total catch weight partitioned by family (nearest Kg) for Gizo in 2005.

Species	Common Name	Total weight (kg)	Percentage
H.longiceps	Pacific Longnose parrotfish	40	26.67
S.oviceps	Dark Capped parrotfish	28	18.67
S.dimidiatus	Turquois-capped parrotfish	20	13.33
C.bleekeri	Bleeker's parrotfish	13	8.67
S.ghobban	Bluebarred parrotfish	10	6.67
S.rubroviolaceus	Redlip parrotfish	8	5.33
C.bicolor	Bicolor parrotfish	7	4.67
B.muricatum	Bumphead parrotfish	7	4.67
C.microrhinos	Pacific steephead parrotfish	5	3.33
Assorted		12	8.00
Total		**150**	**100**

Table 2.7. Total weight for parrotfish (nearest Kg) caught at Gizo in 2005.

Species	Common name	CPUE gm/hr/fisher
H.longiceps	Pacific Longnose parrotfish	426
S.oviceps	Dark Capped parrotfish	298
S.dimidiatus	Turquois-capped parrotfish	213
C.bleekeri	Bleeker's parrotfish	138
S.ghobban	Bluebarred parrotfish	106
S.rubroviolaceus	Redlip parrotfish	85
C.bicolor	Bicolor parrotfish	74
B.muricatum	Bumphead parrotfish	74
C.microrhinos	Pacific steephead parrotfish	53
Assorted		128
Overall		**1596**

Table 2.8. CPUE values for parrotfish species caught at Gizo in 2005.

2004 vs 2005

Common Name	% of Total catch 2004	% of Total Catch 2005
Emperors	2.74	3.83
Goatfishes	2.74	1.49
Groupers	8.59	9.15
Parrotfishes	26.78	31.91
Rabbitfishes	2.91	3.19
Snappers	9.0	2.77
Soldier and Squirrelfishes	3.49	5.96
Surgeon and Unicornfishes	12.66	12.98
Sweetlip and Grunts	2.81	5.74
Travellys	4.8	7.23
Triggerfishes	5.97	2.13
Wrasses	12.66	2.77

Table 2.9. Comparative estimates of total catch distribution between 2004 and 2005.

Species	CPUE gm/hr/fisher 2004	CPUE gm/hr/fisher 2005
H.longiceps	182	426
S.oviceps	186	298
S.dimidiatus	-	213
C.bleekeri	-	138
S.ghobban	277	106
S.rubroviolaceus	191	85
C.bicolor	82	74
B.muricatum	268	74
C.microrhinos	177	53
Assorted	309	128
Overall	**1673**	**1596**

Table 2.10. Comparative estimates of CPUE values for parrotfish between 2004 and 2005.

Catch data provided by the artisanal Gizo fishers depict an interesting scenario. Herbivorous fishes (scarids and acanthurids) appear prominent in total catch figures for both years indicating that spearfishing is most productive with these families. Parrotfishes, in particular, make up close to 1/3 of the total catch for both datasets. However, several other interesting transitions also take place. Most notable of which was the close to four-fold decline in Wrasses and sharp decline in Snapper numbers from 2004 to 2005. Total catch composition for parrotfishes remained steady increasing slightly from 26.78% in 2004 to 31.91% in 2005. However, detailed analysis of total weight partitioned into species reveals an interesting shift. In 2004, the larger scraping *B.ghobban* and *S.rubroviolacues*, and *B.muricatum,* the biggest of all excavating scarids, appeared prominent in terms of CPUE. In 2005, there was a complete transition with smaller scraping species dominating catch numbers and relegating the larger excavating scarids to the bottom half of total weight figures. Given that total catch composition for scarids remained high in 2005, we can draw the conclusion that the brunt of fishing pressure was borne by these smaller species.

Further comparative analysis of CPUE estimates (Table 2.10) clearly shows which species faced increasing fishing pressure. Most notable of all is the more than double increase in CPUE for *H.longiceps* from 186 gm/hr/fisher to 426 gm/hr/fisher, followed by *S.dimidiatus* and *C.bleeker,* both of which did not figure prominently at all in 2004 and were as such combined into the "assorted" group.

2.7.2 Creel survey

Between June and August 2004 five expeditions representing 64 man-hours of fishing were recorded, while between September and October 2005 six expeditions representing 70 hours of fishing was conducted. After bringing the catch back to base, it was possible to partition and weigh species by colour phase. If I was not able to, or unsure in identifying species, they were put in to the 'assorted' group.

2004

Species	Colour Phase	Total weight (kg)	Percentage
S.ghobban	TP	21	6.91
S.ghobban	IP	27	8.88
H.longiceps		44	14.47
S.oviceps	TP	27	8.88
S.oviceps	IP	7	2.30
S.rubroviolaceus	TP	18	5.92
S.rubroviolaceus	IP	12	3.95
C.bicolor	TP	13	4.28
C.bicolor	IP	14	4.61
C.bleekeri	TP	13	4.28
C.bleekeri	IP	7	2.30
C.microrhinos		17	5.59
S.dimidiatus	TP	11	3.62
S.dimidiatus	IP	2	0.66
B.muricatum		10	3.29
S.globiceps	TP	9	2.96
S.quoyi	TP	8	2.63
S.schlegeli	TP	6	1.97
Assorted		18	5.92
Total		**284**	**100**

Table 2.11. Total weight (nearest Kg) of parrotfishes speared at Gizo in 2004.

Species	Colour Phase	CPUE gm/hr/fisher
S.ghobban	TP	328
S.ghobban	IP	422
H.longiceps		688
S.oviceps	TP	422
S.oviceps	IP	109
S.rubroviolaceus	TP	281
S.rubroviolaceus	IP	188
C.bicolor	TP	203
C.bicolor	IP	219
C.bleekeri	TP	203
C.bleekeri	IP	109
C.microrhinos		266
S.dimidiatus	TP	172
S.dimidiatus	IP	31
B.muricatum		156
S.globiceps	TP	141
S.quoyi	TP	125
S.schlegeli	TP	94
Assorted		281

Table 2.12. Gizo CPUE estimates for parrotfishes in 2004.

2005

Species	Colour Phase	Total weight (kg)	Percentage
S.oviceps	TP	58	19.08
S.oviceps	IP	7	2.30
C.bleekeri	TP	21	6.91
C.bleekeri	IP	28	9.21
H.longiceps		37	12.17
S.dimidiatus	TP	22	7.24
S.dimidiatus	IP	1	0.33
S.globiceps	TP	21	6.91
S.ghobban	TP	8	2.63
S.ghobban	IP	11	3.62
S.schlegeli		17	5.59
S.rubroviolaceus	TP	3	0.99
S.rubroviolaceus	IP	13	4.28
C.bicolor	TP	7	2.30
C.bicolor	IP	5	1.64
S.quoyi	TP	12	3.95
C.microrhinos		6	1.97
Assorted		27	8.88
Total		**304**	**100**

Table 2.13. Total weight (nearest kg) for parrotfish species speared at Gizo in 2005.

Species	Colour Phase	CPUE gm/hr/fisher
S.oviceps	TP	829
S.oviceps	IP	100
C.bleekeri	TP	300
C.bleekeri	IP	400
H.longiceps		529
S.dimidiatus	TP	314
S.dimidiatus	IP	14
S.globiceps	TP	300
S.ghobban	TP	114
S.ghobban	IP	157
S.schlegeli		243
S.rubroviolaceus	TP	43
S.rubroviolaceus	IP	186
C.bicolor	TP	100
C.bicolor	IP	71
S.quoyi	TP	171
C.microrhinos		86
Assorted		386

Table 2.14. CPUE estimates for parrotfish species at Gizo in 2005.

2004 vs 2005

Species	% of Total catch 2004	% of Total Catch 2005
S.ghobban TP	**6.91**	**2.63**
S.ghobban IP	**8.88**	**3.62**
S.rubroviolaceus TP	**5.92**	**0.99**
S.rubroviolaceus IP	3.95	4.28
C.bicolor TP	4.28	2.30
C.bicolor IP	4.61	1.64
C.microrhinos	5.59	1.97
B.muricatum	3.29	0
C.bleekeri TP	4.28	6.91
C.bleekeri IP	2.30	9.21
S.oviceps TP	**8.88**	**19.08**
S.oviceps IP	2.30	2.30
H.longiceps	**14.47**	**12.17**
S.dimidiatus TP	3.62	7.24
S.dimidiatus IP	0.66	0.33
S.globiceps	**2.96**	**6.91**

Table 2.15. Comparative estimates of total catch distribution for parrotfish species from Gizo.

Species	**CPUE gm/hr/fisher 2004**	**CPUE gm/hr/fisher 2005**
S.ghobban TP	328	114
S.ghobban IP	422	157
H.longiceps	688	529
S.oviceps TP	422	829
S.oviceps IP	109	100
S.rubroviolaceus TP	281	43
S.rubroviolaceus IP	188	186
C.bicolor TP	203	100
C.bicolor IP	219	71
C.bleekeri TP	203	300
C.bleekeri IP	109	400
C.microrhinos	266	86
S.dimidiatus TP	172	314
S.dimidiatus IP	31	14
B.muricatum	156	0
S.globiceps	141	300
S.quoyi	125	171
S.schlegeli	94	243

Table 2.16. Comparative estimates of CPUE values for parrotfish species from Gizo.

Analysis of catch figures in 2004 revealed that *S.ghobban* and *H.longiceps* made up roughly 15% of total catch weight, respectively, followed by *S.oviceps* at approximately 11%. The bicolour parrotfish, *C.bicolour,* was ranked fifth at approximately 9% of the total catch, leading as the first excavating scarid followed by *C.microrhinos* and *B.muricatum* with 5.6% and 3.3%, respectively. The distribution pattern of the creel survey in 2004 revealed that a couple of larger scraping species made roughly 15% of the total catch component while fishing pressure was generally evenly shouldered by other species.

However, there was a marked shift in species distribution in 2005 with the smaller scraping *S.oviceps* leading the way with just over 21% of the total catch. This was followed by approximately 16% of the catch being attributed to *C.bleekeri* and *H.longiceps* at 12%. Although large scraping species did not appear prominent in catch figures in 2004, their figures for 2005 (*C.microrhinos* 1.97% & *C.bicolor* 3.94%) saw them relegated to the bottom of the list with *B.muricatum* not featuring at all. The gap left was filled by other smaller scrapers such as *S.dimidiatus*, *S.globiceps* and *S.schlegeli*.

Comparative analysis of the two datasets (Table 2.15) shows that catches of terminal phase *S.oviceps* more than doubled while initial phase *S.ghobban* and terminal phase *S.rubroviolaceus* markedly declined in 2005. Catch composition of *H.longiceps* remained high between both years. This is confirmed by comparative CPUE analysis (Table 2.16).

2.7.2.1 Fine-scale size analysis

Two species, *S.ghobban* and *H.longiceps,* were arbitrarily chosen for size structure analysis. Speared fish were identified and measured to the nearest mm (Fork Length) before being grouped into size categories and weighted to the nearest kilogram. Total catch (kg) declined across all size classes for *S.ghobban*, with a noticeable absence of larger individuals above 400mm (FL). The same scenario is repeated for *H.longiceps* with one notable exception. Total catch (kg) for the size class 200-290mm (FL) actually increased in 2005 indicating that a bigger percentage of smaller fishes filled the gap left by larger individuals.

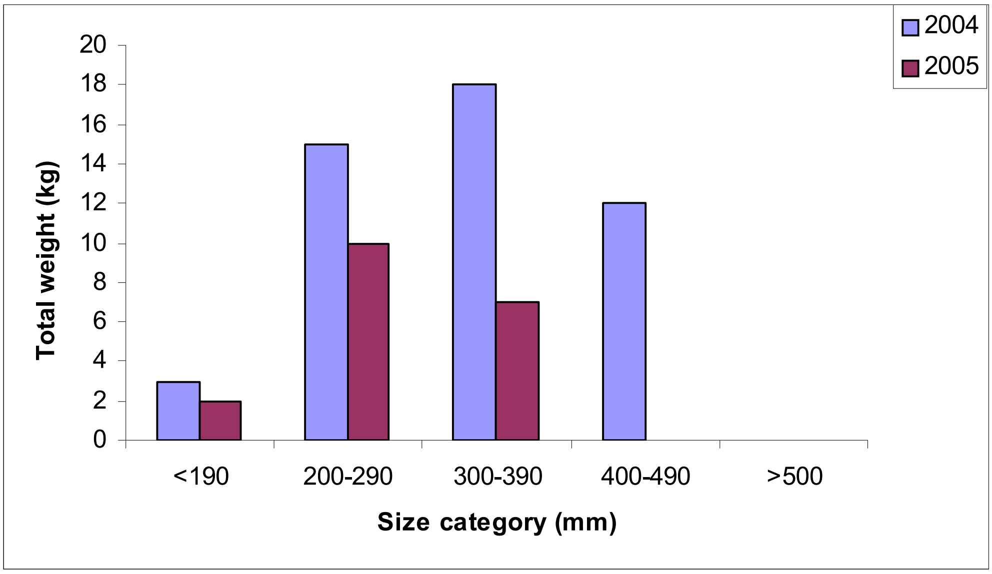

Figure 2.1. Size distribution estimates for *Scarus ghobban* from Gizo.

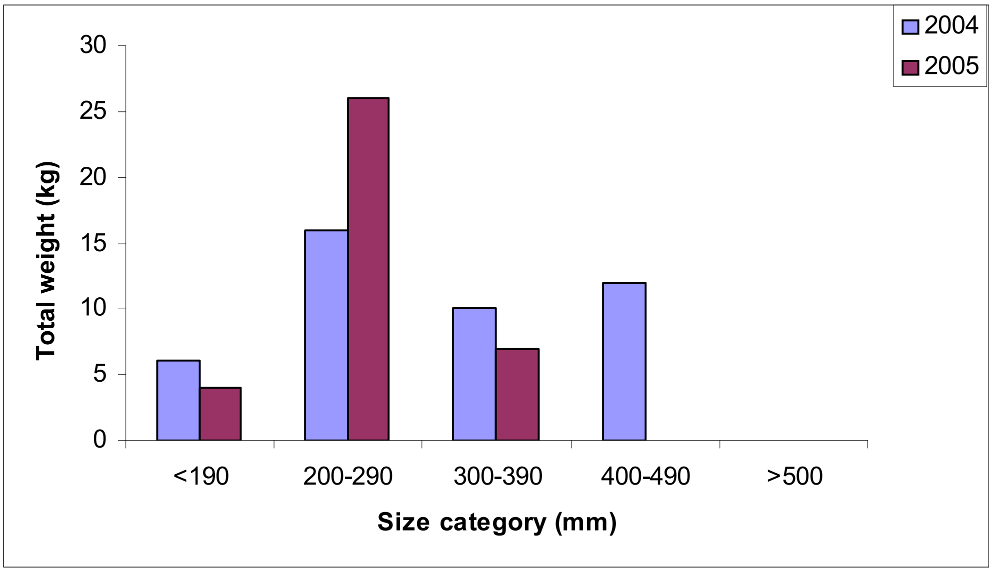

Figure 2.2. Size distribution estimates for *Hipposcarus longiceps* from Gizo.

2.7.3 Fisher survey

Seven questions were asked in a semi-informal manner and elaborated on if necessary (Appendix 1).

The first two questions dealt with ecological knowledge and most effective fishing methods. The fishers displayed detailed knowledge of parrotfish behaviour. All (n=15) interviewees made reference to the fact that parrotfishes are herbivorous and as such netting and spearfishing are the only techniques that can be used efficiently. They all also referred to seeing or having heard of parrotfishes feeding on potatoes, cassava, taro or other plant materials (Also see Hamilton 2004). This may be an indication of nutritional component in their diet, the significance of which could be investigated in the future.

Twelve fishers made specific reference to the fact that some parrotfishes ate corals while some appeared to scrape at different surfaces. Two fishers referred to having personally observed sleeping parrotfishes encased in a mucus bubble. Nine fishers (n=11) claimed that parrotfish actively feed just after dawn and just before dusk, while the remaining respondents said that parrotfish feed at all times of the day. Five fishers (n=5) claimed that rising tides appears to be the time when parrotfish are more actively feeding, the remaining where not sure (n=5 could not answer that question).There was unanimous agreement that parrotfishes are most easily targeted at night using spears as they generally aggregate in sleeping groups and are ‘deep sleepers (N=4)’. Netting was deemed the next most productive fishing methods with six fishers claiming that rising tides just before sunset or just after sunrise provided the best opportunity to net schooling feeding parrotfish.

Nine fishers made specific reference to seeing some parrotfish species display schooling behaviour while others appeared more solitary. The bumphead parrotfish (*B.muricatum*) and Pacific steephead parrotfish (*C.microrhinos*) were pointed out as schooling fish along with the Pacific Longlose parrotfish (*H.longiceps*) and the Dark Capped parrotfish (*S.oviceps*). The schooling behaviour of the above species was personally observed in the Gizo area, with *H.longiceps* and *S.oviceps* also appearing to sleep in aggregations within close proximity. For example nine fishers could recount specific encounters with large numbers of sleeping parrotfish wedged in the branches of coral trees. Nine interviewees made specific claims to knowing the spawning behaviour and periodicity of some parrotfish species. However, there was no consensus with three fishers claiming that all parrotfish species spawn together during the new moon period, while four fishers claimed that they have regularly observed spawning behaviour of parrotfishes at various lunar stages and times of day, and two observers claiming to have seen spawning aggregations of *C.microrhinos* (Pacific steephead parrotfish) at a particular location last for 3 or 4 days. Both these fishers stated that spawning aggregations seems to happen in sandy shallow areas. Again, no consensus was reached on spawning locations, as 13 fishers (n=13) could not associate specific habitat type with spawning. The fact that these fishers could not agree on specific spawning time and periodicity, may be an indication that parrotfishes in the Solomons are serial spawners.

Ten fishers rated parrotfish as very popular with customers. All fishers rated the Bumphead parrotfish as the most desirable and highly profitable species with locals and also commercial food outlets such as restaurants and resorts. One fisher claimed to have a contract with a local resort which required him to supply Bumpheads to the resort’s restaurant on a weekly basis.

Answers to the fourth question, which asked them to name targeted fishing grounds, were varied covering nearly all areas of the Western province from Gizo to the nearby Hapu Hapu reefs and the uninhabited areas of Rarumana island and north-new Georgia. Six fishers claimed to be no longer targeting the Gizo reefs due to low catch rates, while four indicated that they still accessed the Gizo reefs, but also targeted other fishing spots. All fishers indicated that the Gizo reefs were still popular with Ghizo inhabitants who did not have access to motorised boats (i.e. close proximity). Seven informants suggested that non-indigenous Gilbertese fishers were the most brazen at targeting customary owned fishing grounds as far as inner VonaVona lagoon and even marine protected sites such as designated reserves. Although anecdotal evidence appears to support this claim, it needs to be further investigated and validated.

The unanimous perception of the fifteen fishers was that large fish across all families have been steadily declining over the past decade. Nine fishers claimed that fishing effort has significantly increased in the past 3 to 4 years leading to noticeable declines across all targeted species on the Gizo reefs. The overall sentiment was that more people are now resorting to fishing as a way to generate or supplement incomes. When I asked individuals to elaborate on this issue the responses got more interesting. The ethnic Malaitan fishers blamed the civil unrest between 1999 and 2003. This event was responsible for significant numbers of Malaitans being intimidated and forced to keep a low profile or leave their settlement near Gizo. The Malaitan settlement on Ghizo is referred to as the ‘fishery’ or ‘fishing village’, because the majority of inhabitants are artisanal or small-scale commercial fishers and as such are a major driving force behind the local fishing industry. The Malaitan fishers claimed that the demographic nature of the fishery changed during the height of violence and lawlessness when some could no longer practice their trade. The gap left by the absence of some Malaitan fishers was largely filled by local Gilbertese fishers who met demand for fish through intensification of fishing effort on nearby reefs utilising night spearfishing. This method is most cost effective given the relatively short distance of the reefs from Gizo township and also the productiveness of targeting fish at night when they are sleeping and defenceless. Furthermore, night spear fishers have an added advantage of being the first sellers on the daily fish markets before the arrival of other fishers in the afternoon.

This added ethnic component to Gizo's artisanal fishery requires in-depth socio-cultural investigation, which is beyond the scope of this PhD. Both communities are recognized as the dominant artisanal or small-scale fishers. This observation is simply based on the number of fish stalls occupied by each ethnic group. In particular, the perception that as non-indigenous Solomon Islanders, Gilbertese fishers exploit marine resources without any regard for customary marine tenure needs to be investigated. An interesting anecdote told by most interviewees was that in order to avoid detection in customary areas at night, Gilbertese fishers always turn their torch off when at the surface and only turn it back on while under water.

The next question asked each fisher about their perception of current, past and future fishing trends. There was unanimous agreement that parrotfishes had become prominent in fish sales because of an increase in spearfishing, perhaps with the added fishing effort by the Gilbertese. Eight fishers claimed that large-sized parrotfish numbers had noticeably declined within the market in 2005, with five fishers claiming to have noticed absence of specific species such as terminal phase *S.ghobban* and *C.microrhinos*. Three fishers claimed of knowing crews of Gilbertese spear fishers who have spent 12 hours fishing the Gizo reefs. In local vernacular this is referred to as "Six to Six" (i.e 6pm to 6am) where crews of spear fishers visit numerous reefs starting from dusk and lasting through to the early hours of the morning. Their catch is brought to the market by other family members who take responsibility for selling fish while the fishers recover from their laborious ordeal. This was confirmed to me by several Gilbertese fishers.

When pressed on their perception of overall productivity of the Gizo reefs, the respondents unanimously agreed that fish numbers had declined across all families. However, only three fishers perceived this to be a problem as the others pointed to the fact that demand was being met by an expansion of fishing area on to more productive reefs. Four fishers claimed personal knowledge that fish stocks, parrotfish in particular, are still very healthy around the uninhabited areas of neighbouring Rarumana, but did concede that the most productive and cost-effective way to target the area was through mobilizing large crews (between 10-12 fishers) and utilizing nets as long at 200 meters. I personally observed the catch of a Rarumana netting expedition in October 2005 where over 200 *H.longiceps* and *S.ghobban* were extracted.

The final question pressed interviewees on their chosen profession. Twelve fishers indicated that they were happy with fishing as a source of income and intended to continue in the future. The remaining three respondents displayed reservations citing logistical restrictions. Interestingly, no one seemed to suggest that the apparent localised decline of fish stocks around Gizo would force an early cessation of their fishing activity. All fishers indicated a strong desire for improvements in fishing technology, access to fibreglass boats and bigger outboard motors. Nine fishers indicated their desire for independence conceding that their inability to acquire fishing gears (e.g. nets), boats and outboard motors on their own was the major reason why they still persisted with working in crews. One interesting revelation from six respondents was their desire to learn how to SCUBA dive and use air compressors. While others demonstrated reservations, these fishers indicated that the use of SCUBA would allow them to target known sleeping grounds of large fishes such as the Bumphead parrotfish and Humphead wrass, or be able to target more lucrative bech-de-mer species at deeper depths. This is an alarming development given the increasing use of SCUBA spearfishing in the Pacific (Page 1998; Wilkinson 2004; Gillett and Moy 2006).

2.8 Discussion

This study has described the first comprehensive investigation of an urbanised parrotfish artisanal fishery in Western Solomon Islands. In reviewing the major findings from this chapter we are able to see that CPUE data from Gizo confirms herbivorous fishes as important to the local artisanal fishery. On average, scarids made up roughly 30% of the total catch followed by acanthurids at 13%. In the Pohnpei artisanal fishery, Rhodes et al., (2008) reported that acanthurids made up, on average, 28% of the total catch, followed by scarids at 16%. However, they also point out that fishing pressure on parrotfishes increased to a maximum of 23.1% of total catch during the seasonal bans on groupers (discussed below).

Scarids and acanthurids have been shown to be especially vulnerable to spearfishing, making up the largest percentage of total Gizo CPUE catch (kg) in 2004 and 2005 (Table 2.3 & 2.6). This trend is in agreement with various previous studies (Gillett and Moy 2006, NOAA 2002b, Page 1998), especially when night spearfishing is the predominant method used (Rhodes et al., 2008). Distribution analysis of CPUE has shown a sharp decline in larger species (maximum TL>50cm) over the course of this study, highlighting smaller scraping species (maximum TL<50cm) as being more prominent in 2005 when compared to 2004. The trend indicates that fishers targeted the larger and thus more financially rewarding species first before turning their attention to the smaller individuals. This is an interesting transition which requires closer examination.

Analysis of creel data revealed that although the species-specific trend depicted in the CPUE data was not exactly mirrored, the overall transitional shift of larger to smaller scarids from 2004 to 2005 was the same (Table 2.15). Overall, both datasets show that smaller scrapping parrotfishes appeared to fill in the gap left by the decline of larger species such as *S.ghobban*, *S.rubroviolaceus*, *B.muricatum*, and *C.microrhinos*. Size distribution analysis for *S.ghobban* and *H.longiceps* from the creel data showed declines in larger size-classes from 2004 to 2005, which supports evidence of their decline in total catch figures (kg).

The notion that fishing pressure removes the biggest individuals first is not new. For example, in studying the effects of artisanal fishing pressure across a range of families in the Caribbean, Hawkins and Roberts (2004) found that smaller bodied species dominated fish assemblages in heavily fished areas. More specifically, in a study looking at the effects of fishing pressure on parrotfish density, Hawkins and Roberts (2003) found that as fishing intensified biomass of larger species declined and smaller species came to constitute a greater proportion of the total assemblage.

An example of transitional fisher behaviour was also reported by Rhodes et al., (2008) in Pohnpei, where the local fishery has been subjected to periodic fishing bans in order to protect reproductively active groupers, a commercially important family. However, this seasonal ban significantly increased fishing pressure on other families such as parrotfishes. Rhodes et al., (2008), analysed that during the serranid sales ban, “the per catch volume (mean ± SE kg) of scarids (t0.05 (2), 745, P = 0.009), lethrinids (t0.05 (2), 745, P = 0.000) and mullids (t0.05 (2), 745, P = 0.008) increased significantly, suggesting a shift in target species during the ban”. Their presumption that this transition maintains overall catch volume and, therefore income for artisanal fishers, can also be applied to the trend observed in the Gizo fishery.

CPUE of spearfishing estimated from data provided by the Gizo fishers was calculated at approximately 5.8 kg hr^{-1} in 2004 and 4.5 kg hr^{-1} in 2005. These estimate are larger than spearfishing CPUE in Pohnpei (Rohdes et al., 2008), estimated to be at 3.6 kg hr^{-1}, and also in Papua New Guinea as reported by Munday (2000).

In terms of parrotfish CPUE, scarids contributed on average 1.67 kg hr^{-1} or 29% of the hourly catch in 2004, and 1.596 kg hr^{-1} or 36% of the hourly catch in 2005. Although other families were also contributing to overall catch composition, the question is is this level of pressure on scarids sustainable? In relation to parrotfish productivity, creel data revealed that the Gizo reefs yielded approximately 4.44 kg hr^{-1} in 2004 and 4.34 kg hr^{-1} in 2005. In light of the species-specific transition discussed above, these similar estimates appear to reinforce the notion that if pressure in equally applied overall catch volume will remain the same (Rhodes et al., 2008).

Although Gizo reefs exhibit comparatively better spearfishing CPUE than other case-studies cited in this chapter, which may perhaps be a reflection of its smaller population and relative lower fishing pressure, the rapid transition with which parrotfish species changed in catch composition is a definite indication of their vulnerability to relatively lower levels of artisanal fishing pressure. In fact, the speed with which species can decline in abundance can be very rapid. For example, in the Philippines Russ and Alcala (1996) showed that sustained fishing pressure could reduce biomass very quickly. After 9 years of protection in the newly re-opened Sumilon reserve, it only took 1.5 years to severely decline fish stocks that had built up over those years.

The information gathered as part of the fisher knowledge survey allowed for an interesting insight in to the lives and perceptions of the fishers who are in part responsible for the apparent decline in local fish stocks. The fishers were able to demonstrate detailed ecological knowledge pertaining to effective harvesting methods. There was unanimous agreement that spearfishing at night was the only productive method with which to exploit parrotfish schooling and sleeping behaviour, and that it was ultimately responsible for their apparent decline around Gizo. Fishers were acutely aware of the overall declining catch rates from the Gizo reefs but did not perceive it to being a major problem. Fishers indicated that they were successfully applying their trade by expanding fishing effort on to other areas and thus being able to meet demand. However, there is no doubt that demand for fish is also a significant factor in the apparent increase in fishing effort, partly proven by the desire of nearly all respondent in acquiring and improving their technological and logistical capability. For example, using SCUBA for spearfishing is not a new concept, although it is strongly discouraged by all sensible divers. Fortunately access to SCUBA gear and training in the Western province of Solomon Islands is limited to a handful number of tourist resorts and dive operators. Their relative high cost also puts them outside the affordability range of most fishers. The growth of the tourism industry has been very limited in Solomon Islands, partly due to the recent civil unrest and violence, but primarily because of logistical constrains and lack of investment in the sector. This in turn has meant that unlike countries such as Fiji, Samoa, and Vanuatu, where tourism has seen a boom, the local Solomons economy is struggling and acquisition of expensive equipment is relatively harder. However, demonstrated interest by the artisanal fishers interviewed indicates that there is demand to be met and it will be just a matter of time before SCUBA spearfishing becomes increasingly prevalent in Solomon Islands. In fact, Bruno Manele from WWF Solomon Islands, mentioned that there have been some unconfirmed reports of deaths attributed to unlicensed use of SCUBA and air compressor equipment.

Solomon Islands still remains an area rich in marine biodiversity with a recent rapid marine survey declaring that Njari reef, on the western side of Ghizo island, as having one of the richest concentrations of reef fishes in the world (Sabetian and Afzal 2004). However, Njari island and its surrounding reefs are customary owned and as such enjoy some degree of protection. The scenario on the eastern side of the island is markedly different with a bustling township accommodating thousands of permanent and transient inhabitants. The adjacent open reefs thus provide access to subsistence and artisanal fishing, which still remains one of the few livelihood options. Analysis in this chapter appears to suggest that the Gizo reefs are suffering from a localised decline, most noticeably in important functional and predatory species. Widespread fisheries crashes are unlikely given the accessibility of fishers to more productive locations within the Western province (Sabetian and Foale 2006). However, ecological ramifications from localised removal of important herbivorous fishes is serious and can possibly lead to overall ecosystem changes (Dalzell et al. 1996; Hawkins and Roberts 2004; Sadovy 2005).

Pressure on Solomon Island reef fisheries has been low until very recently, and this can be attributed in the main to the unusually low human population density of the region, combined with the relatively weak penetration of Asian marine commodity markets (Sabetian and Foale 2006). However, economic aspirations are contributing to the expansion of urbanised areas such as Gizo. This, combined with a widespread lack of awareness of the limits of coastal fisheries, is contributing to the decline of local fish stocks. The expansion of more efficient fishing methods, such as SCUBA spearfishing, is an inevitable evolution of artisanal fisher behaviour in developing countries where economic realities are dictating changes in the utilisation of marine resources. This will inevitably increase pressure on fish stocks. In the absence of any form of routine stock assessment and management such developments do not bode well, and in the absence of widespread and systematic intervention, the only uncertainty at present is how long it will take for urban fisheries to collapse.

2.8.1 Management issues

In describing the status of inshore fisheries in Papua New Guinea, Kailola (1995:cited in Munday 2000) stated that “the size and composition of subsistence and artisinal fisheries catches varies greatly throughout PNG, however, the artisanal component increases markedly around coastal towns”. More importantly, Lock (1986: cited in Munday 2000) states that the stocks of predatory and large herbivorous reef fishes can be rapidly depleted even at low levels of fishing effort. Furthermore, associated problems of urbanization can exacerbate coral reef fisheries already under threat from over-fishing. For example, In Pohnpei, "overfishing, sedimentation, pollution, coral mining, dredging, deforestation and mangrove loss have all combined to negatively impact the already under threat coral reef ecosystem" (Rhodes et al., 2008).

However, the management of multi-species artisanal fisheries is not only complex from an ecological, socio-economic, and regional perspective, but also prohibitive given the lack of resources in many Pacific Island countries. Management initiatives focusing on spearfishing, highlighted as a major contributor of inshore over-fishing, have generally been met with mixed outcomes (Gillett and Moy 2006, Rhodes et al., 2008). In reviewing some of the major spearfishing management initiatives in the Pacific, Gillett and Moy (2006) have highlighted a number of general trends. For example, in relation to initiatives banning the use of spearfishing, and in particular night spearfishing, legislations at the national level have been ineffective, while at the village or community level success has been limited. Furthermore, they argue that management initiatives dealing specifically with spearfishing, in trying to deal with inshore overfishing, are unlikely to be effective given the multi-geared nature of Pacific Islands fisheries. They cite examples (e.g. Tonga) where attempts to control spearfishing effort in open-access situations have failed.

In relation to the use of SCUBA, Gillette and Moy highlight case-studies where simply banning their use nationally for spearfishing (e.g. American Samoa), has either failed because the fishers move on to other regions, or because of the difficulties of obtaining evidence for court prosecution. Rhodes et al., (2008) highlight the fact that national legislations banning the sale of specific species, in a multi-species fishery, will not work. They cite the Pohnpei example, where substantial volumes of herbivorous fish are removed from the reefs, "ironically" as a result of management attempts to control overfishing of serranids. Interestingly, they also make a specific observation with regards to spearfishing in Pohnpei, where its "traditional" status as a fishing method will make management measures difficult.

The use of more inclusive management measures such as large marine protected areas encompassing multiple essential habitat types needed for many species during ontogenetic development, feeding or reproductive stages, has been advocated by Rhodes et al (2008). However, the establishment of such large scale conservation and management initiatives are inherently difficult in cash strapped Pacific Island countries where multi-lateral collaborations are required between state, provincial and community leaders. Furthermore, MPA cannot protect coral reefs against anthropogenic activities associated with urban development such as changes in land use, pollution, and sewage (Mora, 2008)

However, given the open access and centralised nature of urbanised fisheries, market-based management and monitoring in terms of controlling fisher behaviour may yield more success. For example, Rhodes et al., (2008) point out that the enforcement potential of a commercial spearfishing ban in Pohnpei is high since "(1) markets are centralized, (2) marine resource enforcement agencies are proximate to markets, and (3) speared fish are readily identified".

Another interesting perspective is to consider the 'custodial link theory' proposed by authors such as Dalzell et al., (1996). Ironically, the lack of customary marine tenure and consequent development of open-access fisheries adjacent to urbanised regions, cited by Dalzell et al., (1996) as responsible for inshore overfishing, may also provide the opportunity to exercise centralized management in these areas. For example, Gillett and Moy (2006) allude to the fact that centralised management measure in some Caribbean countries may have been more successful given the lack of indigenous ownership and customary marine tenure.

Whatever the answer, the concerted and comprehensive management of urban artisanal fisheries is the most important priority in the Pacific Islands. This is because not only fishing is an important economic activity for many individuals looking to earn a living, but also because the nutritional security of the population is directly dependent on ecological integrity of the coral reef ecosystem.

Chapter Three

Effects of artisanal fishing pressure on parrotfish abundance and distribution: a temporal analysis from Gizo, Solomon Islands

The findings of this chapter have been published in: Aswani, S. Sabetian, A., (2010). Implications of Urbanization for Artisanal Parrotfish Fisheries in the Western Solomon Islands. Conservation Biology, 24(2) pp520-530. Mr Aswani was not a collaborator in data collection for this exercise.

3.0 Introduction

Understanding the relationship between abundance, harvesting pressure and natural changes in population size is important to “the reliable assessment and management of fished species” (Zeller and Russ, 2000). Knowledge on such issues allows fishery managers to exercise appropriate management measures in relation to ongoing and evolving fishing pressure. However this knowledge requires both catch surveys and fishery-independent assessments of abundance of target species and changes over time. For coral reef fisheries, where most species occur in shallow water amenable to visual census techniques, underwater visual census (UVC) is the best non-destructive and fishery-independent method, that can be used to estimate abundance, biomass, and length frequency distribution of coral reef fish assemblages across different spatio-temporal scales (Connell et al. 1998; Samoilys and Carlos 2000; Zeller and Russ 2000).

The relatively repeatability of visual surveys (Ackerman and Bellwood 2000) makes it a reliable method with respect to measuring density and distribution of non-cryptic, and reef associated species (Rogers et al. 1994; Connell et al. 1998; Zeller and Russ, 2000; Samoilys and Carlos, 2000). Given the heterogeneous nature of coral reef habitats, fish assemblages are seldom evenly distributed. In fact, based on individual ecological requirements, mobility, and trophic status, coral reef fishes are patchily distributed across different gradients. Presumably, the accuracy of an UVC will increase if specific taxa are the subjected of investigation (Floeter et al. 2004). In this study, parrotfishes were the focus of abundance estimates.

Strip-transect census, which uses a visual rectangular column to record fish numbers on a strip of the reef, is best suited for population density estimates (Rogers et al. 1994). Zeller and Russ (2000) further add to this discussion by highlighting the fact that the width of visual strip transects rather than length significantly affects the accuracy of density estimates. In fact, it has been suggested that the width of strip transects investigating the abundance of larger non-cryptic predatory or herbivorous species should range between 5 to 10 meters (Rogers et al. 1994; Connell et al. 1998; Zeller and Russ, 2000; Samoilys and Carlos, 2000). Although underwater visual census estimates may sometime be compromised by various factors, such as high mobility of certain species (Jennings and Polunin 1994; Jennings and Polunin 1995), they still remain the most appropriate method with which to assess relative changes in density and distribution over time (Connell et al. 1998).

3.1 Objectives

Having satisfied the fishery aspects of aim one in the previous chapter, its ecological aspects are pursued here. Parrotfish density and size-distribution estimates from the same region were acquired via an UVC exercise in order to assess the relative accuracy of CPUE and creel data trends via triangulation. In particular, the following objectives were pursued:

1. Estimate temporal variability in parrotfish density and size-structure from the Gizo reefs.
2. Conduct comparative analysis of parrotfish mean-size (TL) over time.
3. Conduct a ‘Power Analysis’ in order to gauge the relative strength of density and distribution estimates.

Fishing pressure is one if the major determinants of reef fish assemblages (Tuya et al. 2006). However, even though an increasing number of studies are showing adverse effects arising from artisanal fishing pressure (Campredon and Cuq 2001; Ruttenberg 2001; Hawkins and Roberts 2004; Tuya et al. 2006), estimates of localised reef fish abundance and distribution remain limited. As discussed in chapter One, the image of artisanal fisheries as benign small-scale fisheries has done little to garner support or focus for large-scale ecological research. In fact, Hawkins and Roberts (2004) state that artisanal fisheries are still perceived to have negligible impact on standing stocks of coral reef fish.

In chapter Two I described the parrotfish fishery of Gizo, concluding that parrotfish catches appeared to had declined over time. In this chapter, I attempt to ascertain if parrotfish stocks within the same fishing grounds have also declined, which would verify and compliment the findings of Chapter Two. I begin by reviewing recent artisanal fishery trends from several case-studies, and also discuss resource exploitation trends and ecological ramifications.

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3.2 The impact of artisanal fishing pressure on fish assemblages; recent trends

Artisanal fishing is often characterised as open-access fisheries (Sabetian and Foale 2006; Adams and Delzell 1994) where labour is the predominant input (Liese et al. 2003). Liese et al., (2003) explored the potential conservation implications of artisanal fisheries in Indonesia by combining a model of open access fisheries exploitation with a distance-based approach. They highlight the fact that open access resources in developing countries are often more degraded close to densely populated areas and healthier in remote locations. Using CPUE and other ethnographic information from coastal fishing villages in Eastern Indonesia, they concluded that “fishing effort decreases in distance from product and labour markets if the product price decays more rapidly over space than does the opportunity cost of fishermen's time". This reinforces the notion that artisanal fishing is more cost-effective and thus more intensive closer to populated regions.

In the recently occupied (<50 years) Galapagos Islands, artisanal fishing pressure has been shown to be affecting exploited marine communities (Ruttenberg 2001). Ruttenberg, for example, cites other historical studies which have indicated that the total abundance and average size of the Sailfin grouper (*Mycteroperca olfax)*, a regionally native species that was highly valuable and exploited in local artisanal fishery, has significantly declined. Where in the 1980's *M.olfax* comprised over 40% of total artisanal catches, by the late 90's catch figures had declined to below 20%. In quantifying the affects of artisanal fishing pressure on commercially targeted species, Ruttenburg (2001) came to the conclusion that because "community structure, abundance and biomass of the primary target species were significantly lower in the heavily fished sites than in the lightly fished sites" (see Figure 3.1), "artisanal fishing has both direct and cascading effects throughout the community".

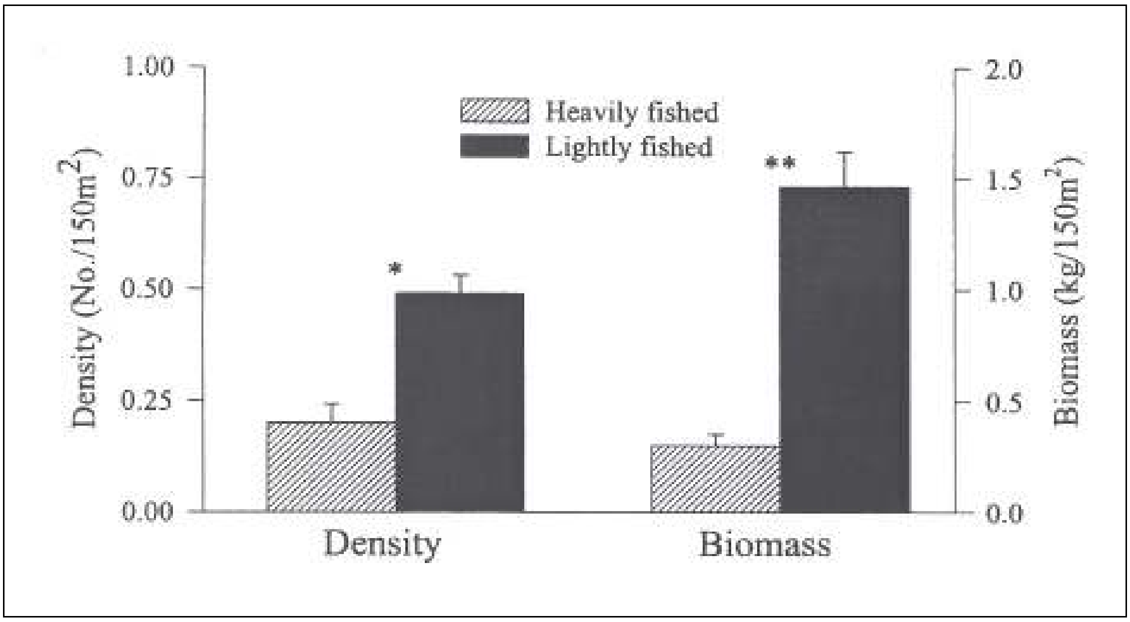

Figure 3.1. Density and biomass of commercial species (mean ± 1 SE) in the Galapagos Islands . * indicates significance at the 0.01 level, ** indicates significance at the 0.005 level. (Source: Ruttenberg 2001).

Of particular interest in this study was the fact that lower abundance of herbivorous fishes allowed for high numbers of other herbivores such as sea urchins (see Figure 3.2). Although the high variability in density estimates meant that fishing effort difference was not statistically significant, Ruttenburg (2001) cites the selectivity of fishing gears (handline and spear) as a possible causative factor for this trend. In fact, he highlights the two-fold impacts of sea urchin predator reduction and herbivorous fish reduction as responsible for relative high sea urchin numbers in heavily fished areas. Not only sea urchins were being subjected to low predation through the decline of several possible predators such as *Bodianus diplotaenia* and *Arothron meleagris*, they were also out-competing less abundant herbivorous grazers such as scarids and acanthurids,.

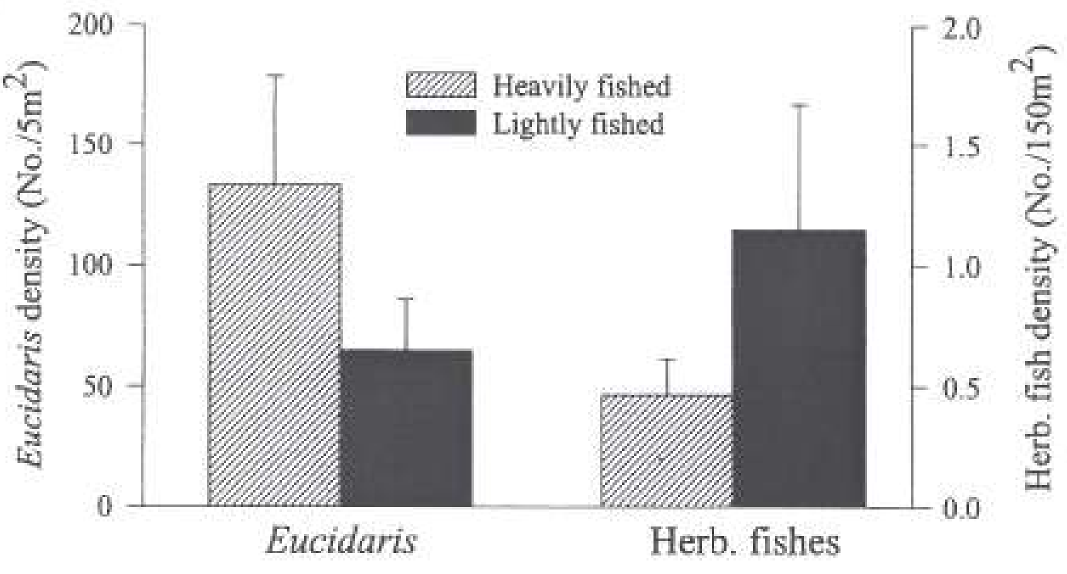

Figure 3.2. Densities of sea urchin *Eucidaris thouarsii* and a guild of herbivorous fishes in heavily and lightly fished sites (mean ± 1 SE). (Source: Ruttenberg 2001).

In 2003 the Atlantic and Gulf Rapid Reef Assessment (AGRRA) Program was published. This report tabled a comprehensive assessment of marine resources within this very large region. I have decided to cite two individuals reports from the Netherlands Antilles and the Bahamas, which describe coral reef fish status from respective artisanal fisheries. My interest in these particular reports was primarily based on the fact that these two locations have markedly contrasting human populations, as and such should exhibit different trends in intact fish assemblages.

Curacao (Netherlands Antilles), is a small oceanic Island (61km long, 443 km^2 area) populated by 155,000 people (Bruckner and Bruckner 2003). Bruckner and Brucker (2003), cite a number of sources to describe the local reef fish communities as overfished due to heavy artisanal fishing pressure from spearguns, fish traps, and gill nets. In quantifying the stock status of targeted fish species on the Island, they report that even the remote Eastern and Western parts of the Island are characterised by low abundance, low size, and low diversity of large-bodied species, with many commercially important food species being absent. They also reported that families often contained only a few species previously reported in Curacao, with average densities being less than 8 individuals/100m^2 (see Figure 3.3) and their mean size between 15-25cm. Bruckner and Bruckner (2003) attribute this trend to the “cumulative impacts of heavy fishing pressure, lack of enforcement of spearfishing ban, along with increased urbanization and degradation of coastal nursery areas”.

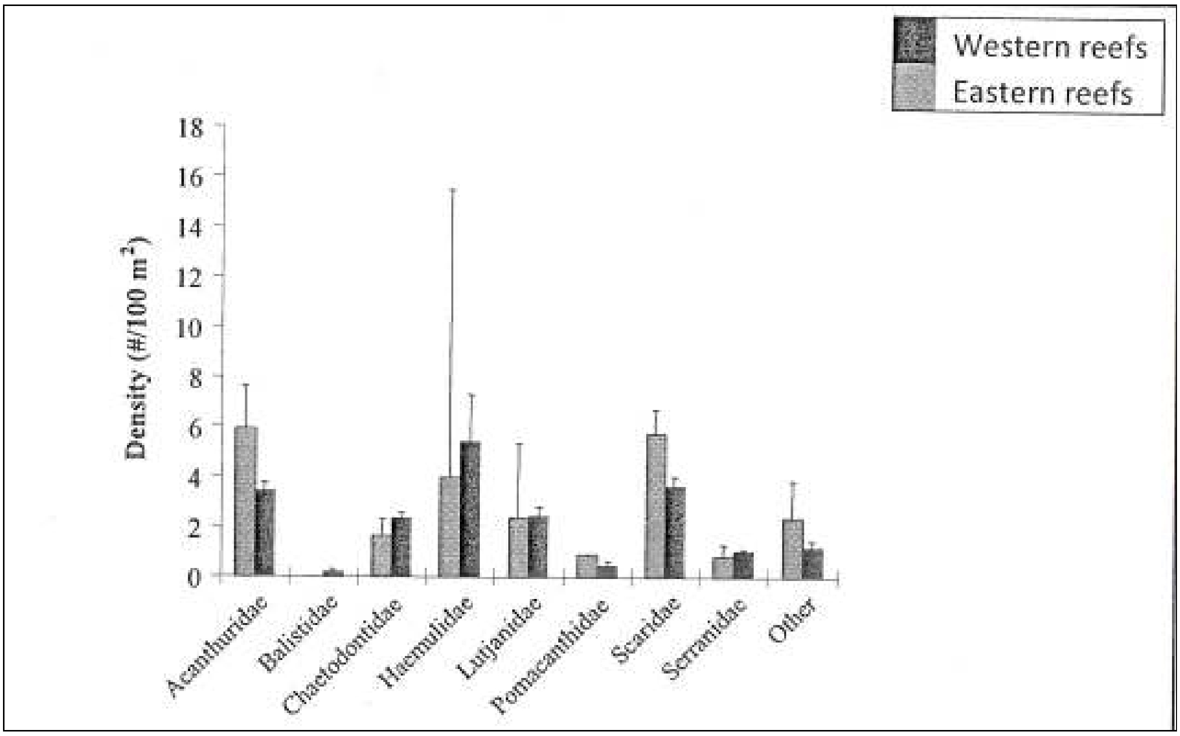

Figure 3.3. Mean fish abundance (no. individuals/100m^2 ± SD) for Eastern and Western Curacao. (source: Bruckner and Bruckner 2003)

Detailed size-structure analysis of important carnivores (snappers and select groupers) and key herbivores (parrotfish, surgeonfish, and yellowtail damselfish) confirmed the poor status of these species (Bruckner and Bruckner 2003) (Figure 3.4). Although parrotfishes were reported as one of the more abundant families on the reefs, figure 3.4 confirms their relatively small size.

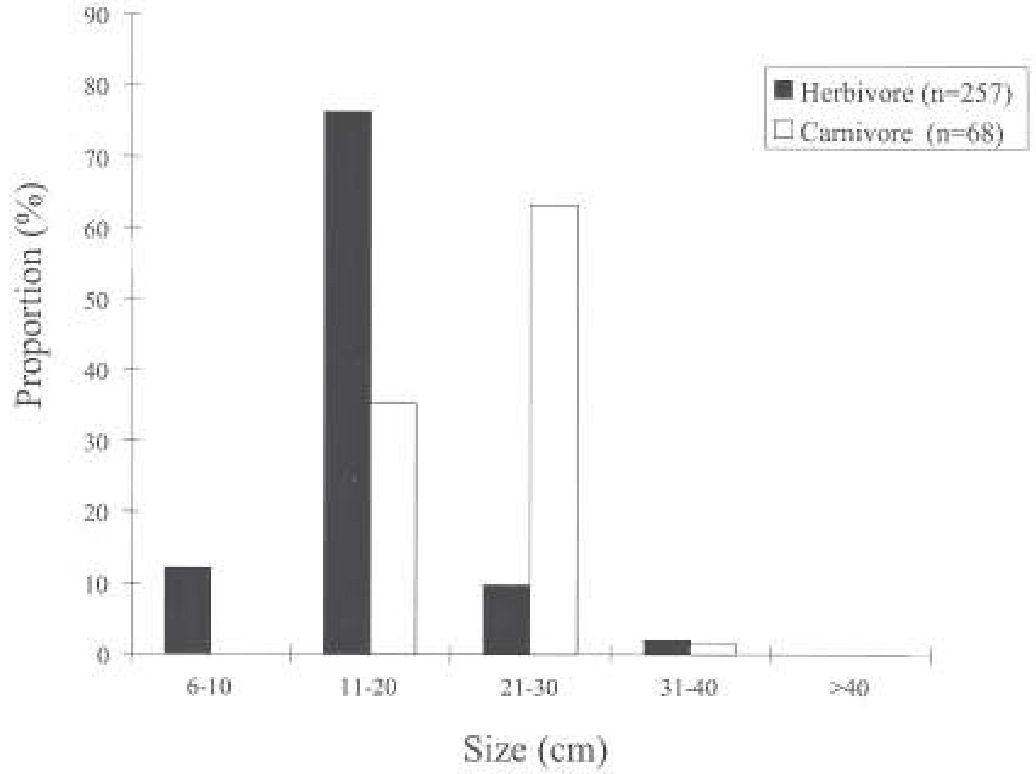

Figure 3.4. Size frequency distribution of herbivores and carnivores in Eastern and Western Curacao. (source: Bruckner and Bruckner 2003)

The second case-study from the AGRRA report is on Andros Island, the largest island of the Bahamas, which is actually composed of three major islands; North Andros, Mangrove Cay, and South Andros. Its total landmass equates to an area of about 6,000 km^2, nearly 170 km long and 70 km wide at its widest point. It has the world's third largest barrier reef, which is over 220 km long. In contrast to Curacao’s 155,000 population, Andros Island has a combined population of 6,000 people, the fewest people per unit area of all of the Bahamas. Kramer et al., (2003) surveyed coral reef fish assemblages along Andros’ eastern side, where the majority of the population lives. They describe the overall status the reef fishery as comparatively better than other regions in the Bahamas, dominated by scarids, haemulids, and acanthurids. However, they point out that the dominance of these species is somewhat misleading given that fish such as angelfish, parrotfish and surgeonfish are not yet targeted heavily by the artisanal fishery.

In fact, species richness within specific families are low, with parrotfish, for example, being only limited to ten species. Although a total of 164 species were identified during the entire survey (200 species is usually considered a benchmark for an excellent fish count), select species density only averaged 37.4 individuals per100m^2 belt transects. Serranids, for example, although ubiquitous, were present in very low numbers (<0.5/100m^2). One major causative factor was the significant concentrated fishing effort on grouper spawning aggregations close to the Island. In fact, Kramer et al., (2003) point to the lack of many commercially important grouper and snapper species as proof that artisanal dominated fishing pressure may be unsustainable on the Island. Unfortunately, they also point out that fishing pressure is difficult to quantify given the lack of statistics from landing sites.

Kramer et al., (2003) make a key observation; “Given Andros’s low population base, its relatively modest fishing pressures, and the presence and extent of well-developed reef-crest and fore-reef habitats, a greater abundance of fishes was expected”. In contrast, they came to the conclusion that given the relative low abundance of fishes, and the growing demand for commercially important species, Andros’s local fish population are likely to be susceptible to even “modest increases in fishing”. This observation is extremely relevant to the Gizo case-study pursued in this thesis, given the fact that Gizo shares similar characteristics with Andros in terms of relatively low population-base and large fishing grounds.

In another case-study, Hawkins and Roberts (2004) studied the effects of artisanal fishing pressure across six different islands in Caribbean region and discovered that intensive pressure had transformed some Caribbean reefs “in ways that seriously compromise their ecological and economic value”. In particular, the abundance of fish populations among heavily fished regions showed significant declines, with “groupers, snappers, parrotfish and surgeonfish showing orders-of-magnitude differences in biomass among Islands”. Hawkins and Roberts (2004) found support for previously cited studies which stated that heavily fished regions are dominated by small-size individuals. They found that in all families, smaller-bodied species dominated fish assemblages in heavily fished islands.

They also found support for the notion that "most vulnerable families and species will be affected at lower fishing intensities than the more resilient". In this case, they ascertained that, at the family level, groupers, snappers, and surgeonfishes, dramatically declined in biomass with increased fishing pressure. In fact, Hawkins and Roberts (2004), support the view that artisanal fishing selectively remove larger-bodied species (see also Page 1998). They points to the lack of large groupers and parrotfishes in heavily fished regions as proof of this. The susceptibility of parrotfish and surgeonfish, as major grazers, showed order-of-magnitude declines, suggesting that artisanal fishing is also seriously reducing levels of herbivory on the studied Islands (Hawkins and Roberts 2004). This was in turn responsible for decline in coral cover, structural complexity, and increased levels of algal cover in heavily fished regions. The above findings led these authors to suggest that fishing pressure on herbivorous species had a much more serious cascading effect on coral reefs.

Tuya et al., (2006) investigated the effects of fishing pressure on abundance and size-distribution of the parrotfish *Scarus cretense* across several Islands of the Canarian Archipelago, the fishery of which is multi-species and entirely artisanal. They made similar observation to Kramer et al., (2003) by highlighting the lack of landing statistics as hampering the monitoring of artisanal fisher behaviour within the Archipelago. Through their own investigation, Tuya et al., (2006) found lower abundance and biomass of parrotfish around the more populated and heavily fished Islands. In particular, there were a lack of parrotfishes above commercial-size range (>20cm) across the entire study area, with large-bodied individuals (>35cm) only being observed at only two locations, where fishing pressure was low and fishing activity was subjected to restrictions. Overall, they estimated mean density of *S.cretense* as $6.79\underline{+}0.46$ individuals per $100m^2$ transect (mean$\underline{+}$SE), identifying clear difference in mean abundance according to fishing intensity across different locations. They also reflected on the disappointing abundance and biomass estimates of *S.cretense* within the archipelago, by stating that this trend appears to support historical studies which have also concluded that the region has been facing "overexploitation and severe depletion of fish populations for the last few decades". Their observation that "the effects of recreational hook and line, and spearfishing, are being overlooked by local authorities" gives us an insight into the problems being faced in the Canarian Archipelago.

What the above case-studies have demonstrated is the common theme that fishing pressure induces direct response from fish assemblages in terms of reduced abundance and size structure. What is also evident is the fact that coral reef fish populations are susceptible to even modest levels of overfishing. This was clearly demonstrated by the aforementioned comparisons of the heavily populated Curacao Island versus the sparsely populated Andros Island. Given this fact, the management of artisanal fisheries are overwhelmingly complex, especially given the socio-economic repercussions involved in enforcing restrictive control measures. Although Liese et al., (2003) highlight the open access nature of artisanal fisheries as exacerbating this problem, they point to the development of alternative employment opportunities as a possible alleviating measure to reduce the role of fishing as the primary employment opportunity in heavily populated coastal towns. This approach, which was advocated to me in personal communications with the late Robert Johannes (a prominent ecological and ethnographic researcher in the Pacific) is increasingly being recognised by funding agencies such as the European Union who are investing millions of dollars into viable employment opportunities in overpopulated urban regions (Solomon Star 04.06.10) .

Because intact fish assemblages are critical to the functioning of coral reef ecosystems, patterns in their diversity, density, and size-structure can be important indicators of ecological and anthropogenic processes (Kramer et al. 2003). The lack of artisanal catch statistics has already been highlighted (Tuya et al. 2006; Kramer et al. 2003) as the main factor hampering the quantifying of fishing pressure. This issue has already been addressed in the previous chapter with regards in the Gizo artisanal fishery. In this chapter, I embarked on an ecological investigation in order to ascertain if the quantified fishing pressure is being reflected in (a) decrease population densities of parrotfish, (b) shifts in size-structure distribution, and (c) reduce average size.

3.3 Methods and Materials

3.3.1 Site description

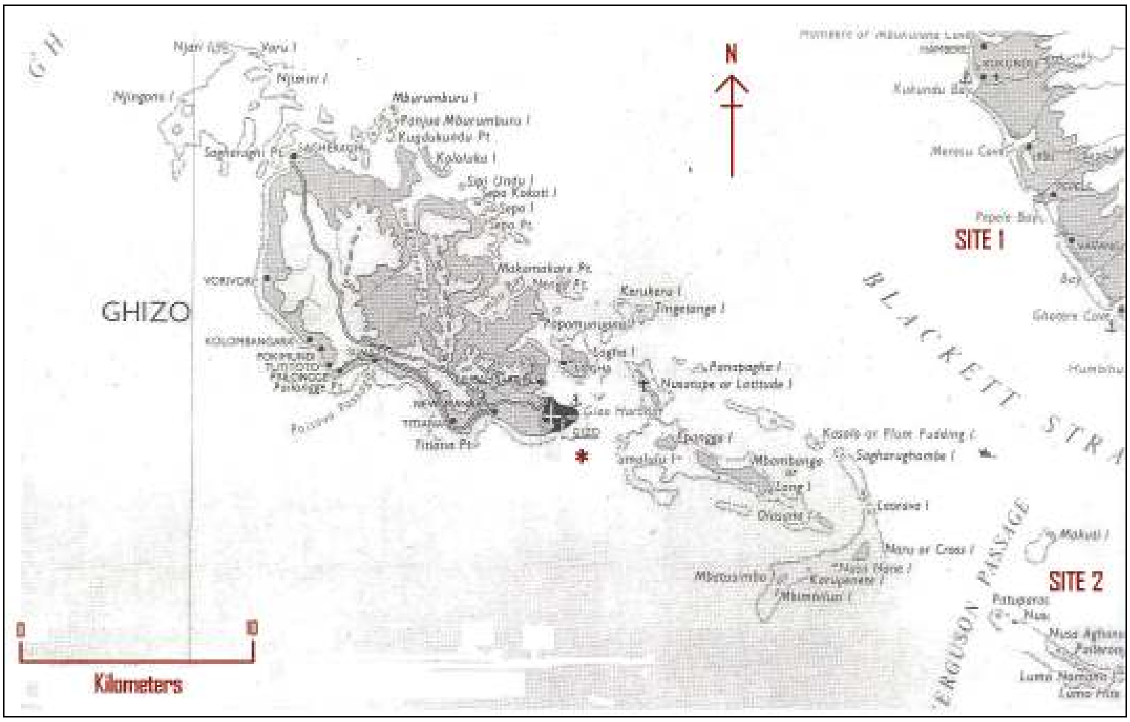

Figure 3.5. A detailed map of Ghizo island showing the locations of Gizo Town (*), Vavanga (SITE 1), and Nusa Aghana (SITE 2).

For the purpose of identification the ‘Gizo’ reefs refer to the open access reefs immediately East and South-eastern side of Gizo township (* Figure 3.5). Specifically, Nusa Aghana (SITE 2, Figure 3.5.), an uninhabited small raised coraline Island, was chosen as the focal point for the UVC exercise. The reefs are hereafter referred to as the ‘Gizo’ reefs. The Hapu Hapu reefs, to which Nusa Aghana and other neighbouring standalone raised coral islands belong, stretch for several kilometres after FERGUSON PASSAGE (or more, based on which customary border is adopted). This area is commonly accessed by artisanal and subsistence fishers and as such is described as belonging to the open-access Gizo group.

The extensive adjacent fringing reef morphology of Nusa Aghana is represented by hermatypic corals which descend down to approximately 40 meters across the entire reef face. However, rugosity, as in indication of structural complexity over a smaller spatial scale (Wilson et al. 2008), appeared to be higher on the seaward (western) side of the Aghana reefs. As a result, the sheltered (eastern) side of the Aghana reefs were subjected to the UVC. The reefs of Vavanga (SITE 1, Figure 3.5) on nearby Island of Kolombangara were also subjected to an UVC exercise for the purposes of power analysis (Discussed in Section 3.3).

3.3.2 Sampling design and procedures

Scarids are ideal candidates for UVC, not only because they are ecologically tied with coral reefs (Choat and Bellwood 1991) and as such are one of the most abundant and widespread families on the reef (Gust et al. 2001), but also because they are easier to identify given their less reactionary response to the presence of SCUBA divers (personal observation). A large scale visual strip transect census was used to quantify density and size distribution of three excavating (*B.muricatum, C.microrhinos, C.bicolor*) and three scrapping (*S.ghobban, S.rubroviolaceus, and H.longiceps*) parrotfishes between 2004 and 2005. The decision to choose these species was primarily based on the fact that they are the six largest species (max TL >50cm, Allen and Steene 1994) within the parrotfish family and as such could be more easily identified.

Between July and August 2004 forty-seven transects were carried out on the Gizo reefs. The same region was again the subject of forty-one transects between May and June 2005. The sampling design focused on establishing accurate density and size-distribution estimates over time. Using SCUBA, transects of dimensions 100 x 10 m^2 were employed at 15m of depth (Figure 3.6). Density estimates, represented by number of individuals per transect, were decided upon instead of abundance estimates represented by biomass per unit area. The reason for this was two-fold; firstly for comparative purposes and to keep in line with above cited studies on parrotfish density, and secondly because numbers of fish per transect is easier to visualise compared to kilograms per hectare.

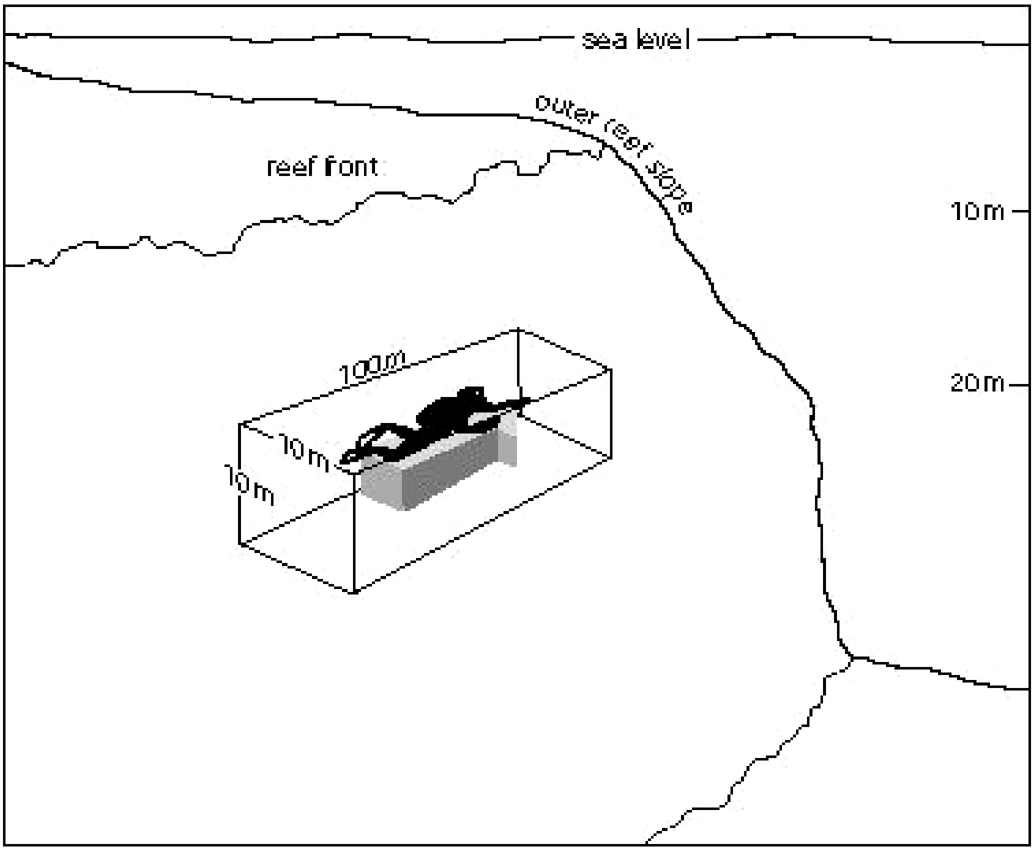

Figure 3.6. Visual depiction of the strip transect column.

The decision to carry out 100 m^2 replicated visual censuses was primarily based on the fact that given the approximately 2km length of the survey sites (Both Site 1 & 2), 100 meters would allow sufficient area of the reef face to be monitored, which upon replication would accurately represent parrotfish density status across those specific reefs. This decision is supported by other parrotfish studies (e.g. Tuya et al. 2006; and others cited within) that have also employed 100 m^2 replicated visual censuses to estimate density and size structure.

Within each transect, the species, frequency, Total Length (TL) and colour phase (where possible) of the six targeted scarids were recorded on underwater paper. The use of a tape measure or a transect line was deemed too time-consuming and disruptive. Instead a timed swim exercise was employed prior to the commencement of fieldwork. Using GPS coordinates, twenty-one pre-measured 100m transects were replicated at 15 meters of depth during which the size and frequency of the six selected species were observed. At the completion of this exercise I was confident that a 5-minute swim accurately represented a 100 meter transect and also give enough time for observation (Appendix 3).

Although other studies still employ fish models (e.g. wooden parrotfish models in Gust 2000) to achieve consistency and accuracy in observer estimation, my estimation of fish length was calibrated according to Sabetian (2003), with slight modification (Appendix 2). Briefly, instead of estimating and then measuring coral heads to achieve an acceptable error in estimated and actual size, the use of live fish were employed for this exercise. Slow moving, feeding, or stationary parrotfish were observed underwater and their TL estimated. After this, I approached the location and measured the length of the identified background substrate. This exercise was also carried out periodically during the course of the survey in order to maintain accuracy. The size-class intervals were thus set at 5 centimetres. Based on my master's research field work within the same region, I have acquired the necessary proficiency to estimate fish within this size range.

The direction of every transect was haphazardly determined based on the prevailing current and tide at the time of each dive. The parameters of the transect (100 x $10m^2$) allowed for $1000m^2$ on the reef face to be visually surveyed for the presence of scarids. Within this dimension, parrotfish numbers were recorded within the 5 minute timed swim representing a 100 meter transect. Individuals were placed within 5cm size categories, with fishes less than 15cm in size ignored, because the inclusion of small fish while conducting surveys of large areas may increase estimation error (Belwood and Alcala 1988). Any parrotfish that entered the 100m strip was recorded, regardless if they proceeded to immediately exit the line of sight.

3.4 Data analysis

UVC count data were entered into Excel spreadsheets and the services of statistical software SPSS (Version 11.02, SPSS Inc., Chicago, IL, USA) was employed for further analysis. Species-specific analysis was preferred over pooled data in order to investigate possible shifts in abundance and distribution patterns of individual parrotfish species.

Density figures were formulated from summary statistics (SPSS) and used to graph the density estimates of scarids. Density estimates are expressed as mean ± standard error of the mean (SE) per 100 meter transect. All other significant interactions including size distribution were graphed using this information. In order to investigate possible temporal shifts in size structure and distribution in Gizo, further analyses were carried out. In relation to size structure, average size of the six species were calculated and plotted. In order to investigate if average size had significantly changed over time a single factor ANOVA was employed to test for differences. The mid points of size categories were allocated to individual counts (e.g. 22cm for size category 20-24cm) and then used as replicates in analysis of variance (Sluka and Sullivan 1998).

In order to investigate potential differences in total density, one-way ANOVA was formulated for each of the six species using SPSS. If a difference was shown to exist (i.e $P<0.5$) further analysis was employed to investigate the results in more detail (i.e particular size-class). As a general rule of thumb most count data are highly skewed thus requiring transformation before analysis of variance can take place. Therefore, homogeneity of variances was tested prior to all ANOVA calculations in SPSS and if heterogeneity of variances was discovered (i.e $P<0.05$) the data was log transformed (log of abundance+1) before analysis was continued.

Power-analysis was conducted on an UVC dataset (fifty-one transects conducted in 2004) from nearby village of Vavanga (Site 1 in Figure 3.5) on Kolombangara Island, the site of my previous research expeditions. CPUE, both conducted for this research program (data not shown) and also previously (Sabetian 2003), confirmed that parrotfishes are subjected to negligible fishing pressure. The presence of this relatively healthy village fishery on the immediate boundary of Gizo's large open access and highly competitive artisanal fishery, is a testament to the restrictive ability of land and marine tenure in Western Solomon Islands. It also highlights the localised nature of coral reef fish assemblages, where different levels of fishing effort within relatively small distances can lead to markedly different standing stocks.

Because of its lightly fished status, data from Vavanga provides the closest estimate to "natural" parrotfish abundance. Power analysis permitted me to calculate how much the population, as a measure of mean density, would have to change before it could be confidently detected. Log transformed data were entered into the following formula (Zar, 1999), where the power was specified as 90% (β=0.1), at a 5% level of significance (α= 0.05).

$$\delta = \sqrt{\frac{s^2}{n}}\,(t_{\alpha(2),\nu} + t_{\beta(1),\nu}),$$

where δ = detectable difference, s^2 = variance, n = sample size, and $\nu = n-1$.

3.5 Results

3.5.1 Density estimates

The following figures represent mean scarid density per transect (±Standard Error) across size categories (TL cm), and also in total. The histograms represent mean number of fish per $100m^2$ averaged over 47 transect for Gizo in 2004, and 41 transects for the same site in 2005.

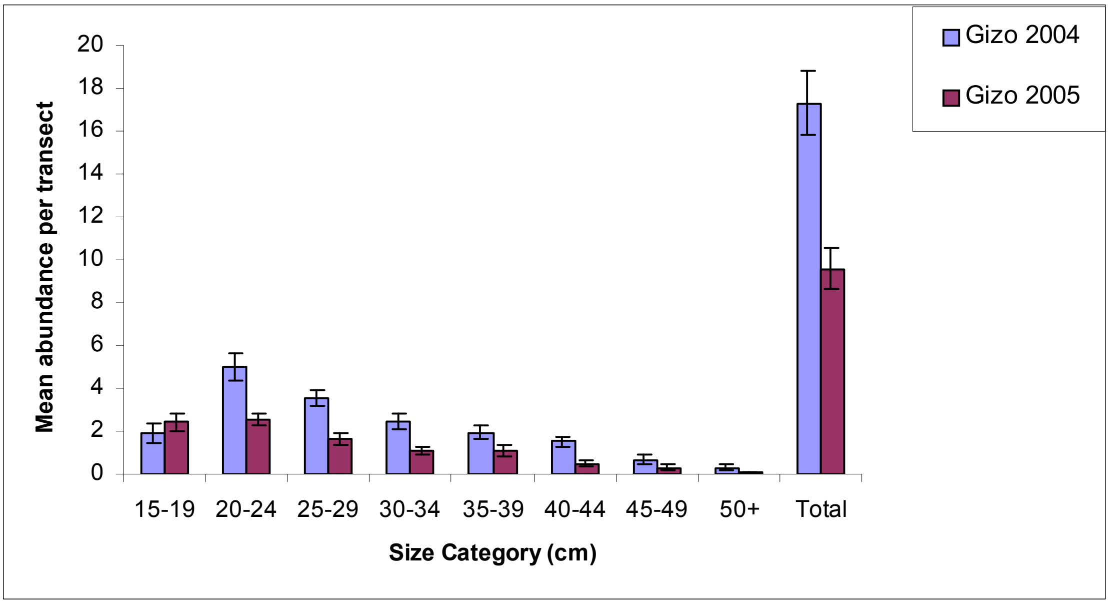

Figure 3.7. Mean overall density partitioned by size-class and pooled (Error bars = ± 1 SE).

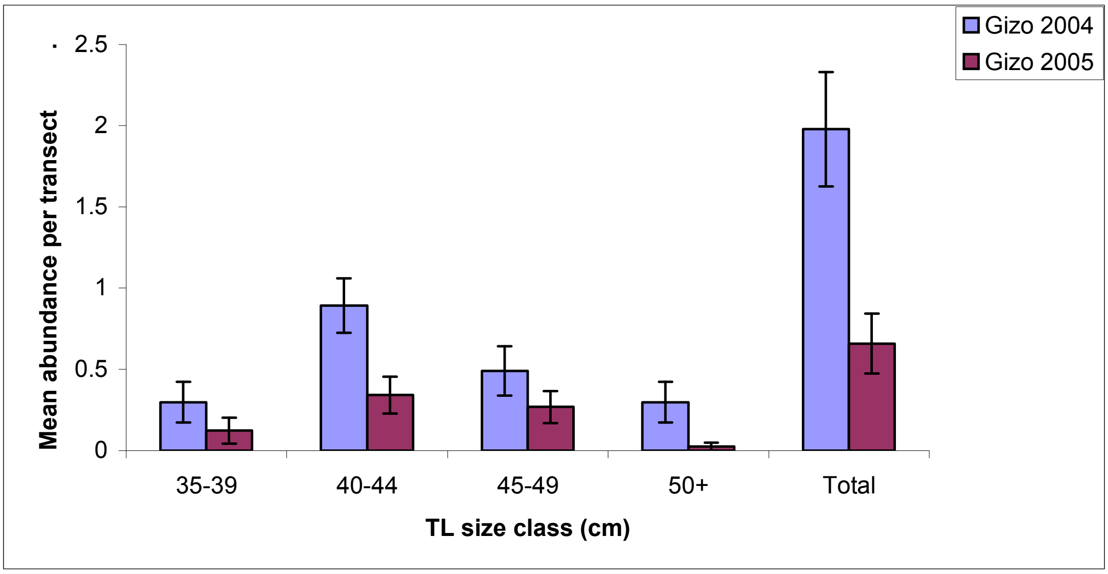

Figure 3.8. Mean density for *Bolbometopon muricatum* (Error bars = ± 1 SE).

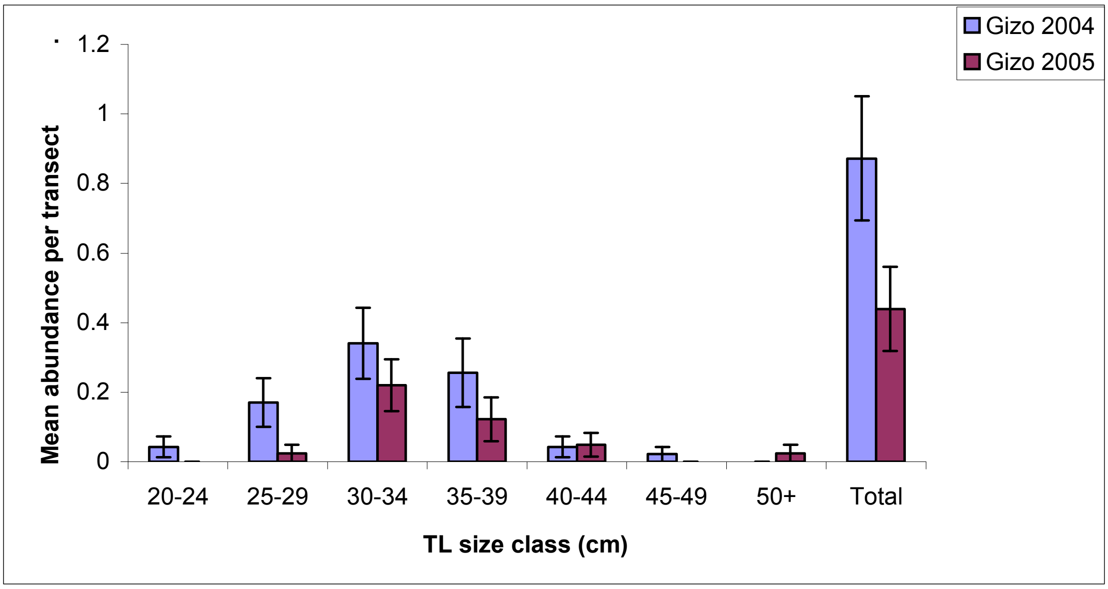

Figure 3.9. Mean density for *Cetoscarus bicolor* (Error bars = ± 1 SE).

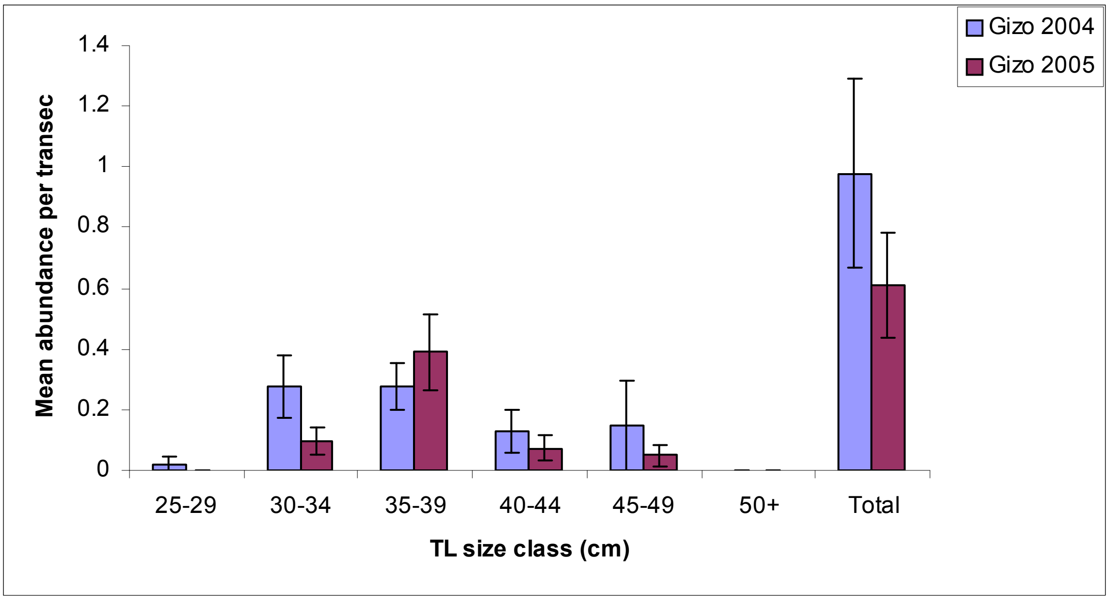

Figure 3.10. Mean density for *Chlorurus microrhinos* (Error bars = ± 1 SE).

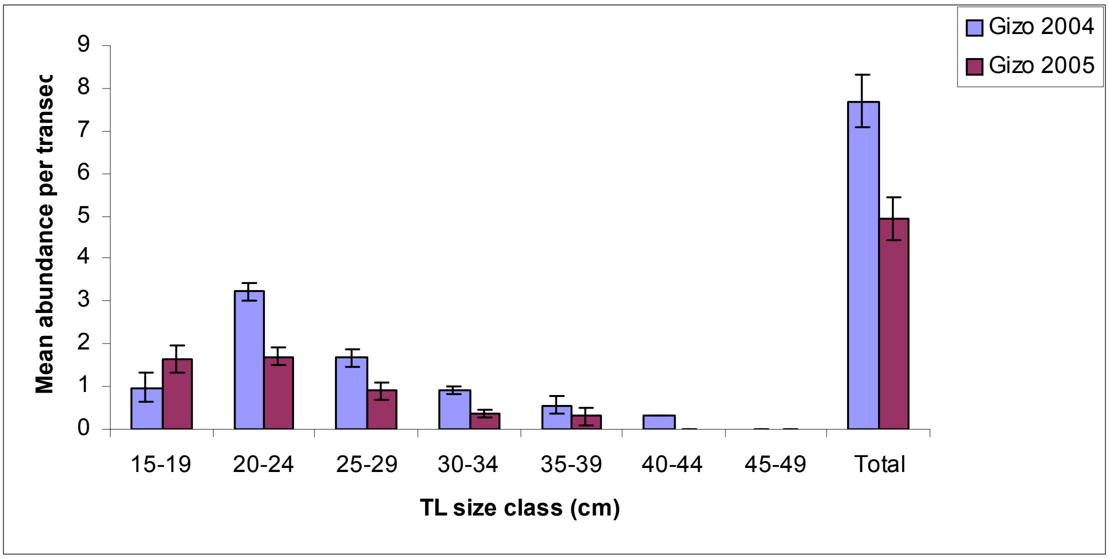

Figure 3.11. Mean density for *Hipposcarus longiceps* (Error bars = ± 1 SE).

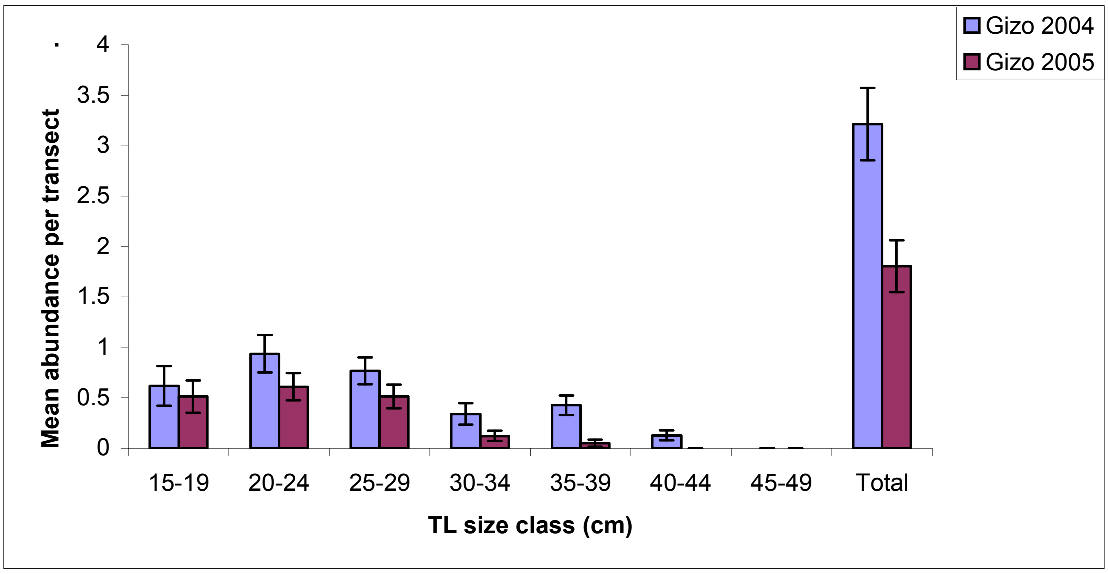

Figure 3.12. Mean density for *Scarus ghobban* (Error bars = ± 1 SE).

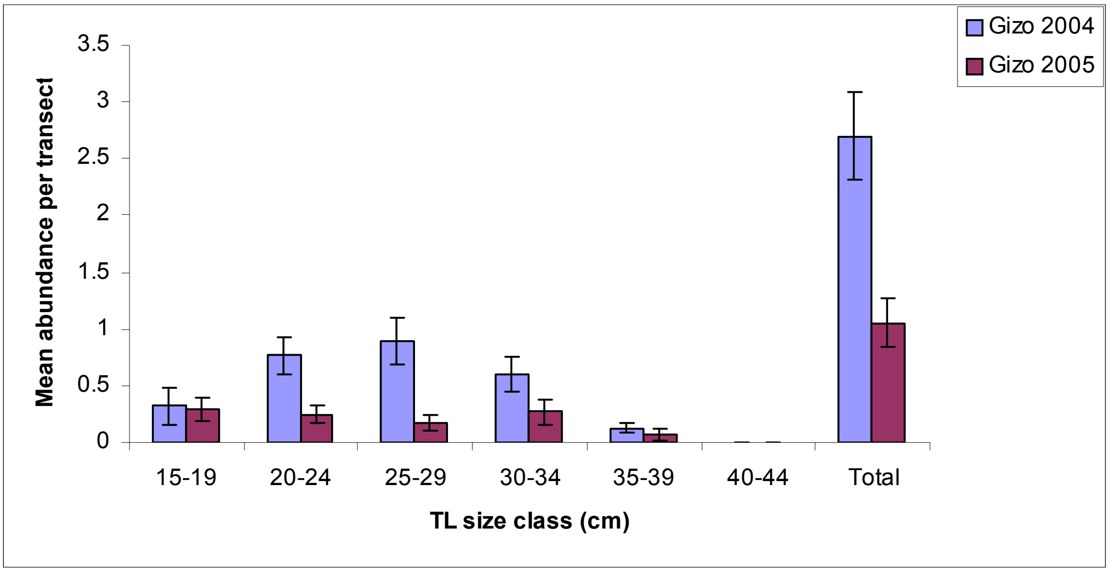

Figure 3.13. Mean density for *Scarus rubroviolaceus* (Error bars = ± 1 SE).

The presence of all six largest scarid species, including the largest (*B.muricatum*), is an indication that parrotfish stocks within the Gizo area are still relatively healthy compared to other locations within the Pacific where they are rarely sighted (e.g. Page 1998). Mean combined parrotfish density (± SE) was 17.0 ± 2.2 individuals per 100m^{-2} for Gizo in 2004, and 12.4 ± 1.9 individuals per 100m^{-2} for Gizo in 2005. However, the overall indication that parrotfish density has declined in the Gizo area can only be accurately examined if we closely analyse individual density estimates over time (Table 3.1).

	Gizo 2004	**Gizo 2005**
B. muricatum	2.0 ± 0.4	0.7 ± 0.2*
C. bicolor	0.9 ± 0.2	0.4 ±0.1
C. microrhinos	1.0 ± 0.3	0.6 ± 0.2
H. longiceps	7.7 ±0.6	4.9 ± 0.5*
S. ghobban	3.2 ±0.4	1.8 ± 0.3*
S. rubroviolaceus	2.7 ± 0.4	1.1 ± 0.2*

Table 3.1. Mean number of individuals per transect ± standard error (* indicates statistical significance; P<0.05).

To establish if the estimated abundance shifts of the Gizo data were statistically significant comparative analysis was carried out via ANOVA. In particular, changes in specific size-class densities of some species were formulated.

3.5.2 Analysis of variance

The following tables represent results for single factor analysis of variance showing the influence of time (2004 vs 2005) on the density of parrotfishes for Gizo. Where deemed important, comparisons of size-class density estimates over time have also been included. P values of less than 0.05 indicate significant differences.

Source	Sum of Squares	DF	F	P
40-44cm	1.608	1	6.438	0.013
50+cm	0.401	1	4.458	0.038
Overall Density	4.066	1	8.089	0.006
Total Residual	43.302	87		

Table 3.2. ANOVA estimates for *Bolbometopon muricatum*.

Source	Sum of Squares	DF	F	P
Overall Density	0.822	1	3.284	0.073
Total Residual	21.345	87		

Table 3.3. ANOVA estimates for *Cetoscarus bicolor*.

Source	Sum of Squares	DF	F	P
Overall Density	0.148	1	4.414	0.522
Total Residual	30.818	87		

Table 3.4. ANOVA estimates for *Chlorurus microrhinos*.

Source	Sum of Squares	DF	F	P
15-19cm	2.611	1	4.790	0.031
20-24cm	1.967	1	4.218	0.043
25-29cm	2.422	1	6.377	0.013
30-34cm	1.334	1	5.321	0.023
40-44cm	0.639	1	7.308	0.008
Overall Density	3.174	1	7.014	0.01
Total Residual	42.086	87		

Table 3.5. ANOVA estimates for *Hipposcarus longiceps*.

Source	Sum of Squares	DF	F	P
35-39cm	1.264	1	14.085	0.000
40-44cm	0.171	1	5.864	0.018
Overall Density	2.740	1	5.586	0.020
Total Residual	44.917	87		

Table 3.6. ANOVA estimates for *Scarus ghobban*.

Source	Sum of Squares	DF	F	P
20-24cm	1.286	1	6.177	0.015
25-29cm	2.454	1	11.279	0.001
Overall Density	5.355	1	10.434	0.002
Total Residual				

Table 3.7. ANOVA estimates for *Scarus rubroviolaceus.*

ANOVA figures shed more light on the temporal density estimates of scarids from Gizo. Abundance of *B.muricatum* (P=0.006), *H.longiceps* (P=0.01), *S.ghobban* (P=0.02), and *S.rubroviolaceus* (P=0.002) significantly declined from 2004 to 2005, while no significant change was observed for *C.microrhinos* (P=0.522) and *C.bicolor* (P=0.073). Further analysis into specific size classes revealed that the abundance of *H.longiceps* significantly declined across all size-classes (except for an increase in 15-19cm), while the number of larger individuals of *B.muricatum* and *S.ghobban* were also significantly lower in 2005 compared to 2004. This trend confirms that fishing pressure impacts disproportionately on larger individuals (As also stated by Hawkins and Roberts 2003).

3.5.3 Power analysis

The purpose of power analysis is to independently assess how accurately a sample size can be used to detect changes in mean density over time. In this case, I have decided to use the Vavanga dataset (sample size of 51 transects, n=51) given that it has historically been subjected to a stable level of subsistence fishing pressure and as such can be referred to as representing a "healthier" parrotfish assemblage. The figures below (Table 3.8) represent the minimal detectable change in average density from the Vavanga dataset. In other words, given their current stock status, density changes above these estimates can be regarded as statistically significant.

Species/genus	**δ(original scale)**	**%of mean(original scale)**
B.muricatum	1.5575	26.04
C.bicolor	1.011	65.27
C.microrhinos	1.406	71.71
H.longiceps	1.424	18.16
S.ghobban	1.347	27.05
S.rubroviolaceus	1.435	44.63

Table 3.8. Minimum detectable change in parrotfish density (n=51).

Comparative temporal analysis on Gizo sheds more light on the possible effects of fishing pressure. From 2004 to 2005 overall mean density estimates (per count) dropped by approximately 67% for *B.muricatum*, 50% for *C.bicolor*, 38% for *C.microrhinos*, 36% for *H.longiceps*, 44% for *S.ghobban*, and 61% for *S.rubroviolaceus*. Power analysis on the Vavanga dataset (Table 3.8) reveals that minimum detectable percentage change in abundance, as a function of average density, would have to be at least 26% for *B.muricatum*, 65% for *C.bicolor*, 71% for *C.microrhinos*, 18% for *H.longiceps*, 27% for *S.ghobban*, and 44% for *S.rubroviolaceus*. Given the similar sample sizes of all three datasets, we can be tentatively confident that Gizo density estimates have indeed declined for *B.muricatum*, *H.longiceps, S.ghobban* and *S.rubroviolaceus*, while the declines for *C.bicolor* and *C.microrhinos* are below minimum detectable levels identified by power analysis and thus statistically weak.

3.5.4 Size distribution

Analysis of size distribution over time allows fisheries managers to detect temporal changes and exercise appropriate reactionary measures. Below, size structure of the six targeted species from Gizo have been used to graph their frequency distribution, followed by analysis of average size over time.

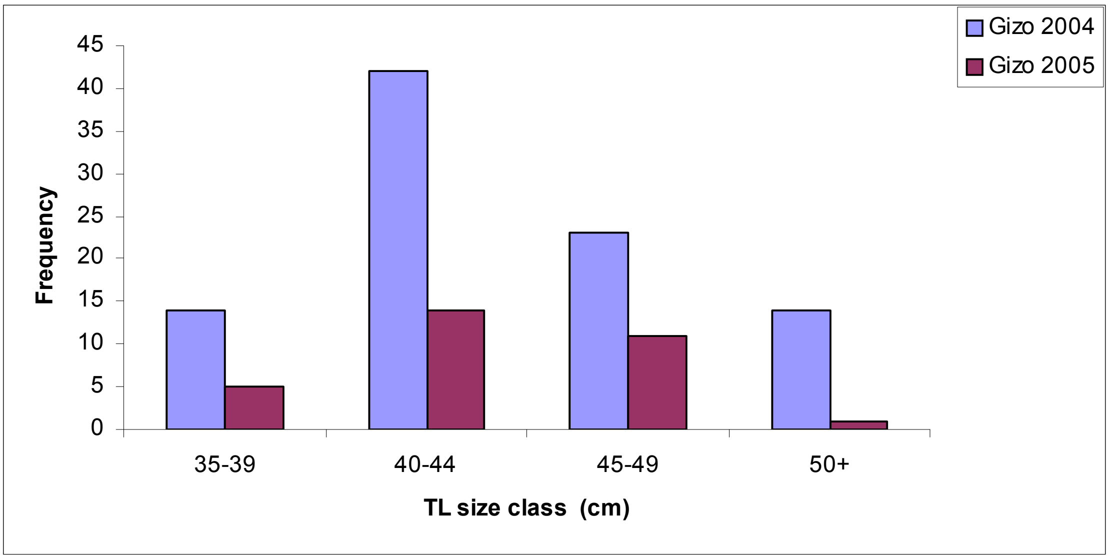

Figure 3.14. Size-class distribution of *Bolbometopon muricatum* from Gizo.

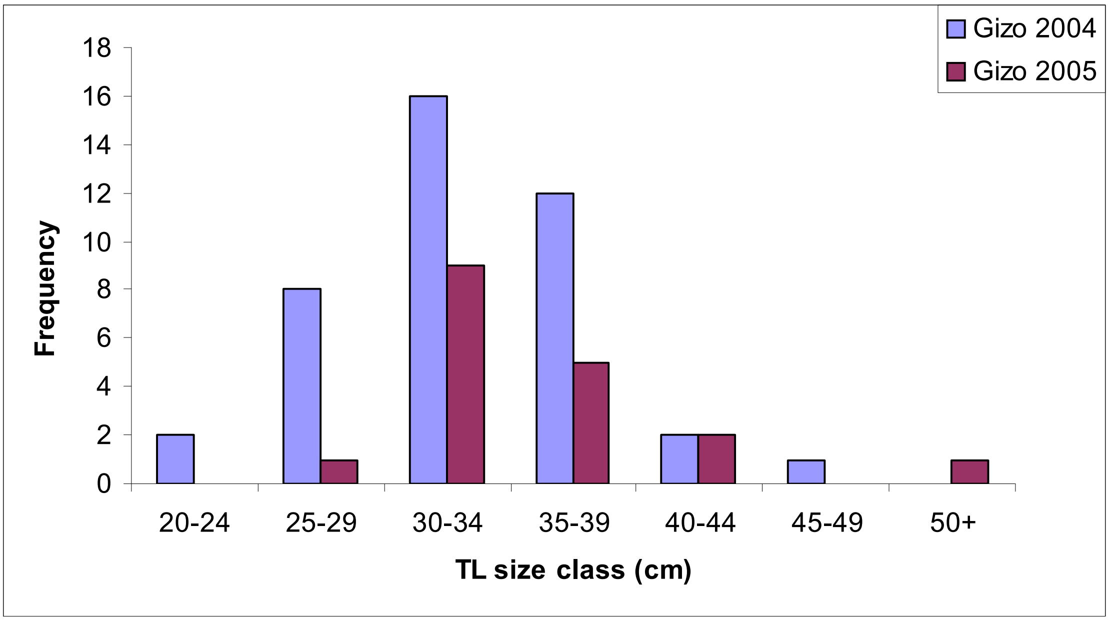

Figure 3.15. Size-class distribution of *Cetoscarus bicolor* from Gizo.

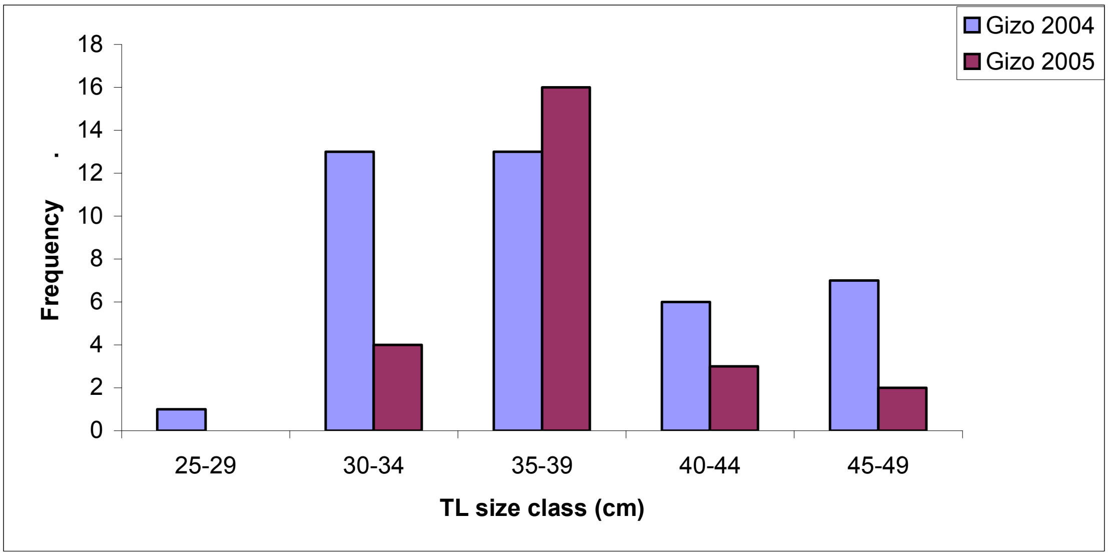

Figure 3.16. Size-class distribution of *Chlorurus microrhinos* from Gizo.

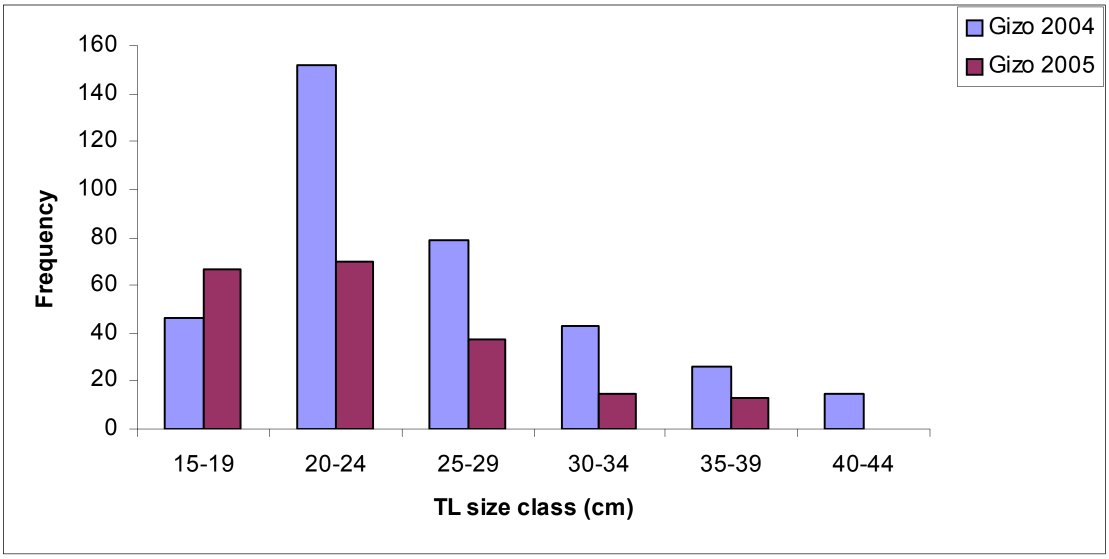

Figure 3.17. Size-class distribution of *Hipposcarus longiceps* from Gizo.

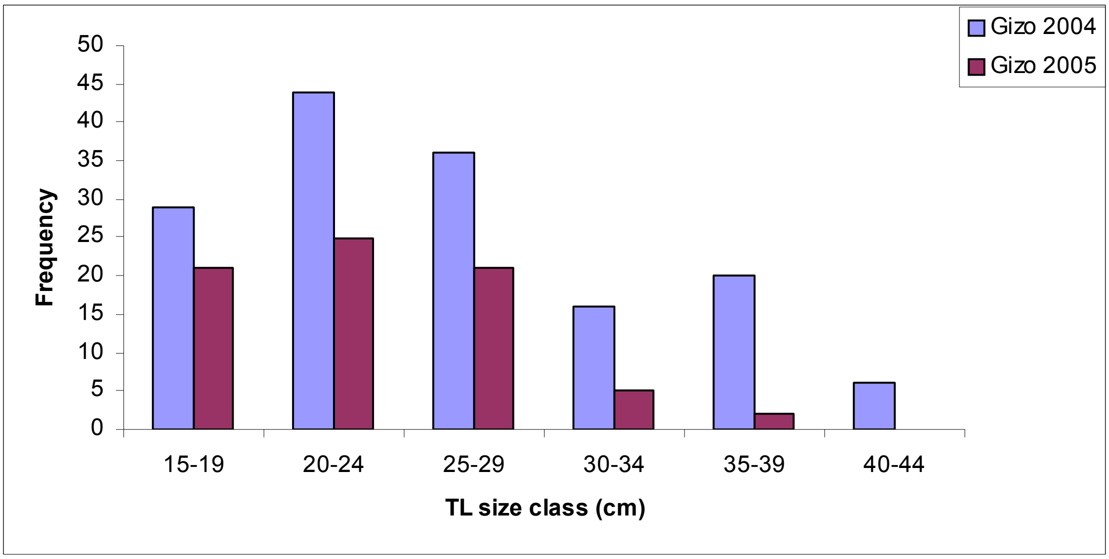

Figure 3.18. Size-class distribution of *Scarus ghobban* from Gizo.

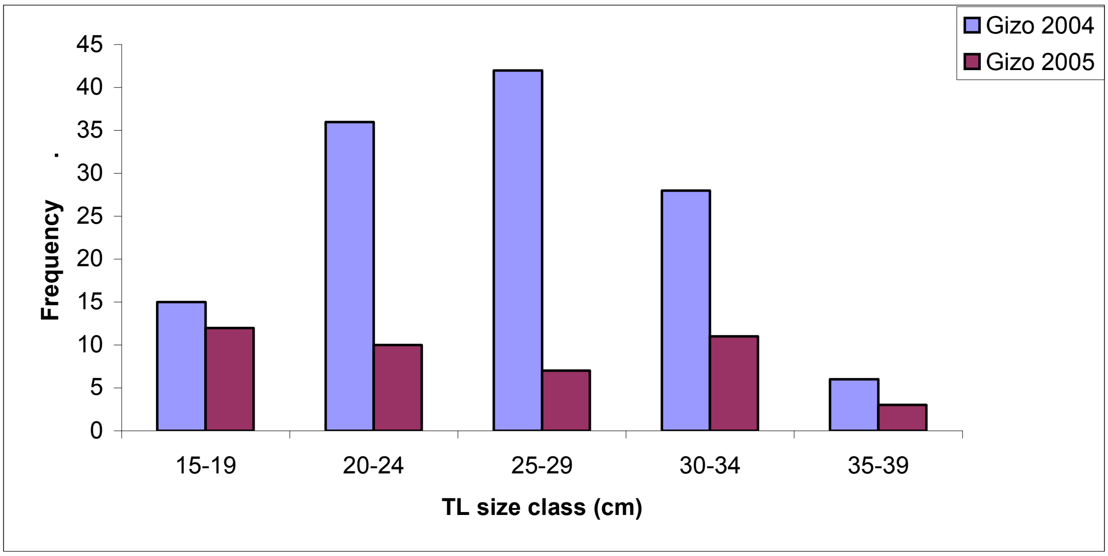

Figure 3.19. Size-class distribution of *Scarus Rubroviolaceus* from Gizo.

Average size (TL) of all six species (mean ± 1 SE) was calculated and compared over time for the Gizo. A single factor ANOVA was employed to test for significant shift in average size over time.

Species	Mean size 2004 (± 1 SE)	Mean size 2005 (± 1 SE)
B.muricatum	43.69 (0.47)	43.14 (1.01)
C.bicolor	32.85 (0.82)	35.22 (1.26)
C.mirorhinos	37.63 (0.9)	37.6 (0.78)
H.longiceps	25.56 (0.34)	22.97 (0.41)*
S.ghobban	26.07 (0.58)	23.24 (0.66)*
S.rubroviolaceus	25.98 (0.47)	25.02 (1.01)

Table 3.9. Mean size± 1 SE (TL) of scarids at Gizo.

Source	Sum of Squares	DF	F	P
B.muricatum	6.693	1	0.410	0.523
C.bicolor	70.174	1	2.528	0.117
C.mirorhinos	0.010	1	0.000	0.985
H.longiceps	871.687	1	22.705	0.000
S.ghobban	397.629	1	8.923	0.003
S.rubroviolaceus	29.182	1	0.904	0.343

Table 3.10. ANOVA estimates for mean size of scarids at Gizo.

Size class distribution has revealed significant differences in total frequency across all species. For Gizo, the general trend appears one of lower total individuals sighted across many size classes between 2004 and 2005. However, *H.longiceps* appeared to defy this trend by showing increased abundance across the smaller size classes (15-19cm). The reason for this may be two-fold. It could either indicate that more juveniles have been recruited into the local population for that year, or that smaller individuals are now more noticeable to the diver in the absence of larger individuals. Average sizes (TL) of all six species were calculated and compared over time, from which *S.ghobban*, *H.longiceps*, and *C.bicolor* showed declining shifts over time. ANOVA estimates revealed the decline in average size for *S.ghobban* (P=0.003) and *H.longiceps* (P=0.000) to be statistically significant, while the increase in size for *C.bicolor* was deemed statistically insignificant (P=0.117), proven by the fact that it was due to a solitary large individual sighted in 2005.

3.6 Discussion

Analysis of data collected as part of this study achieves the three-fold objectives listed in the beginning.

For the first objective, the temporal variability of parrotfish density and size-structure estimates from the Gizo reefs deserves a two-fold analysis. Firstly, we are now able to assess the degree to which density and size-structure has shifted over a 12 month period, but more importantly, we are able to discuss these shifts with consideration to the trends of artisanal and creel CPUE estimates depicted in the previous chapter. Analysis of data from chapter Two (CPUE and Creel) and here (UVC), combined, make a compelling case to support the notion that fishing pressure has had a predictable effect on parrotfish stocks in the Gizo region. In this chapter, the average density of four out of the six targeted species declined from 2004 to 2005. In particular, the density estimate declines of *B.muricatum* (P=0.006), *H.longiceps* (P=0.01), *S.ghobban* (P=0.02), and *S.rubroviolaceus* (P=0.002) were shown to be statistically significant. However, no significant change was observed for *C.microrhinos* (P=0.522) and *C.bicolor* (P=0.073). There may be several reasons for this, including observer bias, or the fact that these species are naturally scarce and only aggregate to spawn in large numbers (personal observation of mass *C.microrhinos* aggregations). Whatever the reason, independent power analysis of the Vavanga dataset confirms the natural scarcity of these two species and vindicates the insignificant P values reached from ANOVA. Power analysis on Vavanga's larger parrotfish assemblage reveals that even in a relatively pristine state, based on current abundance estimates, the density of *C.microrhinos* and *C.bicolor* would have to change by 71% and 65 %, respectively, before they could be statistically noticed.

With respect to the second objective of this chapter, size structure analysis, in terms of density, frequency, and mean body size has produced some interesting results. Hawkins and Roberts (2003), state that fishing pressure has a predictable disproportionate influence on larger individuals. Page (1998) highlighted this point by drawing attention to the fact parrotfish species *S.rubroviolaceous* and *C.gibbus*, the terminal phase males of which attain maximum sizes above 50cm, were heavily targeted in the American Samoan night spearfishery. Page attributed this to the large sizes of this species, which restricts their sleeping sites to more open areas. As a result, they are more accessible to spearfishers, compared to smaller scarids which are able to hide in corals trees or crevices. This notion has been validated by the precipitous decline in size-structure specific density of parrotfishes from the Gizo region. Although density estimates significantly declined (i.e $P<0.05$) across most size classes for the above four highlighted species, larger bodied *B.muricatum* and *S.ghobban* (>40cm TL) also clearly suffered from sustained fishing pressure.

In relation to size class frequency, total numbers sighted for the Gizo assemblage were noticeably lower across most size classes in 2005. One species which did not follow this trend was *H.longiceps*, the total sighted numbers of which increased in the 15-19 cm size range. Given the prevalence and juvenile schooling nature of this species on the reef (personal observation), this may have been as a result of recently recruited juveniles into this particular size class. This is entirely possible given the fact that the UVC ignored fishes below 15cm TL, making their recruitment into this size class more noticeable to the diver in 2005. Finally, the transition in average sizes (TL) of the targeted species reveals some interesting trends. It appears that although density and size class frequency has significantly decreased for most species, only *S.ghobban* ($P=0.003$) and *H.longiceps* ($P=0.0001$) exhibited statistically significant shifts in average size over time. ANOVA estimates show that *S.ghobban* mean size decreased from 26.07± 0.58 cm (±SE) in 2004 to 23.24 ± 0.66 cm in 2005, while mean size for *H.longiceps* declined from 25.56 ± 0.34 cm in 2004 to 22.97 ± 0.41 cm in 2005.

Overall, Gizo's density and size-class distribution portray a parrotfish assemblage that is still relatively healthy, although declining under the quantified fishing pressure. The indications that the local population is susceptible to fishing pressure are the noticeable declines is larger bodied individuals, combined with the decreasing average size in two of the more prevalent species (*S.ghobban* and *H.longiceps*) on the reef. The overall indications are that Gizo's parrotfish stocks are showing definite signs of susceptibility to the current level of fishing pressure, which within the context of the relatively small local human population signifies the inability of parrotfishes to stay resilient against fishing pressure.

The final objective of this chapter was to conduct power analysis on an independent dataset in order to gauge the ability of an UVC monitoring program to pick up declines (or increases for that matter) in density estimates. The Vavanga dataset acted as a proxy for this analysis. The overall conclusion was that even under pristine conditions it is practically difficult to detect small to moderate changes in density estimates. This is especially evident for naturally scarce or rare species. As part of my master's research program in 1999, I conducted a long-term UVC program in Vavanga (Sabetian 2002). In total, I conducted 94 transects to estimate the abundance and distribution of several grouper species. Even though the status of grouper stocks were extremely healthy on Vavanga's reefs, power analysis revealed that density changes in naturally scarce species were difficult to detect. Although one could suggest that monitoring programs need to be more long-term, it is not guaranteed that several hundred transects would definitely increase the strength of estimates. Furthermore, not only such surveys are practically difficult to maintain over long periods, their relative high costs and laborious nature place them outside the capabilities of most local management authorities.

Given that uncertainties are ubiquitous in fisheries (Charles 2000), analysis of overall trends and specific abundance indicators, in combination with fishery related data (e.g. CPUE and Creel) can substitute for long-term UVC surveys in assessing the relative impacts of fishing pressure on fish assemblages. Based on this assertion, the overall results of this chapter confirm that fishing pressure from the Gizo fishery has induced: (a) declining population densities, (b) shifts in size-structure, and (c) reduced average body size. It is therefore reasonable to assume that such transition will have ecological implications given the functional roles of scarids within coral reef ecosystems.

There is increasing pressure on local fishers to maximize their catch or ideally target financially rewarding species. Such exploitation is an inevitable evolution of artisanal fisher behaviour in developing countries where economic realities dictate changes in the utilisation of marine resources. However, the Pacific Islands, with their relatively lower population and extensive areas of coral reefs have been able to meet this demand through expansion of fishing activity onto more productive grounds (Adams et al. 1996; Sabetian and Foale 2006). It is probable to assume that this may have been responsible for masking localised fishery crashes. The markedly different parrotfish stocks of Gizo and Vavanga appear to support this hypothesis.

Observations discussed in this chapter have highlighted the important role of fishing pressure as a determinant of reef fish assemblages. However, it is difficult to speculate on the magnitude and severity of these changes given the fact that obvious ecological stress signs only become evident after severe or irreversible environmental damage (Ruttenburg 2001). This study provides the first evidence of parrotfish community changes from the Gizo artianal fishery. Admittedly, I have not investigated the role of other spatial and temporal operating factors such as habitat availability, recruitment and settlement. However, the data presented here act as an important base-line which can compliment future studies and also provide a vital historical perspective against which fishing pressure can be compared.

The impacts of fishing on coral reefs are afforded more attention as the pressure on these resources is increasing (Polunin and Graham 2003). Prevailing life history and demographic studies also implicate fishing pressure as one of the primary contributors of changing reproductive and growth related parameters (e.g. Hawkins and Roberts 2003). In the next two chapters, I present the results of a comparative biological study where parrotfish demography and life history data are analyzed from several different locations.

Chapter Four

Parrotfish demography; a comparative analysis

Demographic information on five species are presented in this chapter; Pacific Steephead parrotfish Chlorurus microhinos, Indian Steephead parrotfish Chlorurus strongylocephalus, Blue-barred Parrotfish Scarus ghobban, Redlip parrotfish Scarus rubroviolaceus, and Pacific longnose parrotfish Hipposcarus longiceps. The C.strongylocephalus dataset represents the work of Mr Alec Hughes, acquired as part of a collaborative study on parrotfish demography and life-history with Professor Howard Choat.

4.0 Introduction

The previous two chapters highlighted the importance of ecological and fishery-related information in coral reef fisheries. Of equal importance are knowledge of demographic and life-history parameters. Although such information is important in the development of coral reef fish management and conservation initiatives for a majority of targeted species, it remains sparse or non-existent. This chapter summarizes information on the demography of a number of parrotfish species that are relevant to the fisheries of the Solomon Islands.

There is now a concerted effort to investigate and contribute to the knowledge-base of growth and reproductive related information in order to help answer some of the most critical management and conservation questions. These include the issue of habitat-specific plasticity in demographic variables, the significance of fishing pressure in inducing life-history shifts, and the ability of depleted stocks to recover through protection from fishing pressure. Demographic and life-history information have key roles in helping to address these issues.

Because tropical reef fishes display a wide range of temporal and spatial plasticity in demographic and life-history parameters, this has strong implications in development of regional management initiatives. Demographic variation may occur at very local scales (Gust et al. 2002) or across broad geographic regions (Choat et al. 2003).

The role of demography and life-history specific traits in over-exploitation have also been emphasized in other studies (Jennings et al. 1999; Dulvy et al. 2003). The general notion is that the vulnerability of coral-reef fishes is accentuated by specific life-history adaptations where faster growing and high turnover species are able to sustain higher fishing pressure, compared to slow growing and low turnover species. Given that some species have been shown to display high levels of phenotypic plasticity (Gust 2000) the link between population parameters and vulnerability to over-exploitation also needs to be further investigated. Furthermore, studies are also showing that sustained fishing pressure can alter sex-specific ratios through increased selective mortality, thus inducing species to change their internal reproductive clock in relation to maturation, reproductive output and sex change (Hawkins and Roberts 2003; Palumbi 2001).

Given the susceptibility of scarids to overfishing, the removal of fishing pressure through the establishment of permanent no-take zones or Marine Protected Areas (MPA) may perhaps be the most appropriate method with which to exercise conservation and rehabilitation. Although debate surrounds the speed with which species may respond to removal of fishing pressure (Halpern and Warner 2002; Russ and Alcala 2004), we do know that the capacity to respond to protection is both taxon and demography specific (Russ and Alcala 2003).

Reliable age estimation of coral reef fishes have advanced since the validity of sagittal otoliths were confirmed (e.g. Choat et al. 1996, Cappo et al. 2000, Pilling et al. 2000, Choat and Robertson 2002). Age-based examination of coral reef fish demography has been able to dispel some old myths and inform us of new interesting characteristics. For example, it was historically assumed that coral reef fish exhibit high turnover rates, based on faster growth, short longevity, and high relative natural mortality (Sale 2002). However, more recent studies have clearly highlighted the complex nature of coral reef fish demography. Gust (2000), for example, underlined the fact that sound management and conservation initiatives must be based on the fact that coral reef fish demography and life history patterns are heterogenous. This was perhaps best demonstrated by the contrasting conclusions of Gust (2000) and Hamilton (2004), where the former author showed that the growth of smaller parrotfish species maybe density dependant, while the latter author showed that the opposite was true with the largest parrotfish, *B.muricatum* (see discussion). This example clearly dispels any generalised notions about coral reef fish demography, even within the same clade.

4.1 Objectives

This chapter satisfies the demographic aspects of aim two by conducting a comparative age-based analysis of parrotfishes across locations from several ocean systems. The specific objectives were to:

(1) Assess the spatial plasticity of parrotfish age and sex structure.
(2) Examine differences in size and growth rates.
(3) Examine differences in longevity.
(4) Examine differences in mortality.

4.2 Methods and Materials

4.2.1 Sites and study species

This study presents analysis on several datasets acquired from numerous locations within the Pacific, Indian and Caribbean ocean systems from a number of parrotfish species. In total, age related information on five species have been included in this study; Pacific Steephead parrotfish *Chlorurus microhinos,* Indian Steephead parrotfish *Chlorurus* *strongylocephalus,* Blue-barred parrotfish *Scarus ghobban,* Redlip parrotfish *Scarus rubroviolaceus,* and Pacific longnose parrotfish *Hipposcarus longiceps.*

Otolith samples from the Solomon Islands were collected. Otoliths from *S.ghobban* specimens were extracted during night time spearfishing expeditions from the Gizo reefs (referring to the open access reefs discussed in chapters 2 and 3), while *H.longiceps* were extracted from the uninhabited region of neighbouring Rarumana Island. The *H.longiceps* dataset is therefore representative of a lightly fished region. Samples of *C.microrhinos* and a handful number of other scarids from the Great Barrier Reef (GBR) were primarily acquired with the assistance of Dave Welch and the Townsville spearfishing club. The majority of samples in this analysis represent fish that were speared during an annual spearfishing competition, and as such represent a size- and age-biased dataset. A small number of GBR specimens were also provided by Professor J.H Choat. All other specimens from Western Australia, Oman, Seychelles, Taiwan and Panama were provided by Professor J.H Choat as part of a large scale coral reef fish demographic investigation. The Cocos Keeling dataset for *C.strongylocephalus* was kindly provided by Alec Hughes who is also a collaborative participant of Professor Choat's programme. This dataset is also currently being used in preparation of a co-authored manuscript for publication.

Where possible complete datasets are presented in a comparative manner. However, on several occasions these datasets are supplemented with small numbers of opportunistically acquired specimens. Although the latter samples cannot be included in the overall study, they can be used to make anecdotal assumptions. More specific information on each species and location are presented below.

4.2.2 Sampling procedures

4.2.2.1 Ageing

Prior to the removal of sagittal otoliths from the brain cavity of specimens, information such as size (SL and FL), Total weight (gm), colouration phase (IP/TP) and any interesting features (e.g. deformations, scars) were recorded. Where possible gonads were sexed macroscopically. However, histological procedures were also used to sex *S.ghobban* from Solomon Islands*, C.microrhinos* from the GBR*, and C.strongylocephalus* from Cocos Keeling. Gonads removed for this purpose were labelled and stored in preservative solution for histological analysis (chapter five). Both sagittal otoliths were removed (if possible), washed and cleaned in water, dried and stored in culture plates. Once in the laboratory a single otolith was weighed to the nearest 0.0001 mg, and subsequently used for ageing analysis. The other otoliths from the original pair were stored separately as spares, if needed.

Otoliths were prepared for age analysis using the methodology described by William Robbins (unpublished guidebook, James Cook University). Otoliths were positioned on the edge of a glass slide using thermoplastic glue (Crystal Bond^{TM}) and ground down close to the nucleus using an 800 grade abrasive diamond encrusted plate. The glue was melted again using a hotplate before the half-cut otolith was mounted grounded-face down on the middle of the slide and ground down again leaving only a thin section. Sectioned saggitae were then positioned back on the hotplate and smeared in crystal bond to improve the visibility of opaque bands. Subsequently, each otolith was viewed under a high-powered microscope with transmitted light which was electronically linked via a mounted camera onto a computer screen. Given the large number of studied species in this chapter, validation of annual markings using chemical tracers was not practical. However, the internal opaque incremental bands clearly visible in the sectioned sagittae were presumed to be annual markings based on descriptions of internal otolith structure of the age-validated Caribbean stoplight parrotfish *Sparisoma viride* conducted by Choat et al., (2003) and also other parrotfish species (Choat et al., 1996). Anticipating that otoliths weighing less than 0.01 mg would not display visible annual increments, they were sectioned using a 1200 grade abrasive diamond encrusted plate (if possible) or ground by hand and polished on a 3 or 9 µm grade lapping paper. These were then read under a high-powered microscope (similarly connected to a computer screen) using the daily ring increments described in a validated study on juvenile parrotfish by (Lou and Moltschaniwskyj 1992). Daily counts were converted into annual estimates by dividing the estimate by 365. In all occasions, sectioned otoliths were read twice for annuli if both estimates were the same or read further time if the first and second estimates did not match. If two out of three estimates matched then that number was used as final age, but if all three estimates differed, then expert advice was sought from other experienced colleagues until a figure was agreed upon.

4.2.2.2 Analysis

There are a number of growth functions available to fishery scientists. However, by far the most popular is the Von Bertalanffy growth function (VBGF) which is used for length-based analyses of growth and mortality. The VBGF was deemed most appropriate for providing the best fit to size-at-age data according the parameter estimates of the Schnute (Schnute 1981) growth function. Von bertalanffy growth curves were generated using a Macro function developed by James Cook University graduate John Ackerman in Excel^{TM} and later modified by Will Robbins. The function itself is described as $L_t = L\infty \{1\text{-exp} [-k (t - t_0)]\}$ where Lt is the length at age t, L is the mean asymptotic length, K is the growth coefficient that describes the rate at which L reaches asymptote, t is the age of the fish in years and t_0 is the hypothetical age at conception or at length zero. Given the large number of datasets, and the lack of representative populations containing sufficient samples of very young specimens, it was deemed difficult to ascertain size at settlement. Thus based on scarid settlement information provided by Leis and Rennis (1983) all samples were initially analysed using a constrained VBGF macro in Excel^{TM} where t_0 was constrained to 20mm. Subsequent estimation of t_0 after analysis is described.

Regression analysis was conducted to ascertain the relationship between fish age and otolith weight. It is generally accepted that otoliths must grow throughout the lives of a fish at a perceptible rate (Choat et al., 2003) and a significant quadratic relationship is indication of a positive relationship between age and otolith weight. For example, Choat et al., (2003) demonstrated that the sagittae of the parrotfish *Sparisoma viride* accreted weight in a predictable fashion over the lifespan of the fish.

Where possible, size and age structure were constructed either by sex or colouration. In general, colour phase is closely synchronised with sex in scarids (Choat and Robertson 1975) where all fish in the terminal phase (TP) are mature male, while most, but not all with initial phase (IP) colouration are female (Hawkins and Roberts 2003). Allocating gender based on phenotypic characteristics as an alternative to histological gonad analysis has been previously carried out on scarids (Hawkins and Roberts 2003) and will also be utilised here to draw conclusions on populations from which internal gonad examination was not possible. Given that small sample size can impact on accuracy of estimates, the oldest 15% of all datasets were used to project mean maximum size and age using a macro in Excel^{TM} developed by James Cook University graduate Elizabeth Lamantrip.

A comparison of von Bertalanffy parameters between different location was carried out using Likelihood Ratio Tests (LRTs) (Kimura 1980) where 95% confidence intervals were generated around the parameter estimates of K and $L\infty$. This analysis was achieved with an Excel^{TM} macro developed by James Cook University graduate Jake Kritzer. Non overlapping confidence regions indicated significant difference in the growth parameters being tested. The $\boldsymbol{t_0}$ value generated for each sample from VBGF analysis was used in assigning a common $\boldsymbol{t_0}$ value to all samples, which then can subsequently be used to acquire confidence interval parameters based on that shared constrained $\boldsymbol{t_0}$ value.

Instantaneous rate of mortality (Z) is calculated by adding Fishing mortality (F) with natural Mortality (M). Here I use age based curves described by Beverton & Holt (1957) and Pauly (1984) to estimate Z. The assumption in these methods is that recruitment rates remain spatially constant. Instantaneous mortality is then calculated by plotting the natural logarithm of the number of fish sampled from each age class, against their corresponding age. Comparison of mortality rates were carried out using analysis of covariance (ANCOVA) where the assumption of heterogeneity of slopes was tested in SPSS according to the procedures of Zar (1999). No attempt was made to decouple species based on colouration as Jones (Jones 1980) highlights the fact that it is not possible to distinguish between IP and TP mortality rates because it is not possible to distinguish between morality and sex change events in the initial phase for protogynous hermaphrodites such as scarids. Thus for the purpose of this study samples were pooled together to arrive at combined mortality estimate.

4.3 Results

4.3.1 Pacific Steephead parrotfish *Chlorurus microrhinos* and Indian Steephead parrotfish *Chlorurus strongylocephalus*

Figure 4.1. Pacific Steephead parrotfish *Chlorurus microrhinos*

Species complex is referred to a situation when species are reproductively isolated but are morphologically very similar. This is the case with the *Chlorurus* genus where the species complex comprise of *Chlorurus gibbus* in the Red Sea, *Chlorurus strongylocephalus* in the Indian Ocean and *Chlorurus microrhinos* in the Pacific. As such a comparative demographic analysis between these species would provide an interesting perspective between species that are ecologically very similar but phylogenetically distinct. All three representative species of the *Chlorurus* genus are reef excavators, breaking chunks of the reef matrix with their heavy beaks, and are known to reach a maximum size of 70cm (TL) (Lieske and Myer 1994). In terms of colouration, the TP phase of *C.microrhinos* occurs across the full distribution of size classes, however, I have personally witnessed small red and yellow IP phase specimens from the GBR. Although the IP/TP colouration of *C. strongylocephalus* is similar to *C.microrhinos*, the IP colour phase is more prevalent. Although *C.microrhinos* appears to be a generally a solitary species, I have observed mass spawning aggregations in Solomon Islands.

Analysis for *C.microrhinos* and *C.strongylocephalus* was based on three population samples. A total of 92 *C.microrhinos* specimens were extracted from the GBR representing a dataset from the Pacific Ocean, while 57 *C.strongylocephalus* specimens were sourced off granitic reefs from Mahe (the main Island of Seychelles), and 95 from the Island of Cocos Keeling (provided by Alec Hughes), thus representing the Indian Ocean. A handful number of other specimens were also opportunistically acquired from Solomon Islands, and also from the carbonate reefs of Seychelles' Farquhar groups of islands, 700 kilometres southwest Mahe. These individuals do not collectively form an independent sample size suitable for statistical analysis. However, they are used in a comparative attempt to ascertain tentative size at age attributes.

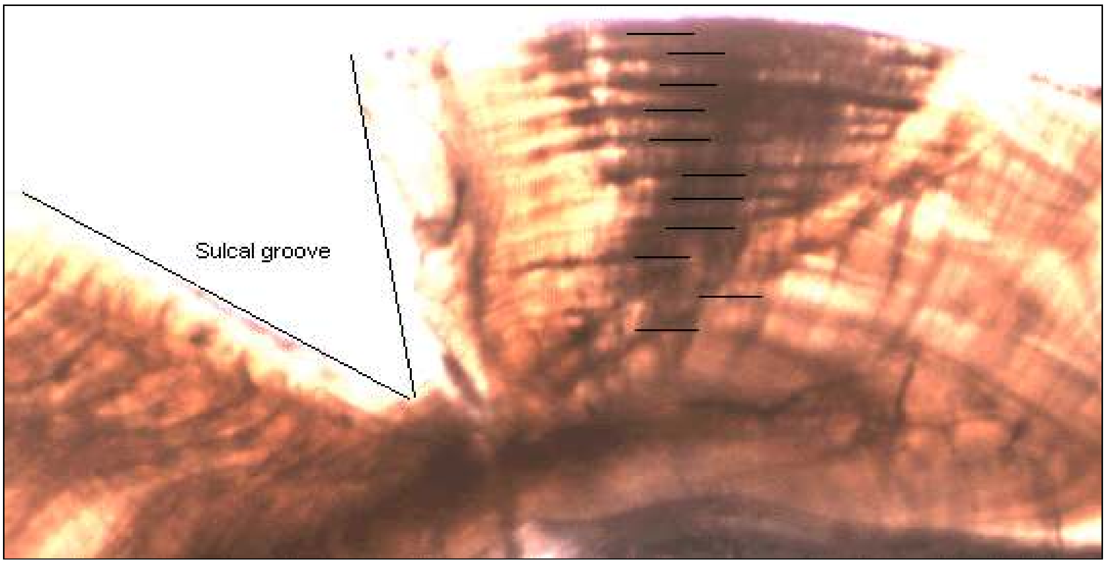

Figure 4.2. *Chlorurus Microrhinos*. Sectioned sagittal otolith of an eleven year old from the GBR.

Location	n	Size (FL mm)	L∞	K	t_0	r^2	Mean max age	Mean max Size (mm)
GBR	92	221-580	498.87	0.42	-0.096	0.55	10.0	502
Seychelles	57	166-499	447.64	0.65	-0.070	0.85	8.1	427
Cocos Keeling	95	182-462	394.90	0.59	-0.088	0.50	7.4	383

Table 4.1. Sample sizes, size ranges and VBGF parameters for three populations by location. Mean max age and size were calculated based on the oldest 15% of each population.

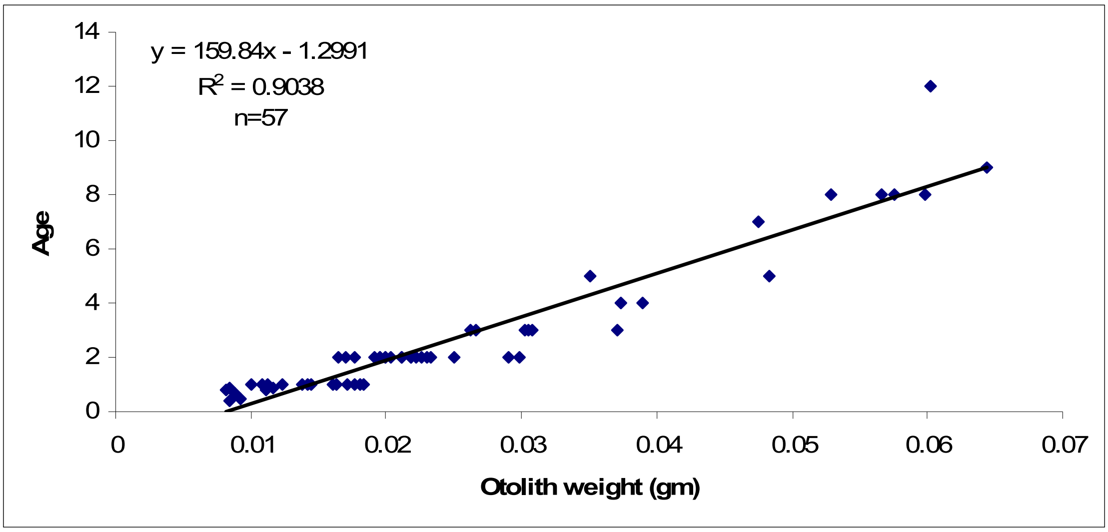

Figure 4.3. *Chlorurus strongylocephalus from* Seychelles. Regression of sagittal otolith weight vs age.

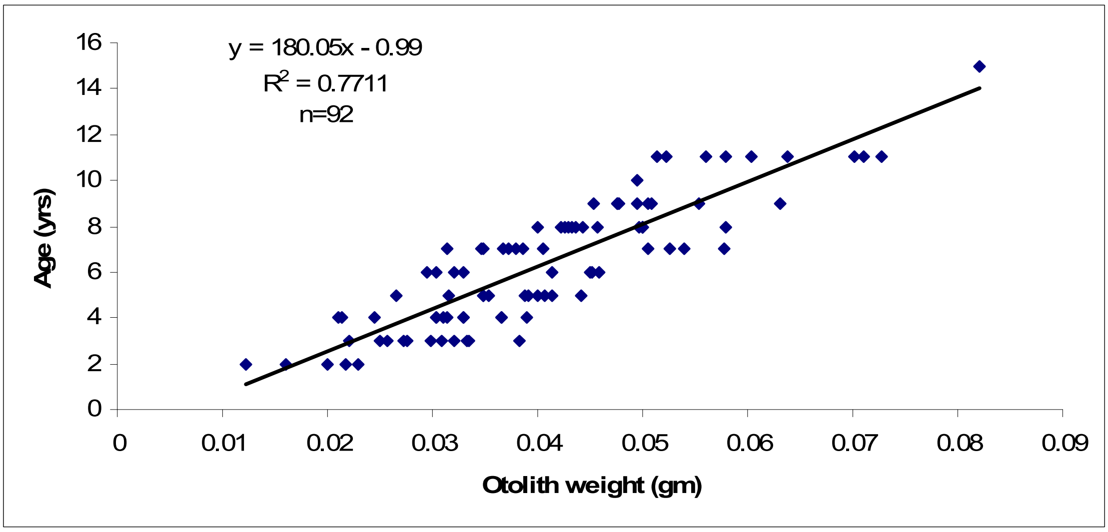

Figure 4.4. *Chlorurus microrhinos* from GBR. Regression of sagittal otolith weight vs age.

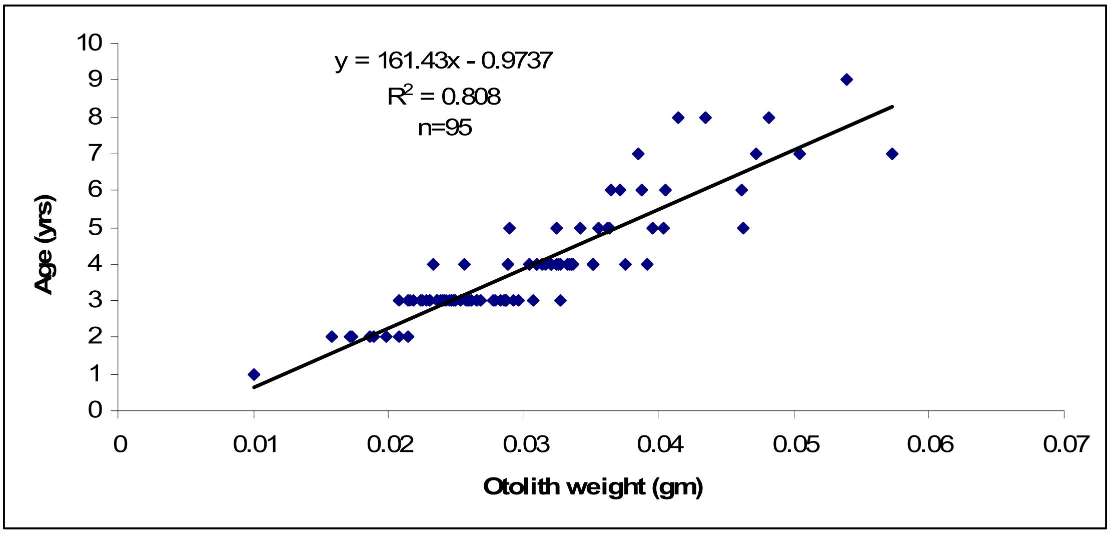

Figure 4.5. *Chlorurus strongylocephalus* from Cocos keeling. Regression of sagittal otolith weight vs age.

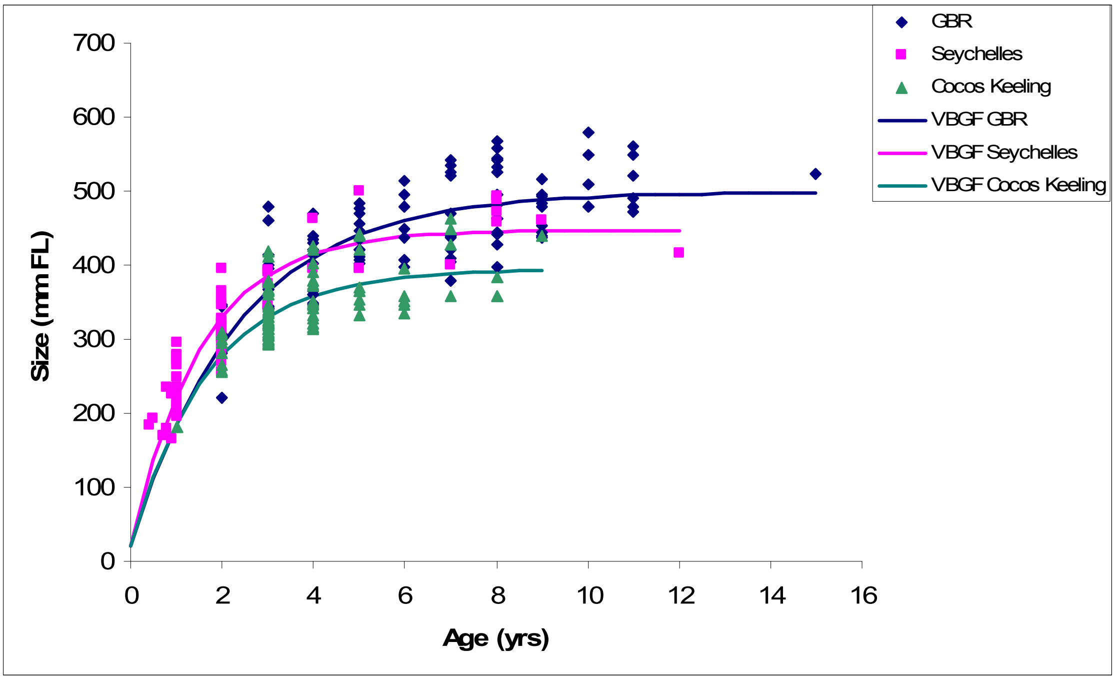

Figure 4.6. Size at age plots with fitted von Bertalanffy growth function curves for three populations. (VBGF parameters in Table 4.1).

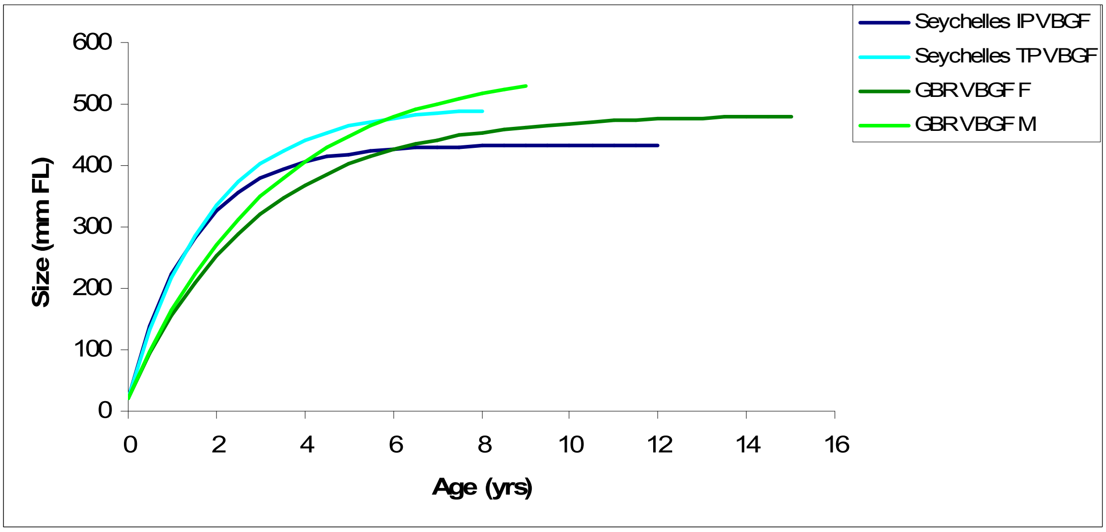

Figure 4.7. Projected VBGF growth trajectories for two populations partitioned by colour-phase and sex.

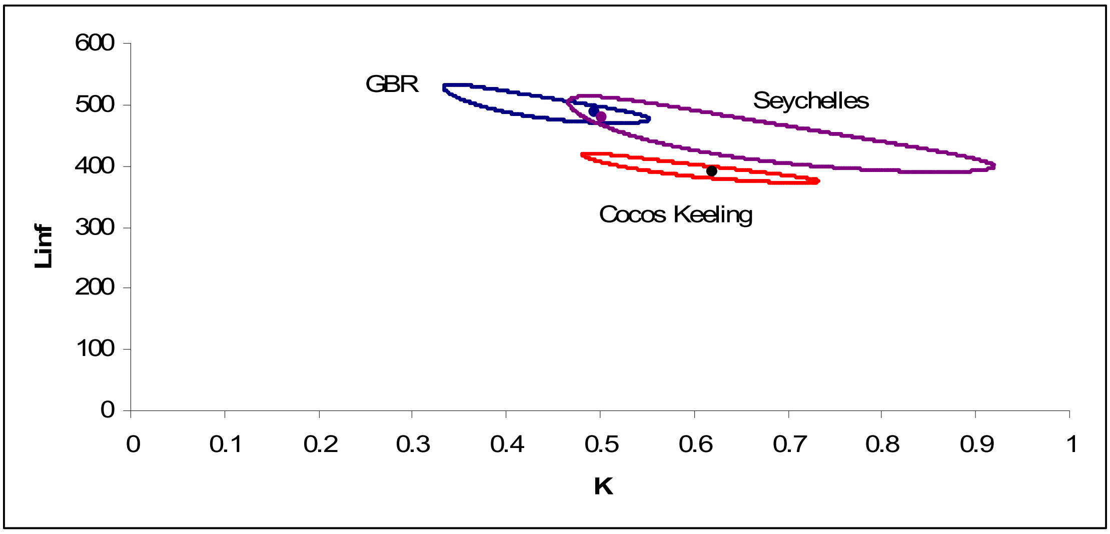

Figure 4.8. Comparison of VBGF curves for three populations showing 95% confidence regions around least squares estimates of K and L∞ following Kimura (1980). T0 for this analysis was constrained to -0.085.

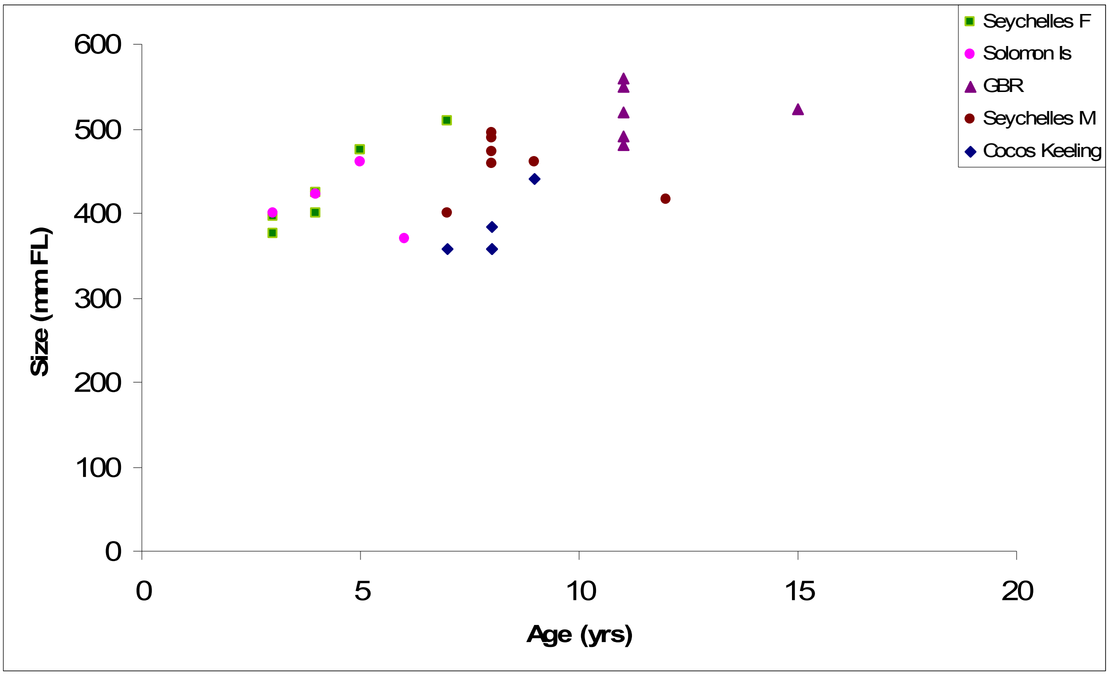

Figure 4.9. Size at age plots for oldest individuals from the three investigated regions, in addition to Solomon Islands and Seychelles Farquhar.

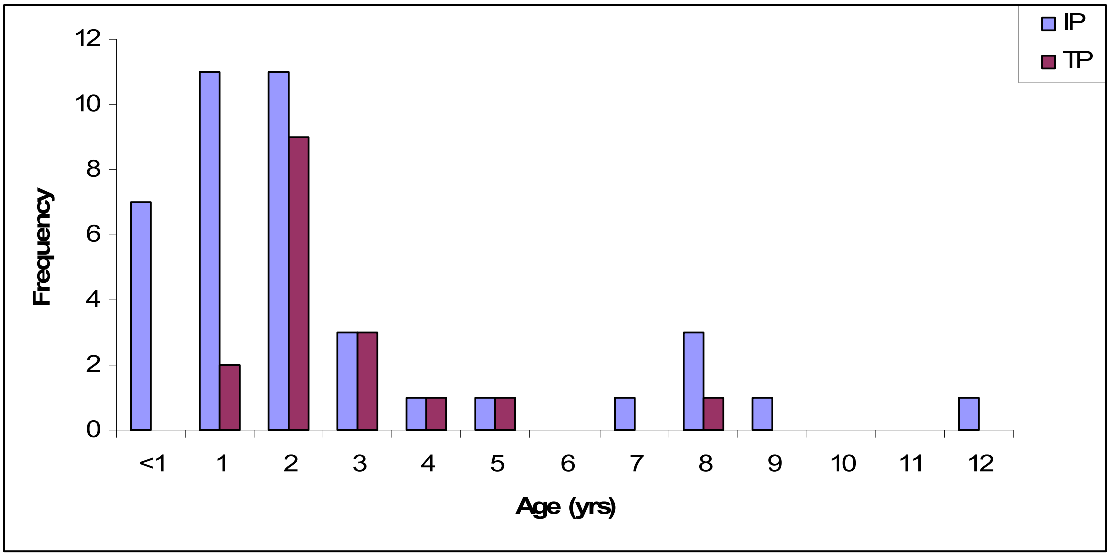

Figure 4.10. Age-frequency distribution for *Chlorurus strongylocephalus* from Seychelles.(IP=40 TP=17)

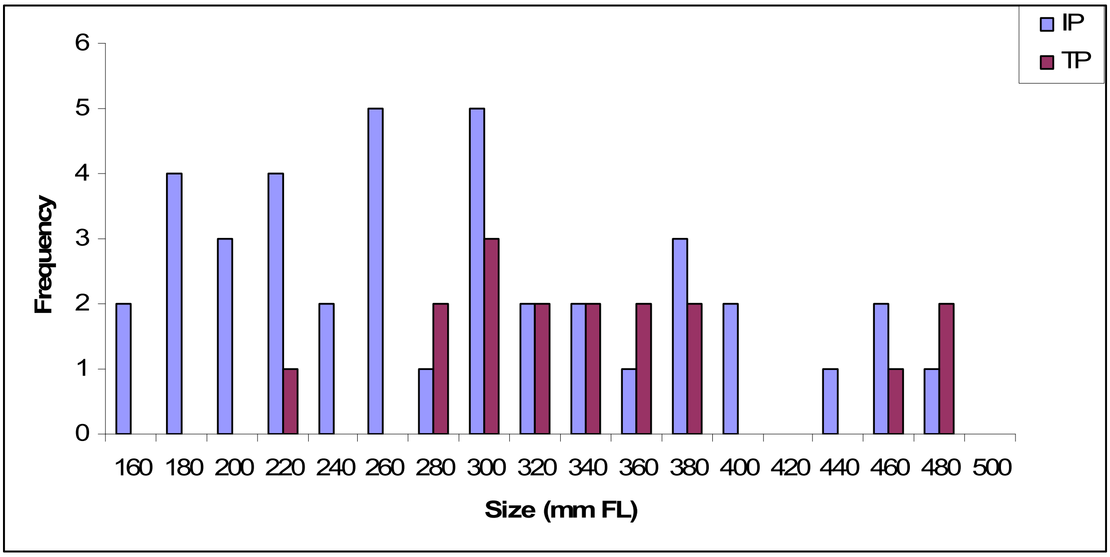

Figure 4.11. Size-frequency distribution for *Chlorurus strongylocephalus* from Seychelles. (IP= 40 TP=17)

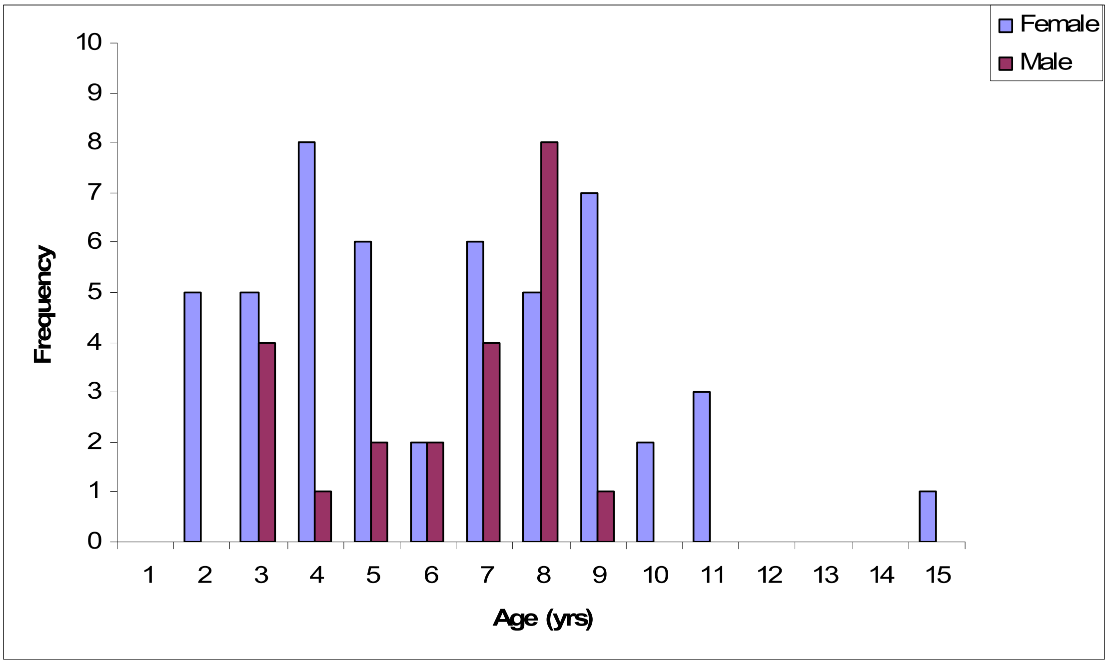

Figure 4.12. Age-frequency distribution for *Chlorurus microrhinos* from GBR. (F=50 M=22)

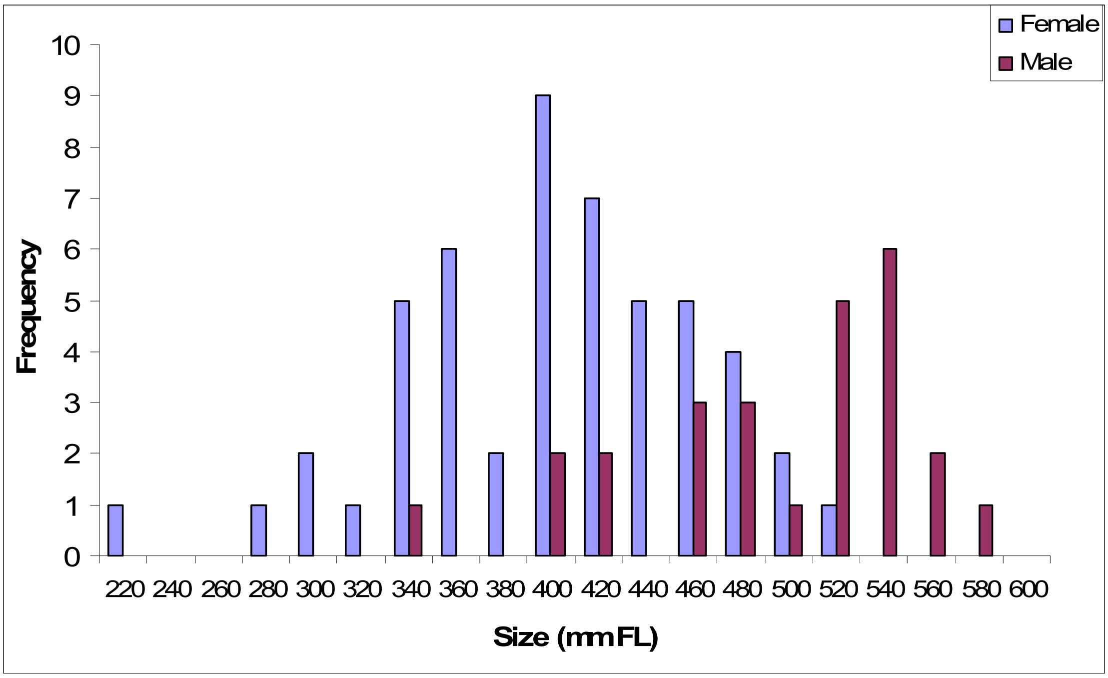

Figure 4.13. Size-frequency distribution for *Chlorurus microrhinos* from GBR. (F=51 M=26)

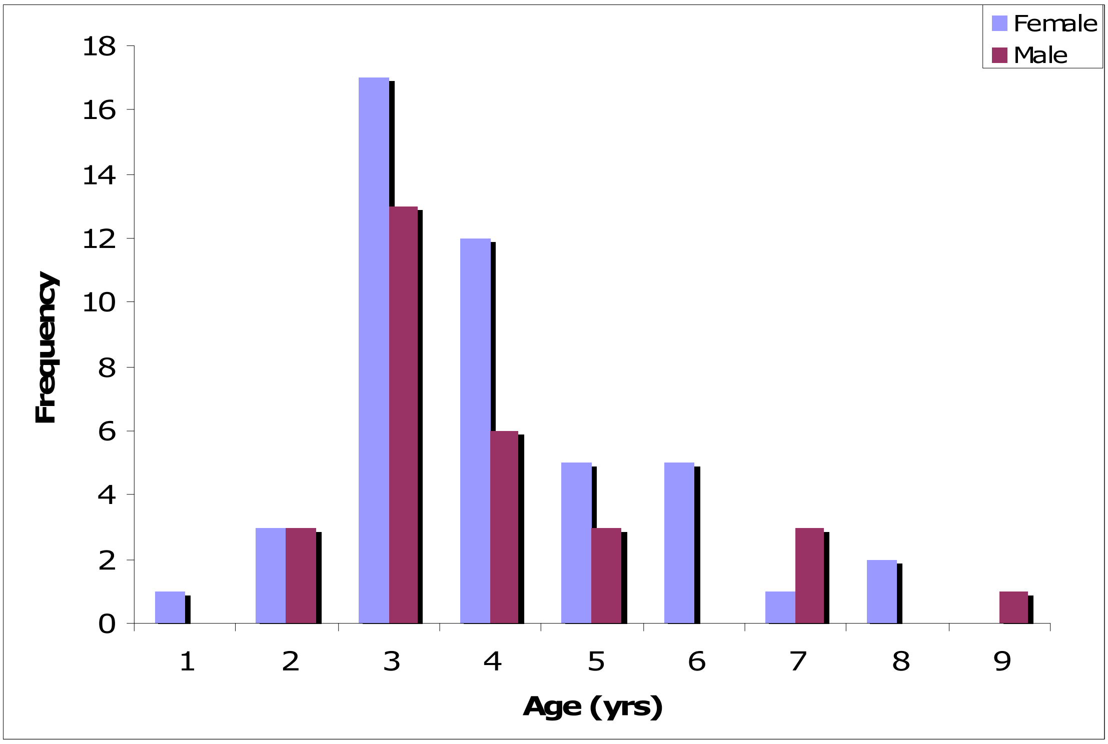

Figure 4.14. Age-frequency distribution for *Chlorurus strongylocephalus* from Cocos keeling. (F: 46 M: 29)

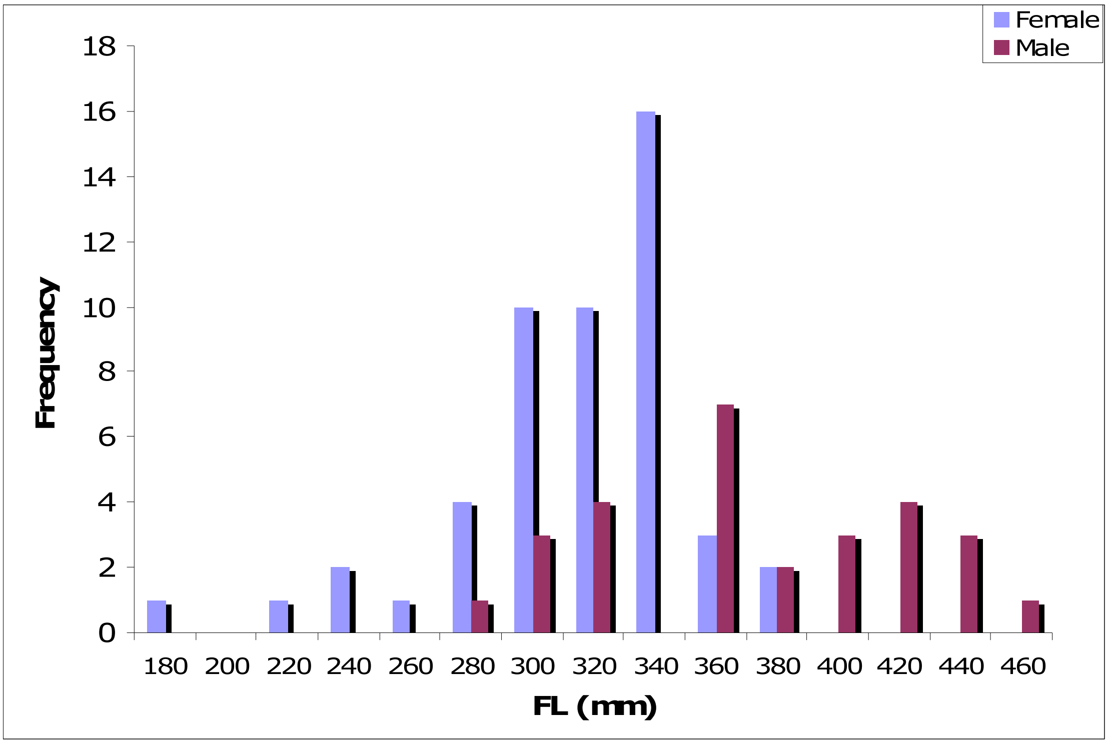

Figure 4.15. Size-frequency distribution for *Chlorurus strongylocephalus* from Cocos Keeling. (F:50 M: 28)

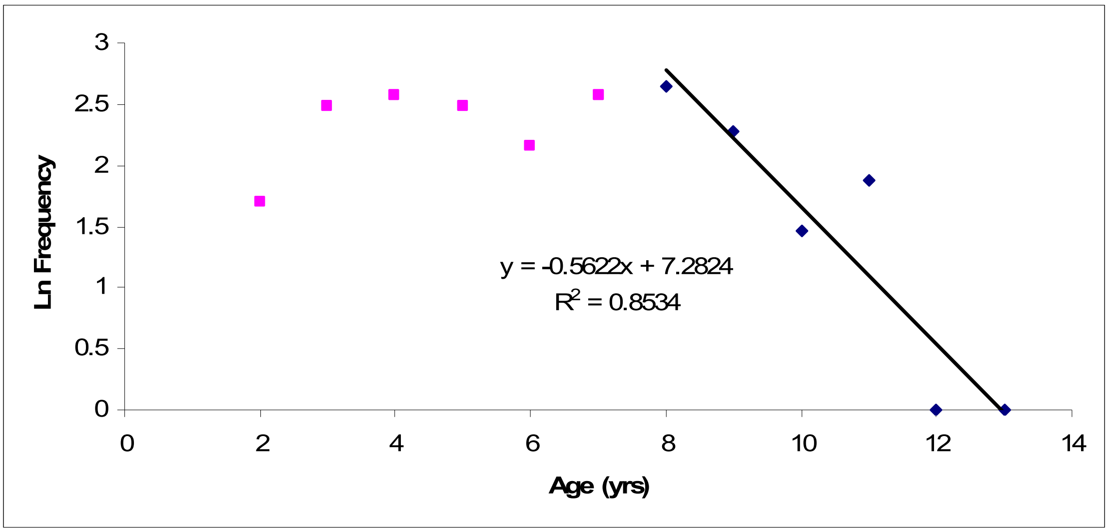

Figure 4.16. Regression of age based catch curves for estimation of *Chlorurus microrhinos* mortality from GBR.

Figure 4.17. Regression of age based catch curves for estimation of *Chlorurus strongylocephalus* mortality from Seychelles.

Figure 4.18. Regression of age based catch curves for estimation of *Chlorurus strongylocephalus* mortality from the Cocos keeling.

4.3.1.1 Summary

Sectioned sagittal otoliths examined for growth increments displayed regular opaque bands within the matrix of the otolith when viewed under transmitted light (Fig 4.2). The older individuals displayed deep sulcal groove as observed in Figure 4.2. All three populations displayed well-defined increments, especially in older individuals. Regression of otolith weight versus age portrayed a strong linear relationship with high r^2 values for all three populations. However, variability for the GBR population was slightly higher, indicative of its comparatively lower r^2 value. The reason for this may be two-fold. Firstly, accretion slows with increasing age leading to outliers (Choat et al. 2003), and secondly, because younger and smaller individuals were underrepresented the dataset is age-biased. The oldest individual in the GBR population was estimated to be 15 yrs of age, 12 yrs for Seychelles and 9 yrs for Cocos keeling. Based on values obtained from fitting von Bertalanffy growth functions to each data set (table 4.1), it appears that asymptotic length, mean maximum age, and size estimates support the notion that *C.microrhinos* attains the biggest size and oldest age in the GBR. However, this is only a tentative conclusion given that all three samples may not be sufficiently represented across all age classes.

Size at age plots allow for a comparative analysis of growth trajectories with locations (Fig 4.6). There is a lack of younger individuals in the Cocos keeling and GBR populations and older individuals in the Seychelles population. However, we can assume that all three populations appear to follow a similar growth pattern in the early life stages, although the growth trajectory for each population appears to reach asymptote at different lengths. The GBR population clearly achieves a greater size at a given age than those from Cocos keeling or the Seychelles. It is furthermore possible that growth trajectories could be affected by stages in life-history such as sexual maturation and spawning periodicity. A comparison of VBGF growth curves partitioned by sex and colouration (figure 4.7) appears to suggest that growth rates for IP and female individuals reach asymptote at a smaller size compared to TP or male samples. This sex-specific growth is of especial interest given that females appear to persist through all age classes in the populations (Figs 4.10 – 4.15).

The 95% confidence intervals plotted around the VBGF parameters of the three populations requires closer examination (Fig 4.8). Initial inspection appears to show overlapping values of K and $L\infty$ for Seychelles and GBR, indicating no significant difference in VBGF curves for the two populations. However, the dataset from the Seychelles is underrepresented in the older age classes while the GBR data set suffers from the exact opposite. The age-class bias (also discussed below in relation to mortality) in the two population samples, in conjunction with a constrained T_0 value (-0.085) required for comparative statistical analysis, may be responsible for the apparent discrepancies we observe in the confidence ellipses. Therefore to conclude no significant differences in the VBGF parameters of Seychelles and Cocos keeling populations based on the projected 95% confidence ellipses would be premature.

Size at age plots for some of the oldest individuals from the three datasets, complimented with the handful number of opportunistically acquired specimens from other locations (Fig 4.9), appear to show that largest size for equivalent age is achieved by fishes from the Solomons and the granitic reef areas of Seychelles, while longevity and ultimate maximum size appears to be occurring in the GBR, tentatively confirming the same conclusion reached above. Age frequency distribution (Figs 4.10 – 4.15) for the three populations appear to depict a similar situation whereby females and IP individuals persist through all age classes, sometimes appearing as the oldest fish in a population sample (Fig 4.12). However, size-distribution analysis suggests that maximum sizes attained by males differ from females at any given age, with males appearing to achieve larger maximum sizes than equivalent age females. Given this apparent sex-specific growth, drawing a conclusion on the age of an individual based on its size would be unwise.

Estimates of mortality acquired from regression analysis generate different R^2 values for each population suggesting that mortality is higher in Cocos Keeling, followed by the GBR and the Seychelles. Given the fact that fishing pressure on scarids is insignificant in all of these locations (pers comm. Howard Choat), total mortality (Z) should theoretically equal natural mortality (M) given that fishing mortality (F) is equal to zero. The higher mortality in Cocos Keeling could thus indicate that other factors, such as predation, may be influencing overall higher natural mortality. In fact, Gust et al (2002) state that natural fish mortality rates are more dependent upon the level of predation rather then other factors.

However, projected figures must be viewed cautiously given that they may not depict an accurate picture of morality from these locations. The reason for this is the skewed age distribution in the Seychelles and Cocos keeling populations and also the selective method with which the GBR sample was acquired (one-off spearfishing competition). Ideally, collection of specimens should be carried out repeatedly by sampling cohorts of fish through time. Choat et al., (2003) highlighted this by pointing out that in order to avoid biases through the over-selection of large fishes, they sampled individuals on a haphazard "first encounter" basis. Analysis of covariance values comparing mortality estimates reveal significant differences (ANCOVA F=9.397, P=0.001). No significant differences were found between the GBR and Cocos keeling (P=0.15), or the Cocos keeling and Seychelles (P=0.98). However, there appears to be significant difference between the GBR and Seychelles (P=0.000) in mortality.

4.3.2 Bluebarred parrotfish *Scarus ghobban*

Figure 4.19. Bluebarred parrotfish *Scarus ghobban.*

As one of the most widely circumtropically distributed parrotfish species (Lieske and Myer 1994), *S.ghobban* has a distinctive yellow IP colouration which turns into a more blue-green colour in the TP. From personal observation, IP individuals appear to swim in small to medium sized schools, while numbers in TP groups seem smaller. Known as a scraping scarid, *S.ghobban* mainly feeds on the reef surface with its small beaks, scraping up algae and detritus. A wide range of maximum sizes has been reported, but generally considered to reach its largest at 90cm (TL) in the Indian ocean (fishbase.org). Personal observation, both from fish markets and on the reefs, appears to suggest that maximum TL in the Solomon Islands for this species are markedly smaller than their Indian ocean counterparts. Observation of demographic records from other locations from the Pacific region (including in this chapter) appear to support this observation.

Analysis for this species was based on two population samples. In total, 92 specimens were extracted from Taiwan and 60 from Solomon Islands. The datasets enabled me to conduct a comparative analysis of two latitudinally separate populations from the same ocean system. Both populations were partitioned by colouration for describing growth, size and age distribution. Small number of other specimens were also acquired from Oman, Panama, Western Australia and the Seychelles (Granit reefs of Mahe). These individuals did not collectively form an independent sample size suitable for statistical analysis. However, they are used in a comparative attempt to infer possible size at age attributes.

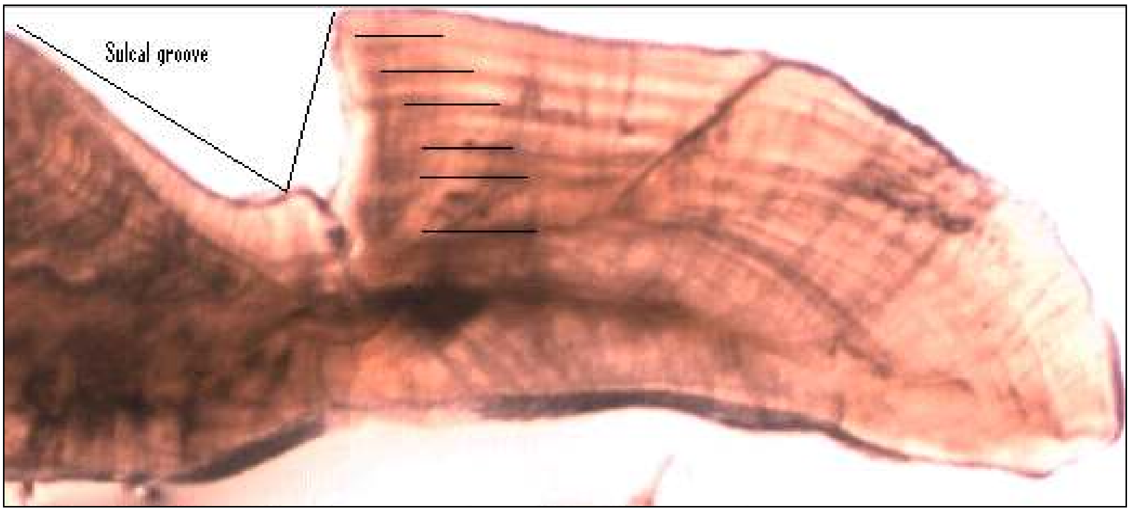

Figure 4.20. *Scarus ghobban*. Sectioned sagittal otolith of a six year old from Taiwan.

Location	**n**	**Size (SL mm)**	**L∞**	**K**	**t_0**	**r^2**	**Mean max age**	**Mean max Size**
Solomon Is	60	205-350	297.3	1.59	-0.04	0.52	3.44	298.33
Taiwan	92	161-523	428.7	0.65	-0.07	0.73	6.00	471.14

Table 4.2. Sample sizes, size ranges and VBGF parameters for two populations by location. Mean max age and size were calculated based on the oldest 15% of each population.

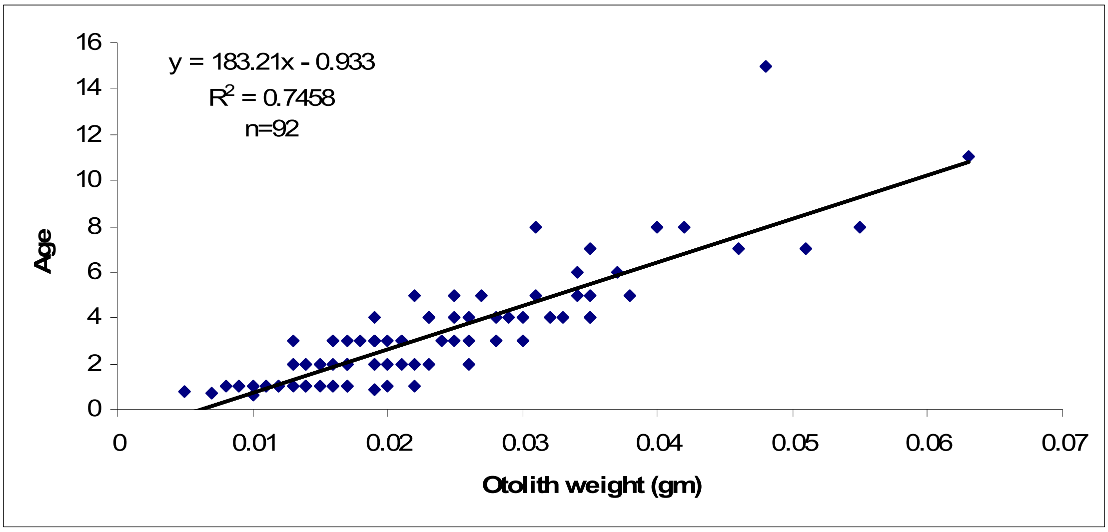

Figure 4.21. *Scarus ghobban* from Taiwan. Regression of sagittal otolith weight vs age. Outlier is a 15 year old female (Age=15 IP, SL=436)

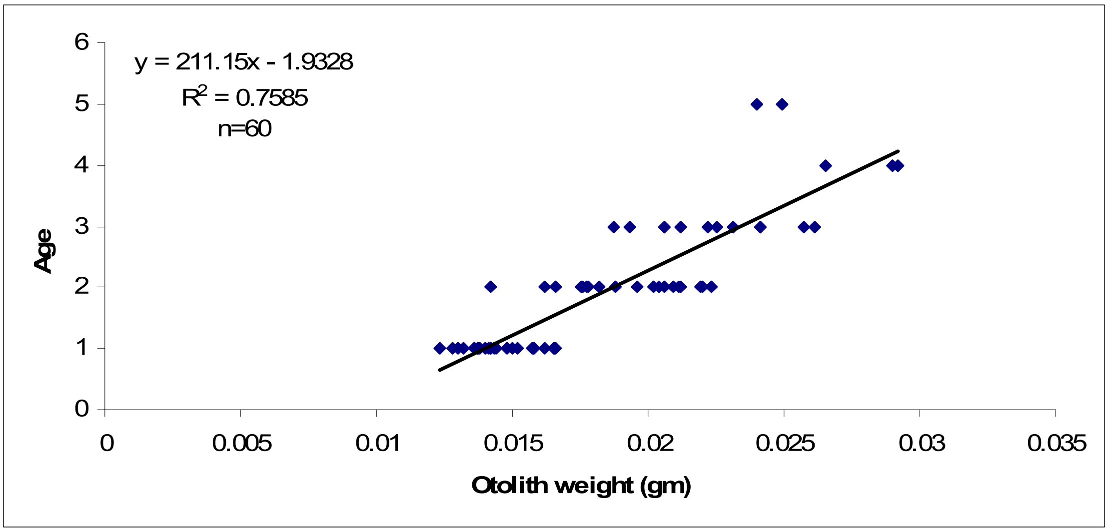

Figure 4.22. *Scarus ghobban* from Solomon Islands. Regression of sagittal otolith weight vs age. Outliers are two 5 year old males (Regression without outliers =0.8079)

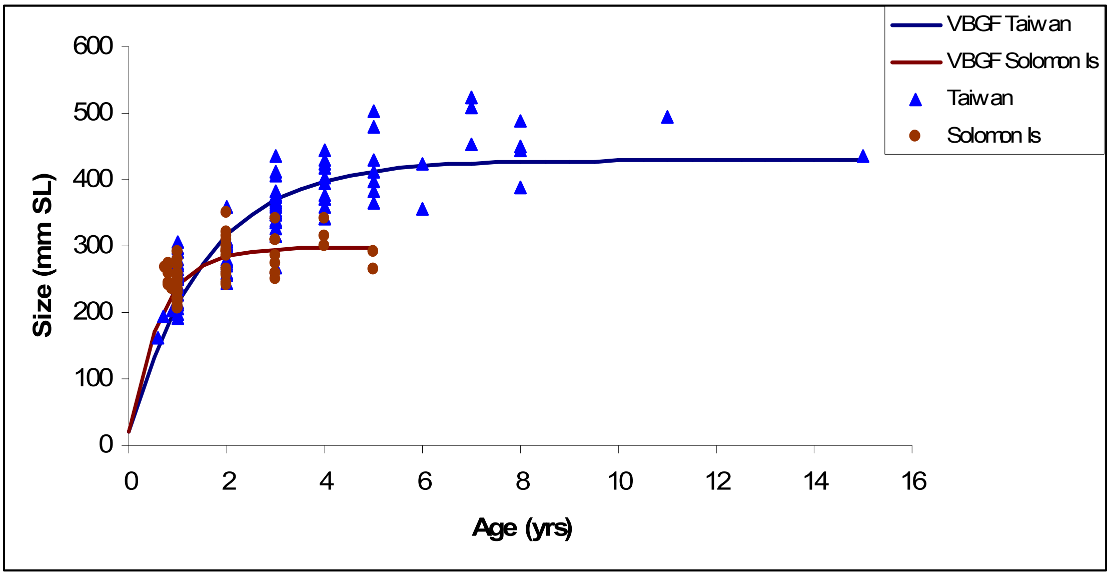

Figure 4.23. Size at age plots with fitted von Bertalanffy growth function curves for *Scarus ghobban* populations from Taiwan and Solomon Islands. (VBGF parameters in Table 4.2)

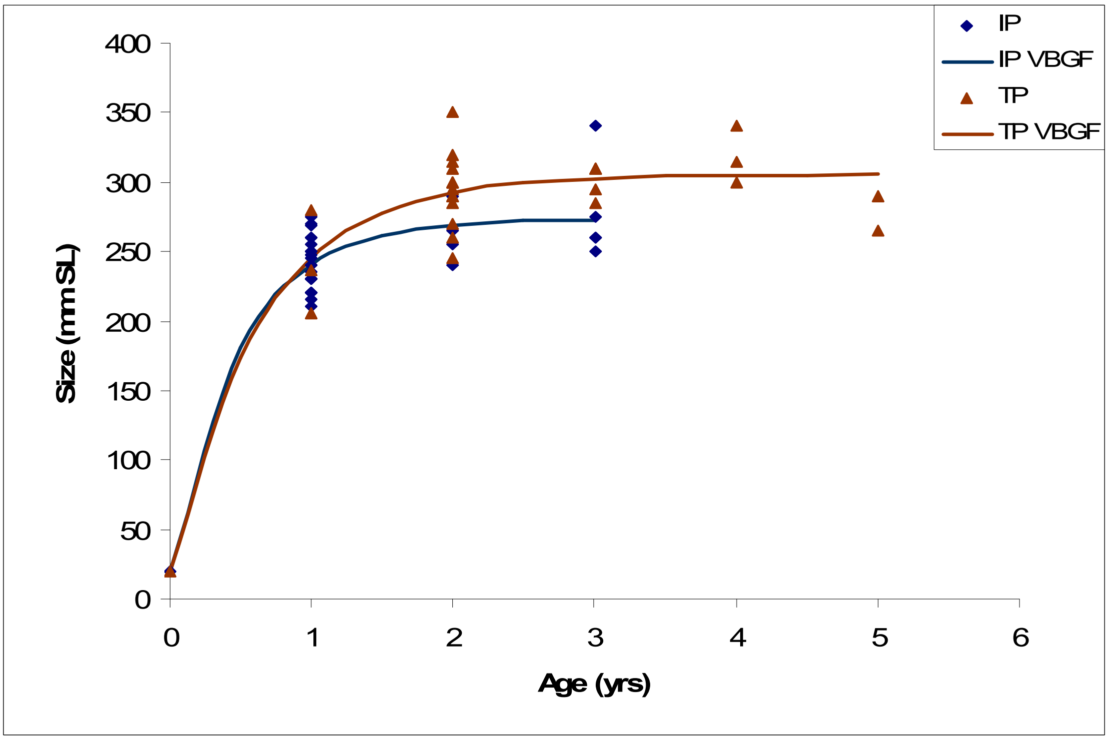

Figure 4.24. Size at age plots with fitted von Betalanffy growth function curves partitioned by colouration for *Scarus ghobban* from Solomon Islands.

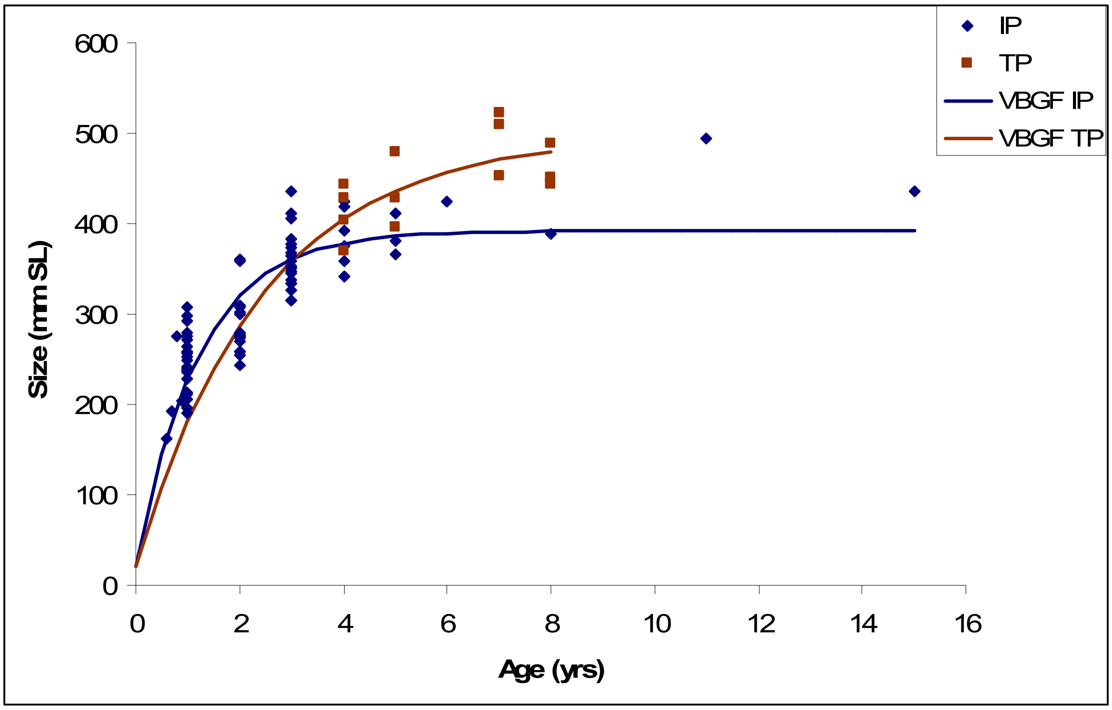

Figure 4.25. Size at age plots with fitted von Betalanffy growth function curves partitioned by colouration for *Scarus ghobban* from Taiwan.

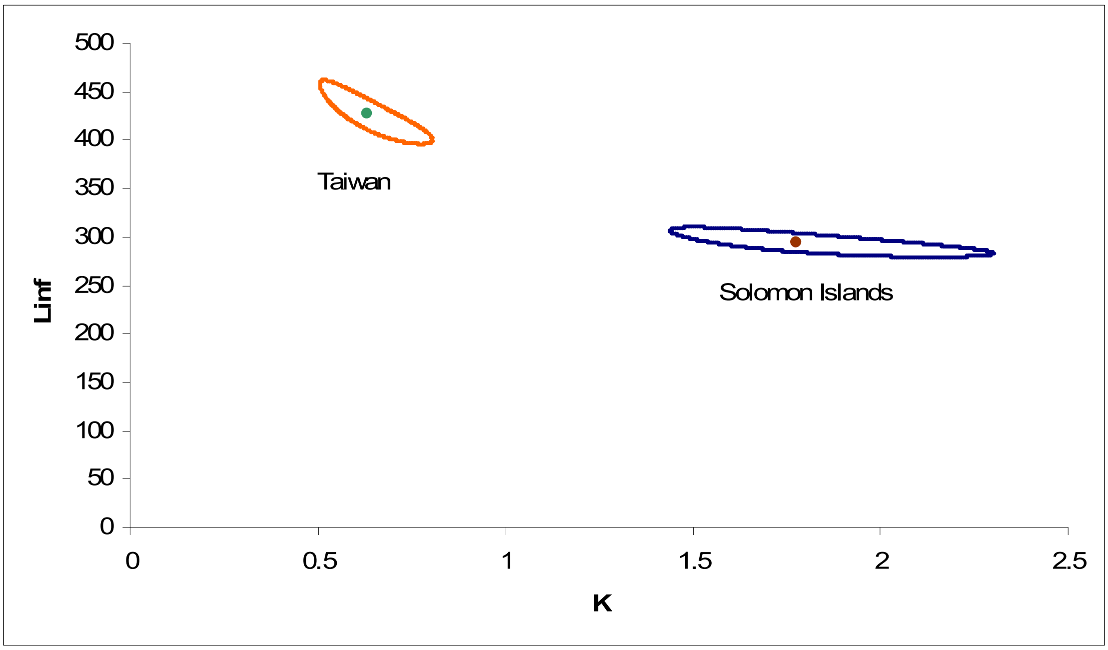

Figure 4.26. Comparison of VBGF curves for two populations showing 95% confidence regions around least squares estimates of K and L∞ following Kimura (1980). T0 for this analysis was constrained to -0.055.

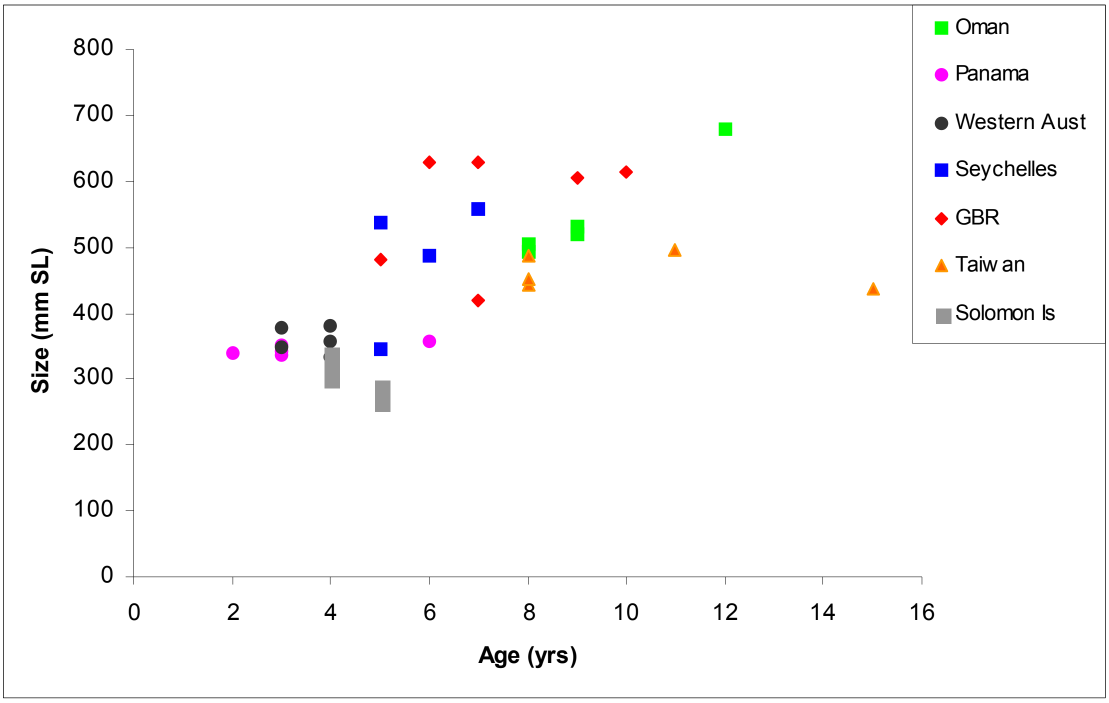

Figure 4.27. Size at age plots for small number of oldest individuals from seven regions.

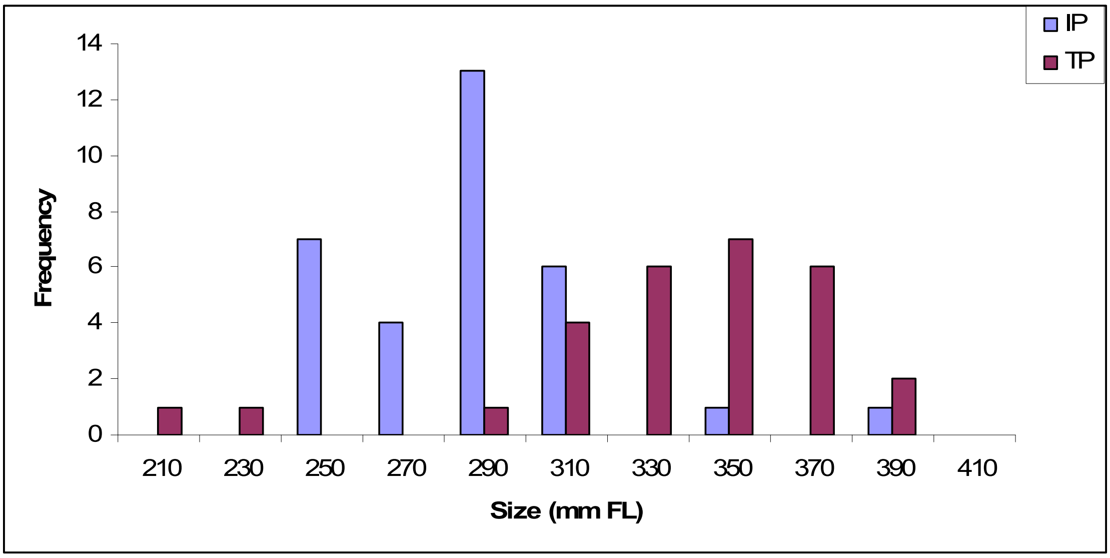

Figure 4.28. Size-frequency distribution for *Scarus ghobban* from Solomon Islands. (IP=32 TP=28)

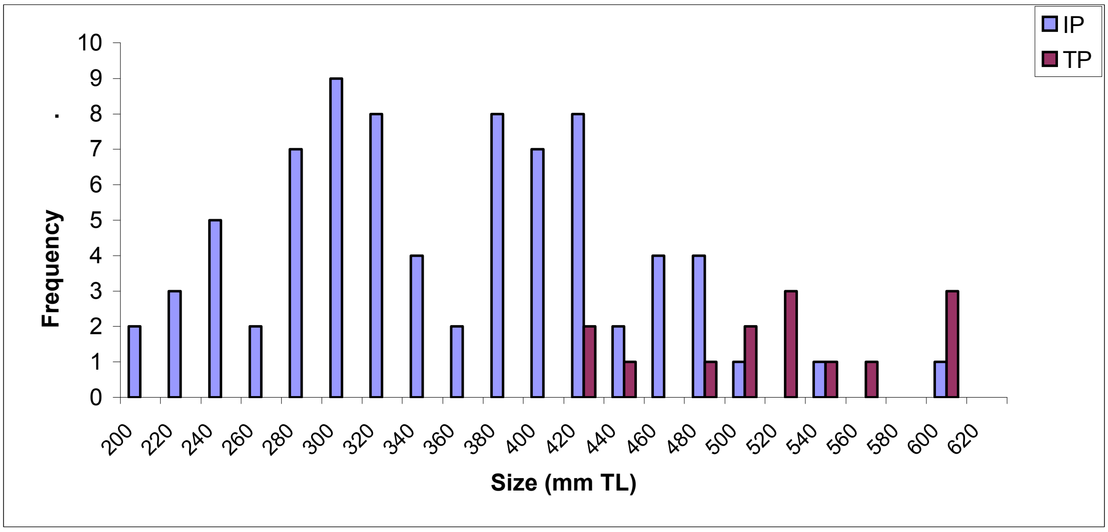

Figure 4.29. Size-frequency distribution for *Scarus ghobban* from Taiwan. (IP=78 TP=14)

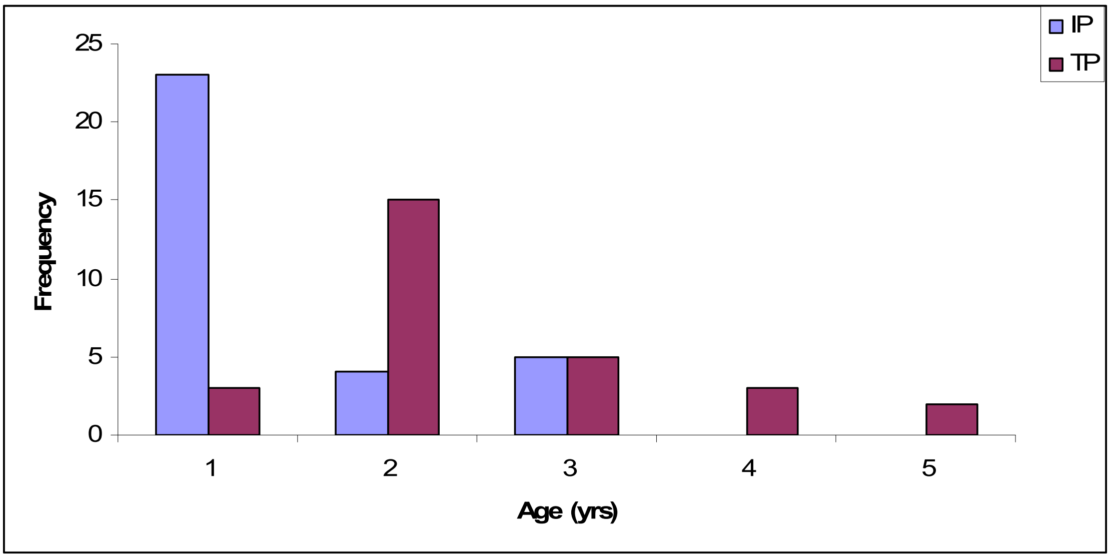

Figure 4.30. Age-frequency distribution for *Scarus ghobban* from Solomon Islands. (IP=32 and TP=28)

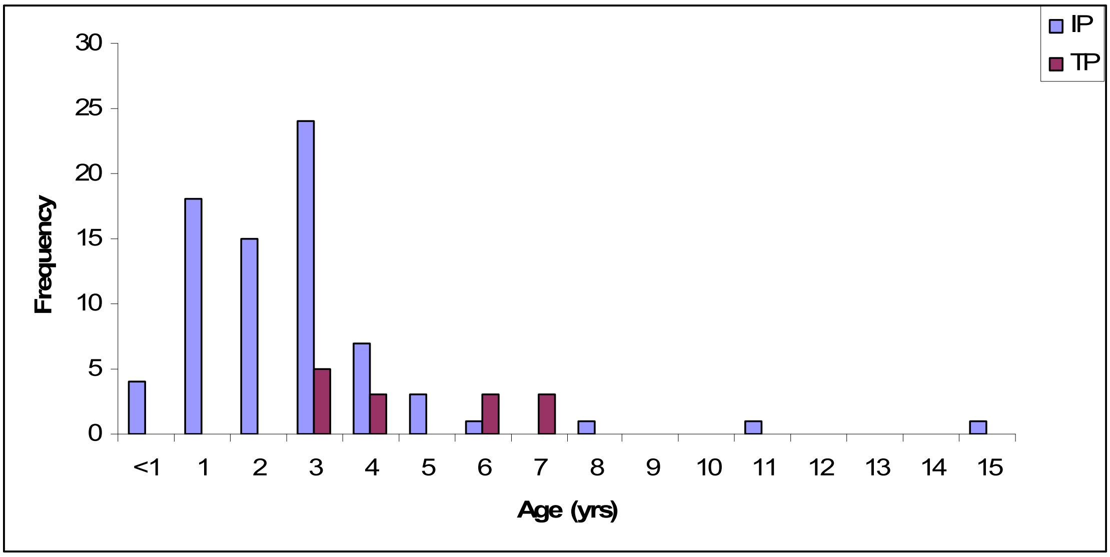

Figure 4.31. Age-frequency distribution for *Scarus ghobban* from Taiwan. (IP=75 TP=14)

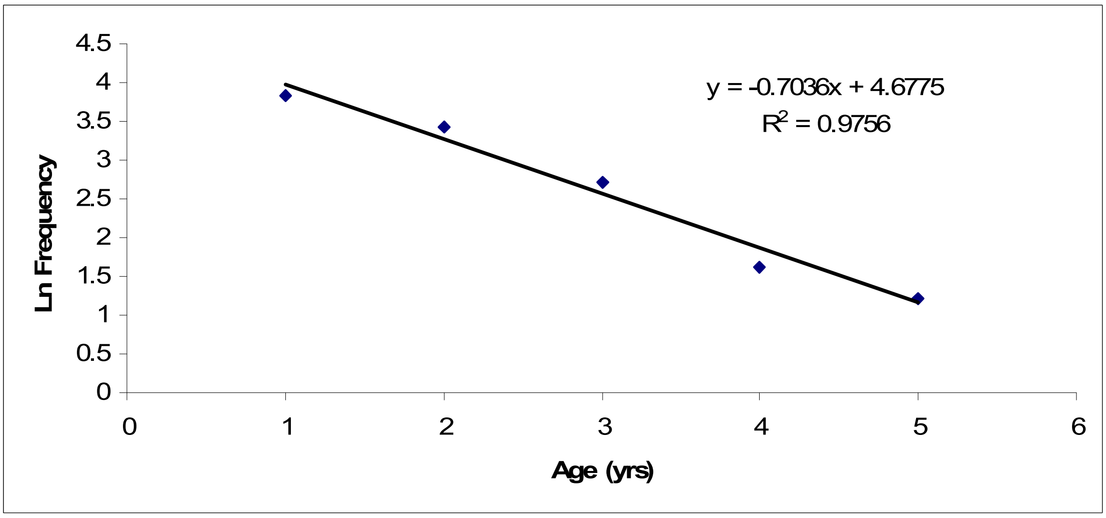

Figure 4.32. Regression of age based catch curves for estimation of *Scarus ghobban* mortality from Solomon Islands.

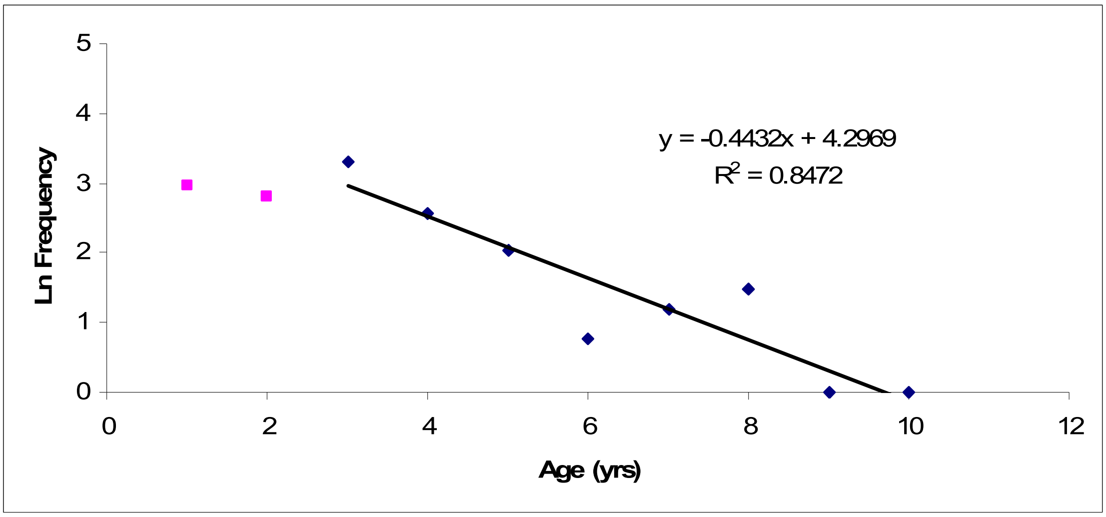

Figure 4.33. Regression of age based catch curves for estimation of *Scarus ghobban* mortality from Taiwan.

4.3.2.1 Summary

Sectioned sagittal otoliths examined for growth increments displayed regular opaque bands when viewed under transmitted light (Figure 4.20). Both populations displayed well-defined increments, especially in older individuals. Regression analysis of otolith weight versus age revealed expected linear relationship with statistically strong values. The Taiwan dataset contained two outliers which consisted of older females with the ages of 8 and 15, respectively. If these two samples were removed the r^2 would improve significantly (R^2=0.8097). However, the presence of older females persisting through age classes is an important issue and must not be overlooked. The dataset for Solomon Islands also had two outliers which reduced its r^2 value from 0.8079 to 0.7585. Overall the r^2 values for both data sets are statistically significant, but it is possible (but unlikely) that increments may not have been read accurately in some instances. There was significant difference in the longevities of the two datasets with the oldest individuals from Taiwan being estimated at fifteen yrs of age, while the Solomons could only yield two five year olds. Based on values obtained from fitting von Bertalanffy growth functions to each data set (Table 4.2) it appear that there are marked differences in population parameters. The length at which each population reaches asymptote is significantly different with the Taiwan dataset displaying a much larger L∞. This infers that latitudinal difference in growth trajectories of *S.ghobban* are significantly different between Solomon Islands and Taiwan. Mean maximum age and size, which were calculated based on the oldest 15% from each population, is significantly bigger for the Taiwan dataset. This reinforces the view that based on age estimates longevity is higher for *S.ghobban* in Taiwan than in the Solomon Islands.

Comparisons of estimated growth trajectories (Figure 4.23) confirm differences in age at size parameters . Although both populations appear to enjoy relatively similar size at equivalent age in the earlier life stages, clearly the difference is more pronounced in the older ages. However, this comparison is somewhat hampered by the apparent lack of sufficient older individuals in the Solomons dataset.

As expected, size at age plots partitioned by colouration (Figure 4.24 - 4.25) confirms difference in sex-specific growth by showing that terminal phase individuals generally grow larger and reach asymptote at a bigger size. However, when examined closely both datasets reveal interesting characteristics. Firstly, in the Taiwan dataset there is a total absence of TP individuals below the age of four, while the two oldest individuals are large females. A large female outlier is also present in the Solomons population (3 yrs old) pointing to the persistence of females through older age classes. Secondly and most importantly, unlike the Taiwan population, TP individuals appear very early in the younger age classes from the Solomons.

The 95% confidence ellipses fitted around the VBGF parameters (Figure 4.26) confirm significant differences in the growth trajectories of the two populations. Size at age plots for the small number of other opportunistically acquired samples (Figure 4.27) reveal that longevity may be significantly higher in the GBR, Taiwan and Oman. It also appears that *S.ghobban* from the GBR may be much bigger in size at equivalent age compared with the other locations.

Size and age frequency distribution (Figure 4.28 – 4.31) show that both colour phases appear in the larger sizes, although clearly TP individuals appear in much smaller size categories in the Solomons. Age frequency distribution is particularly interesting with IP individuals appearing in older age classes, with the Taiwan data boasting the oldest individual in the entire population sample. Clearly, size at age data based on colouration is complex, thus assuming age on the basis of size would not be prudent.

Estimates of mortality acquired from regression analysis generate similar R^2 values for each population. Analysis of covariance confirms no significant difference in mortality rates between the two regions (ANCOVA F= 2.741, P=0.129). This analysis may be unreliable, but given the relative high fishing pressure on Gizo reefs (established in chapter two) and Taiwan (Burke et al., 2002), it is possible that mortality rates are similar in both regions.

4.3.3 **Redlip parrotfish *Scarus rubroviolaceus***

Figure 4.34. Redlip parrotfish *Scarus rubroviolaceus.*

Although *Scarus rubroviolaceus* occurs from eastern Africa to Panama, it is not found in the Atlantic and thus not circumtropical. With its distinctive bumpy forehead, TP colouration is green with red stripes around its mouth, while IP individuals are brown. Associated with coral reefs as scraping species, they have been reported to reach a maximum size of 70 cm (TL) (fishbase.org). Personal observation from Solomon Islands revealed that *S.rubroviolaceus* appear not to swim in large groups, but do use coral trees, holes and crevices as sleeping sites along with other parrotfish species.

Analysis of this species was based on three population samples. The first population consisted of 39 individuals from Taiwan, while the other population samples were from the Seychelles. However, this dataset was partitioned by location whereby the 35 individuals that were extracted off the coast of Mahe (hereafter referred to as Seychelles M) were analysed separately from 34 individuals that were collected from the carbonate reefs of Seychelles' Farquhar group (hereafter referred to as Seychelles F). A handful number of specimens from the GBR, Oman and Solomon Islands have also cautiously been used to compare size at age estimates.

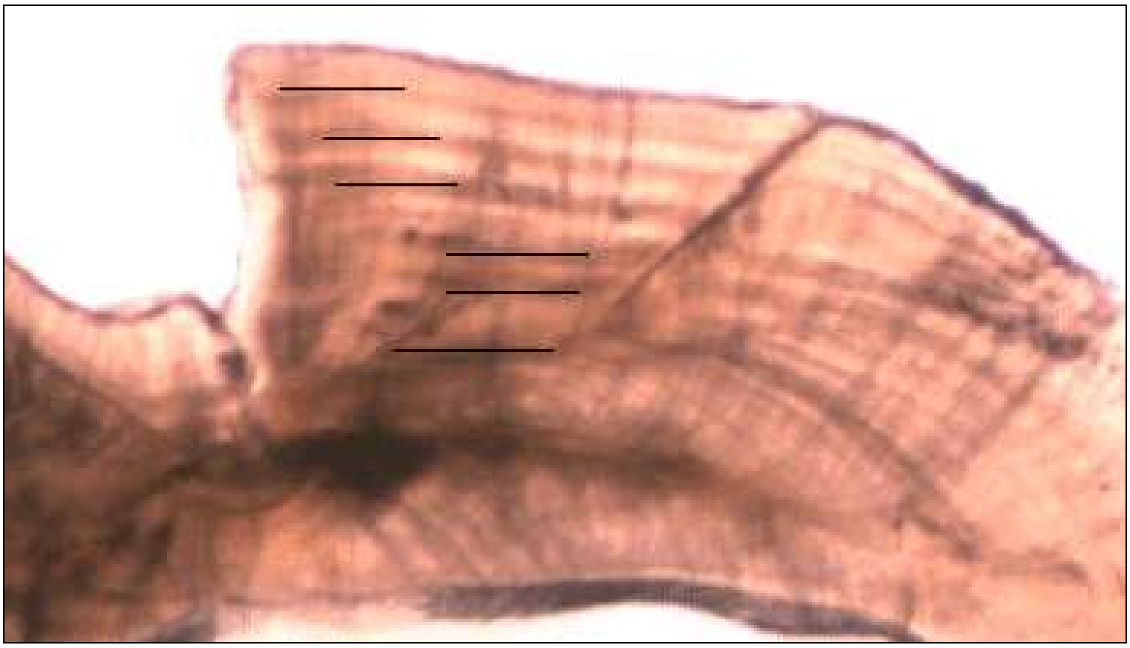

Figure 4.35. *Scarus rubroviolaceus*. Sectioned sagittal otolith of a six year old from Taiwan.

Location	n	Size (FL mm)	L∞	K	t_0	r^2	Mean max age	Mean max size
Taiwan	39	206-487	405.4	1.05	-0.04	0.74	5.3	392.59
Seychelles M	35	194-484	385.8	0.96	-0.06	0.74	7.11	402.33
Seychelles F	34	191-621	532.68	0.56	-0.07	0.78	10.67	500.00

Table 4.3. Sample size, size ranges and VBGF parameters for two populations by location. Mean max age and size were calculated based on the oldest 15% of each population.

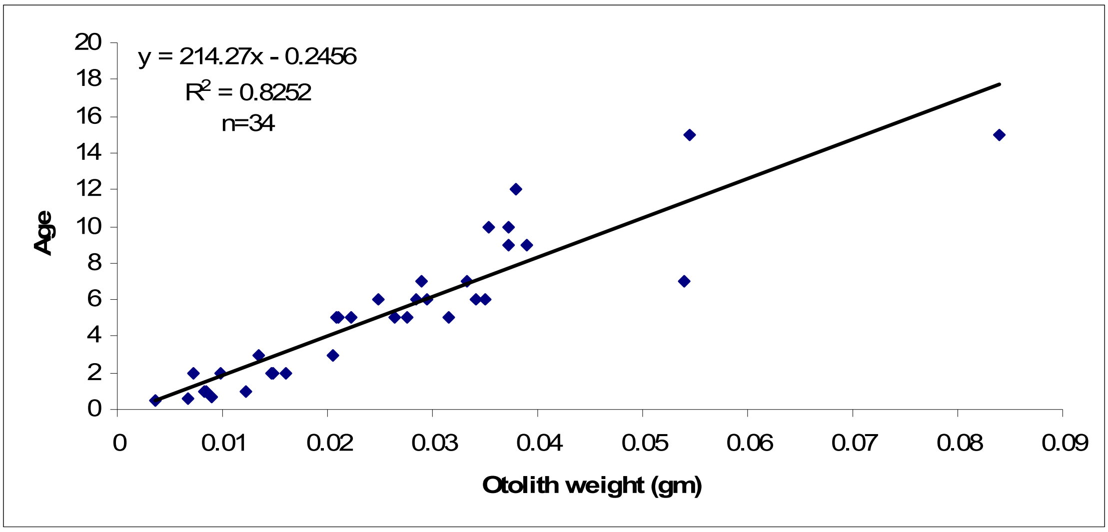

Figure 4.36. *Scarus rubroviolaceus* from Seychelles F. Regression of sagittal otolith weight vs age.

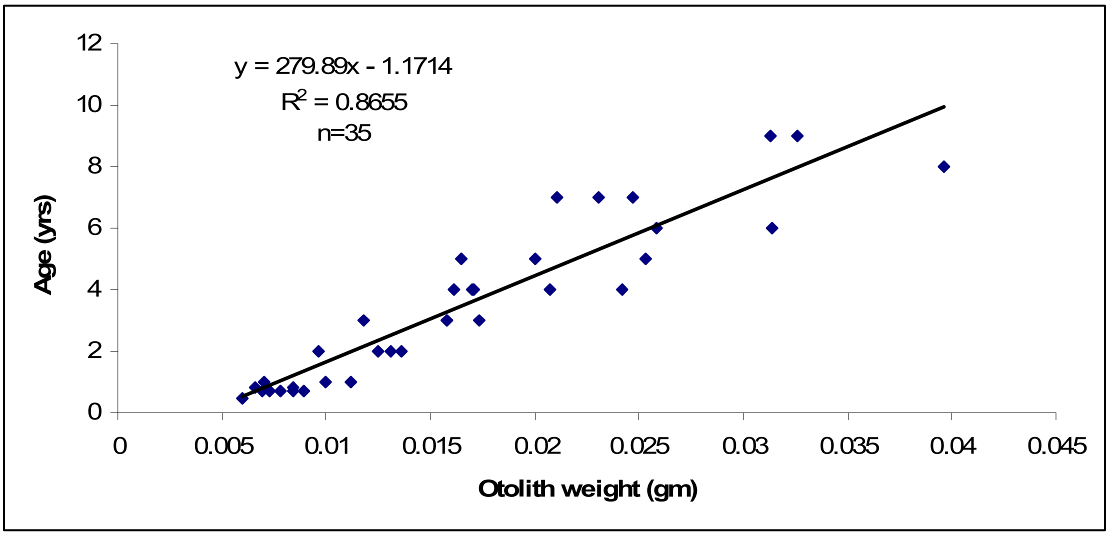

Figure 4.37. *Scarus rubroviolacues* from Seychelles M. Regression of sagittal otolith weight vs age.

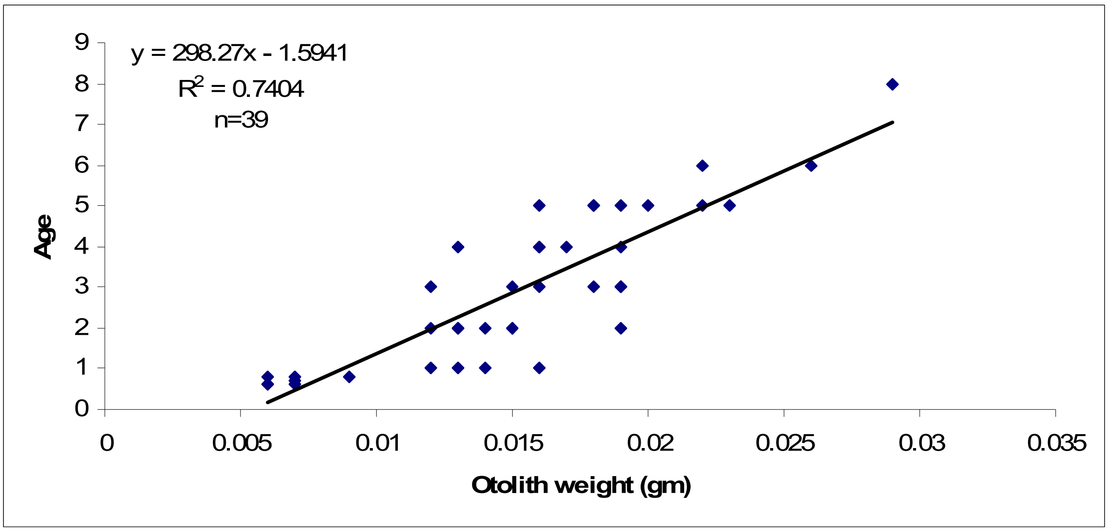

Figure 4.38. *Scarus rubroviolaceus* from Taiwan. Regression of sagittal otolith weight vs age.

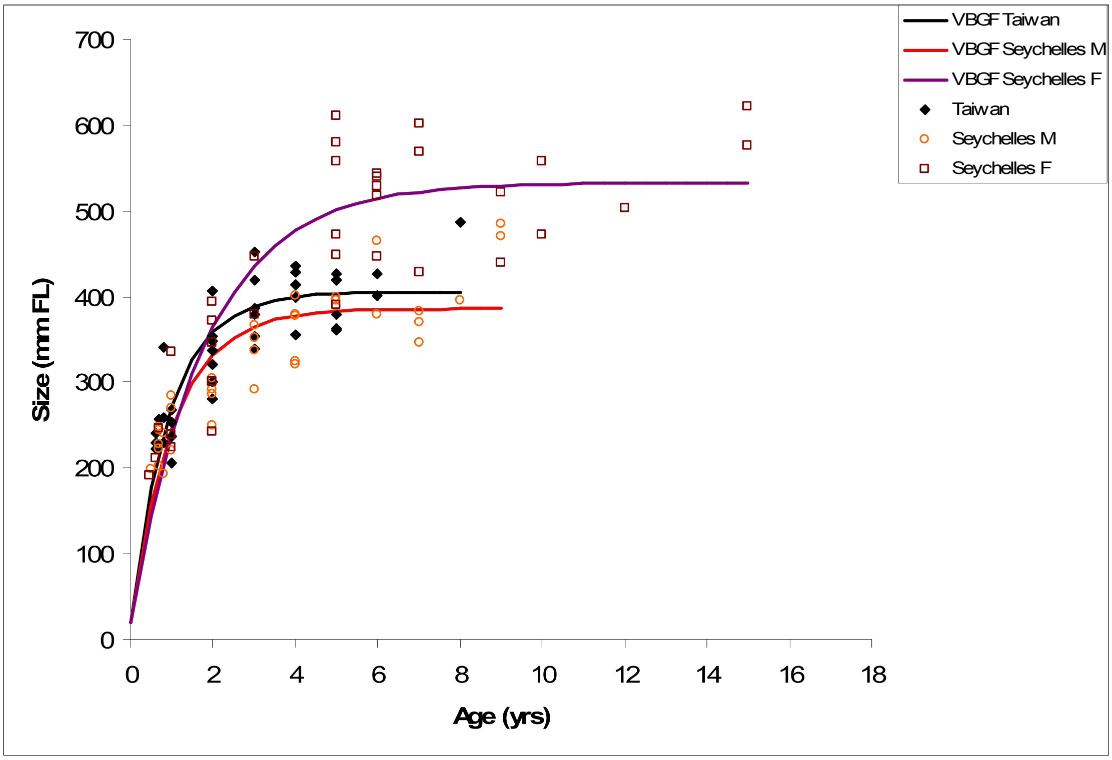

Figure 4.39. Size at age plots with fitted von Bertalanffy growth function curves for populations from three locations. VBGF parameters in Table 4.3.

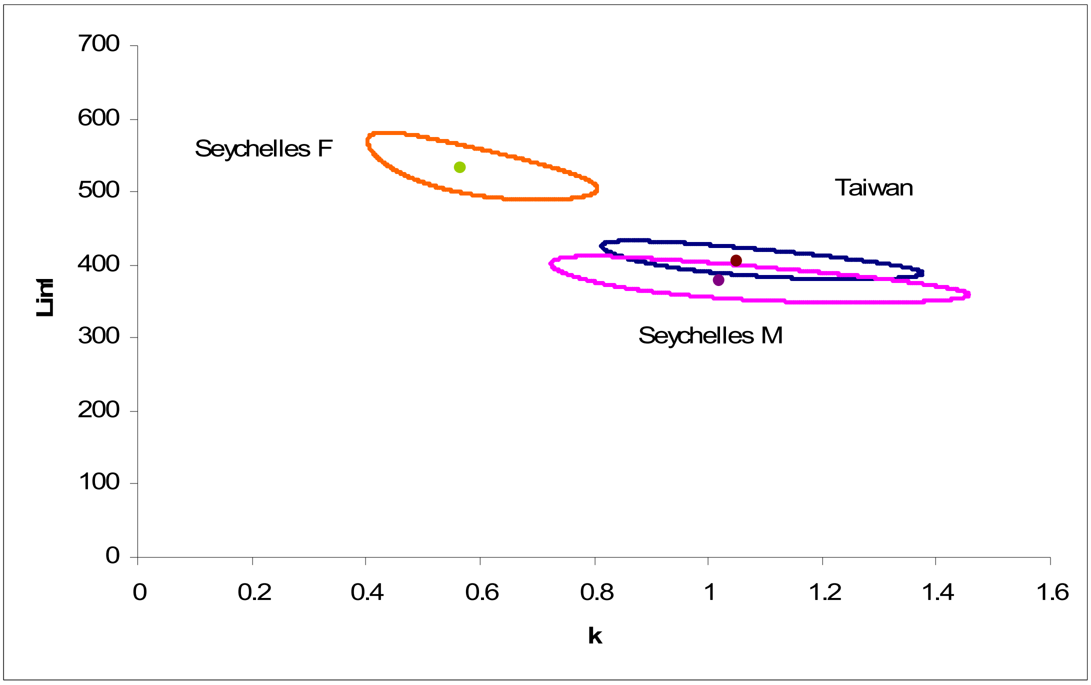

Figure 4.40. Comparison of VBGF curves for three populations showing 95% confidence regions around least squares estimates of K and L∞ following Kimura (1980). T0 for this analysis was constrained to -0.053.

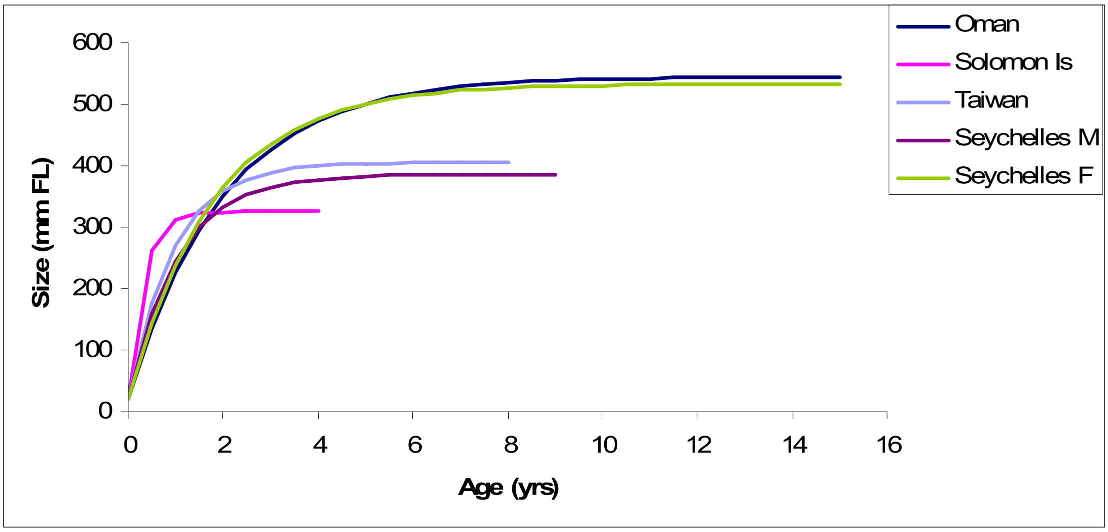

Figure 4.41. Comparative projections of VBGF growth trajectories of Oman (n=25) and Solomon Islands (n=18) versus the other three populations.

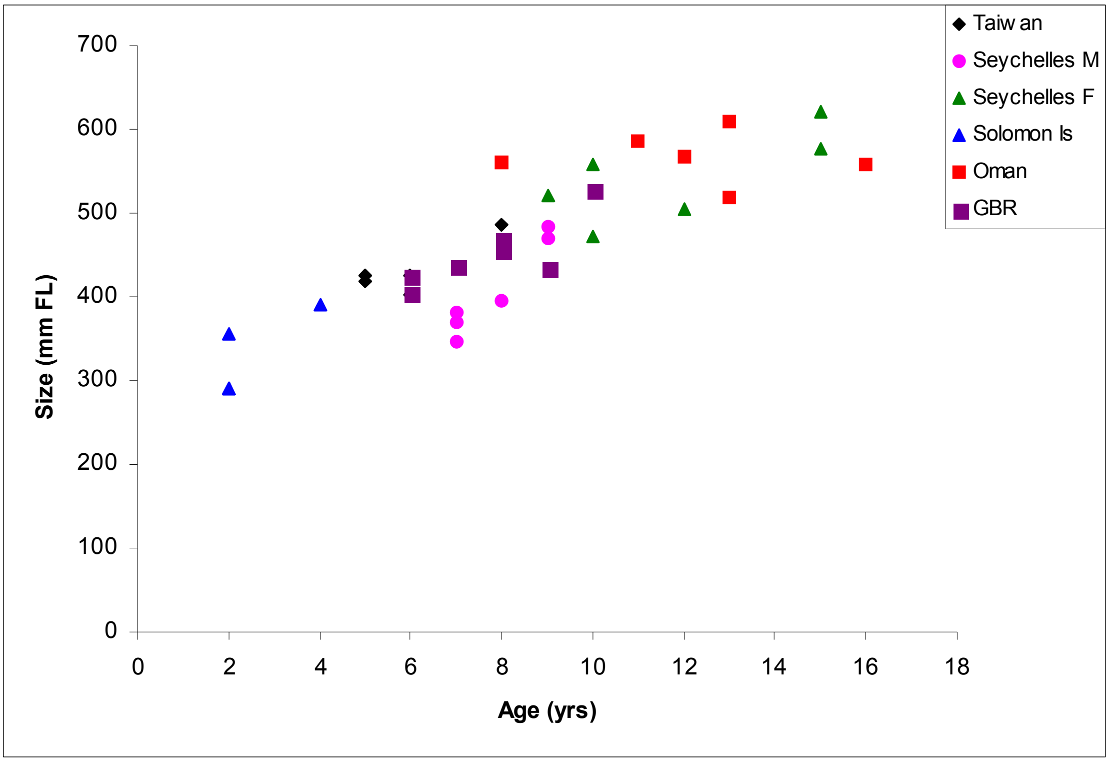

Figure 4.42. Size at age plots for small number of oldest individuals from six regions.

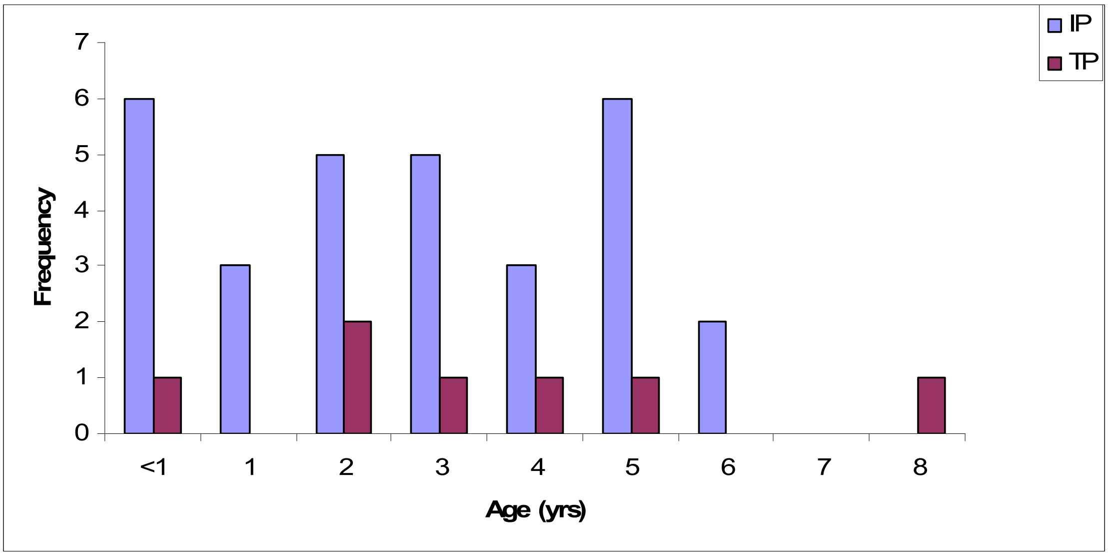

Figure 4.43. Age-frequency distribution for *Scarus rubroviolaceus* from Taiwan. (IP=30 TP=7)

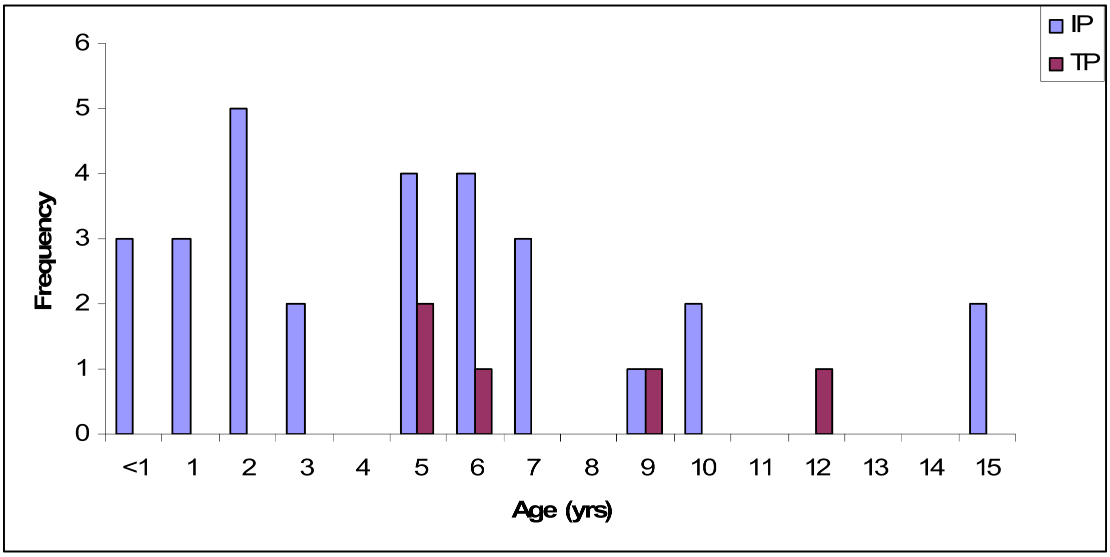

Figure 4.44. Age-frequency distribution for *Scarus rubroviolaceus* from Seychelles F. (IP=29 TP=5)

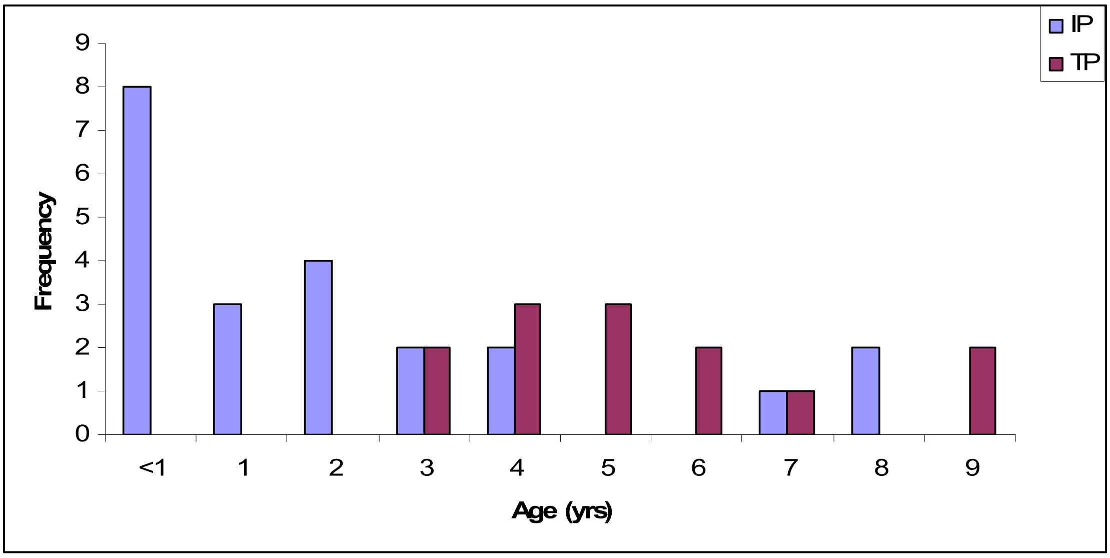

Figure 4.45. Age-frequency distribution for *Scarus rubroviolaceus* from Seychelles M. (IP=22 TP=13)

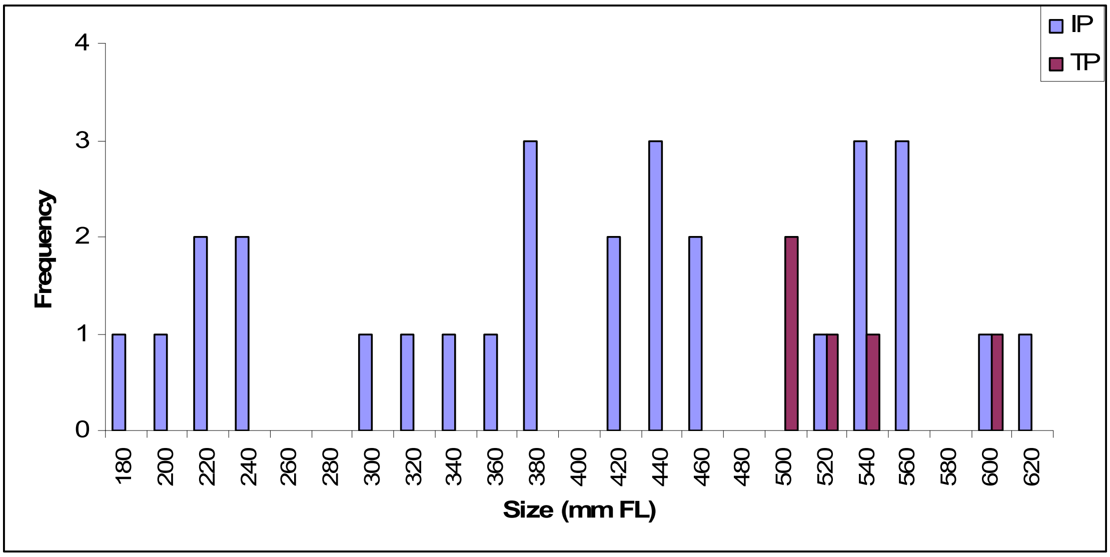

Figure 4.46. Size-frequency distribution for *Scarus Rubroviolaceus* from Seychelles F. (IP=29 TP=5)

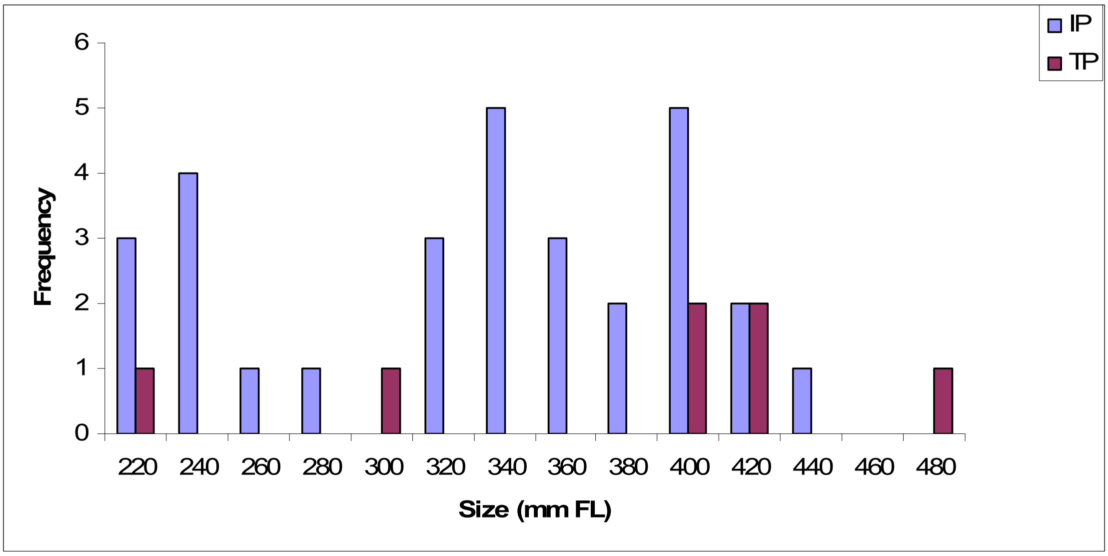

Figure 4.47. Size-frequency distribution for *Scarus rubroviolaceus* form Taiwan. (IP=30 TP=7)

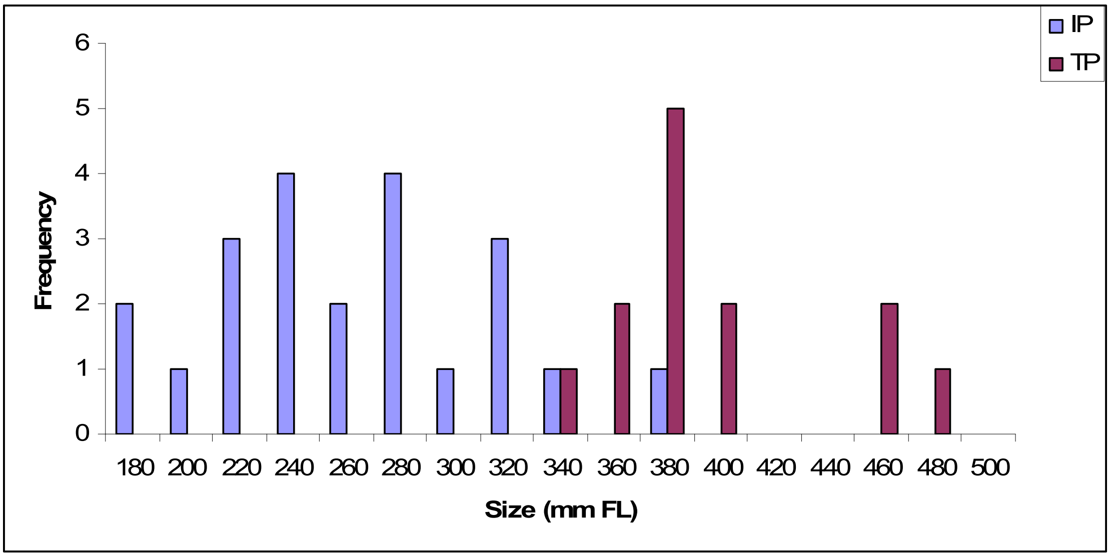

Figure 4.48. Size-frequency distribution for *Scarus rubroviolaceus* from Seychelles M. (IP=22 TP=13)

Figure 4.49. Regression of age based catch curves for estimation of *Scarus rubroviolaceus* mortality from Taiwan. Ln value for age class zero represent individuals less than 1 year of age.

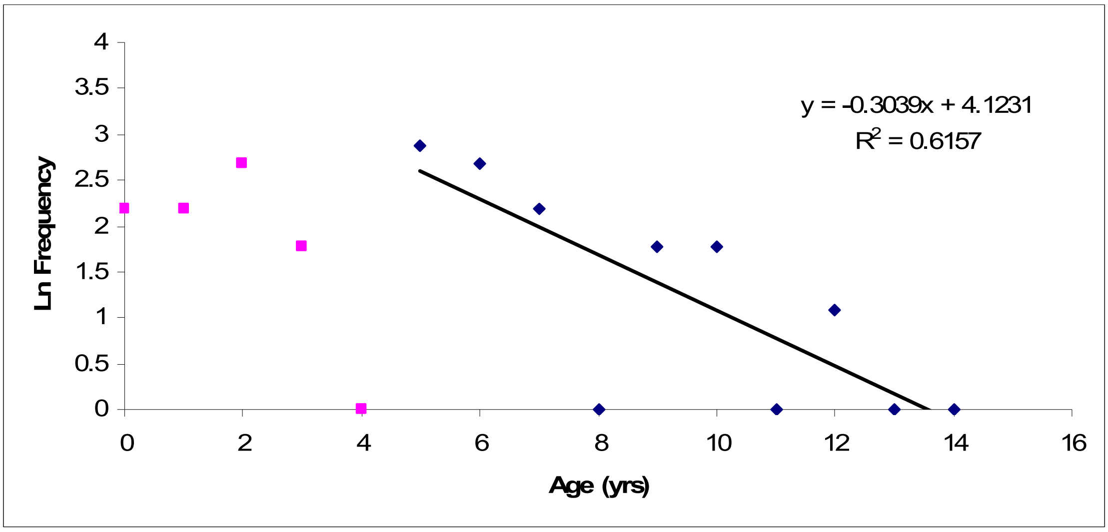

Figure 4.50. Regression of age based catch curves for estimation of *Scarus rubroviolaceus* mortality from Seychelles F.

Figure 4.51. Regression of age based catch curves for estimation of *Scarus rubroviolaceus* mortality from Seychelles M. Ln value for age class zero represent individuals less than 1 year of age.

4.3.3.1 Summary

Regular opaque bands were visible in sectioned otoliths when viewed under transmitted light (Figure 4.35). As with otoliths from the other parrotfish species, older individuals displayed deep sulcal grooves with prominent annual increments. Regression analysis of otolith weight versus age shows a linear relationship confirmed by strong r^2 values, especially for the Seychelles populations (Figures 4.36-4.38). However, the sample size of all three data sets are relatively small (n= 34, 35, 39) which can contribute to projections of unreliable statistical relationships.

Von bertalanffy growth functions were fitted to each data set, the results of which are displayed in table 4.3. The Farquhar population from Seychelles displayed the biggest size (621mm FL) and oldest age (15 yrs). Asymptotic length, mean maximum size and age projections appear to suggest that *S.rubroviolaceus* from Farquhar live longer and attain a bigger size compared with other locations. The oldest individuals from the Seychelles Farquhar dataset, along with other older specimens from Seychelles Mahe and Taiwan, exhibited IP colouration, indicating that females also persist through all age classes in this species.

A comparison of VBGF projections (Figure 4.39) utilizing size at age plots, puts in perspective differences of asymptotic length for each population. Clearly the Seychelles dataset from Farquhar appears to attain much larger size at equivalent age. A tentative comparison of VBGF growth trajectories (Figure 4.41) using two very small datasets from Oman (n=25,) and Solomon Islands (n=18) appear to suggest that *S.rubroviolaceus* from Oman enjoys the same relative bigger size and longevity with the Farquhar population from Seychelles. Size at age comparison of the oldest individuals from each population sample (Figure 4.42) confirm this assumption, while the Solomon Islands population displays the shortest longevity and relatively fastest growth.

Confidence interval analysis around the VBGF parameters (Figure 4.40) appear to suggest that the Seychelles Mahe and Taiwan populations do not differ significantly. However, given the small sample sizes of each population, caution must be exercised in making definitive conclusions. Age and size frequency distribution (Figure 4.43 - 4.48) decoupled by colour phase confirms expected population characteristics, including the persistence of females through older age classes, but also the absence of males in younger age classes from the Seychelles data sets. The significance of the presence of young TP individuals in the Taiwan dataset is also worth noting. Terminal phase individuals are presented in small size-ranges from Taiwan while only occurring in larger size classes in the Seychelles datasets. Estimates of mortality acquired from regression analysis appear to generate similar R^2 values for the three locations. ANCOVA estimate suggest that no significant differences exist between the three populations (ANCOVA F=0.712, P=0.5). However, the significance of higher fishing pressure in Taiwan (Burke et al., 2002) compared to relatively low fishing pressure in the Seychelles make this analysis unreliable.

4.3.4 **Pacific longnose parrotfish *Hipposcarus longiceps***

Figure 4.52. Pacific longnose parrotfish *Hipposcarus longiceps.*

Recognised for its distinctive pointy nose, TP Pacific Longnose parrotfish *H.longiceps* are white with light blue with orange scale margins, orange upper lip and fins, and have been recorded as reaching a maximum length of 60 cm (TL) in the Pacific Ocean (Allen and Steene 1994). Initial phase individuals are nearly white with some yellow tinges around the tail and dorsal area (personal observation). Known as scraping scarids, they appeared to be one of the more abundant species on coral reefs, with IP individuals swimming and foraging on benthic algae in large schools, in the company of TP species (personal observation). The Indian ocean sister species *H.harid* are also very similar in appearance, although they enjoy a different colouration and have been recorded as reaching a maximum size of 75cm (TL) (fishbase.org).

Analysis for this species was based on a population sample extracted from Solomon Islands (n=67). The entire sample was extracted from an unfished area adjacent to Rarumana in Western Solomons. Logistical constraints hampered our efforts in travelling back to this location in order to build a bigger and more representative sample. A further sixteen individuals (n=16) acquired opportunistically from the GBR were used to cautiously compare size at age plots with the Solomon Islands data.

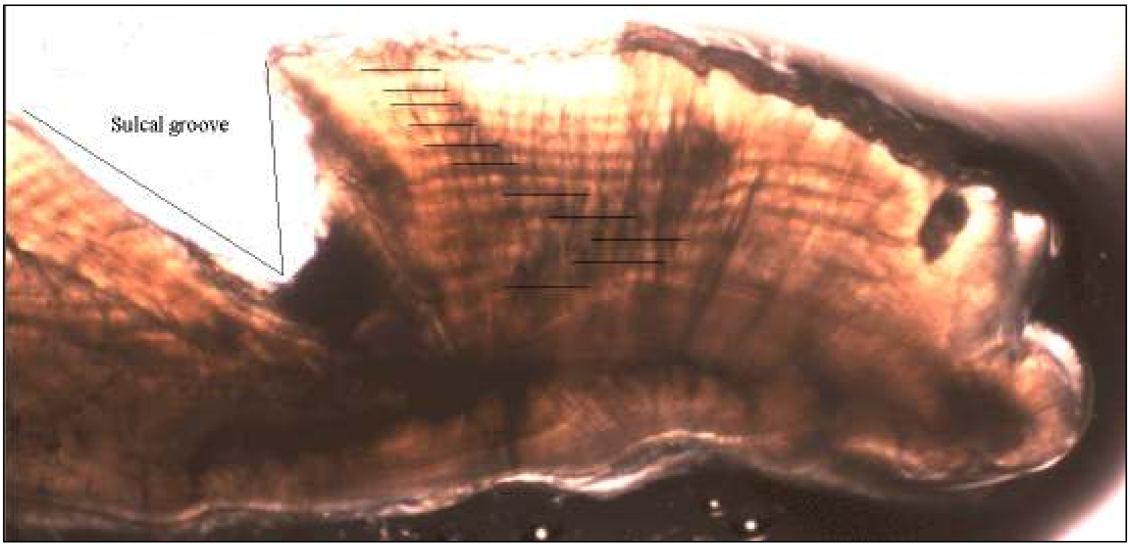

Figure 4.53. *Hipposcarus longiceps*. Sectioned sagittal otolith of an eleven year old from the GBR.

Location	n	Size (SL mm)	L∞	K	t_0	r^2	Mean max age	Mean max size
Solomon Is	67	190-330	286.0	1.19	-0.06	0.57	3.5	297

Table 4.4. Sample size, size ranges and VBGF parameters for population from Solomon Islands. Mean max age and size were calculated based on the oldest 15% of the population.

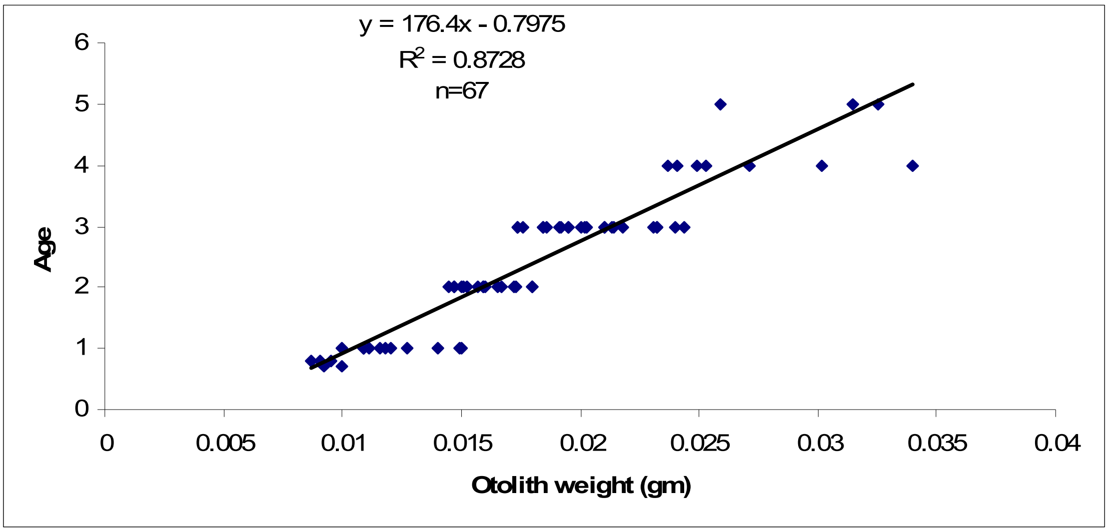

Figure. 4.54. *Hipposcarus longiceps* from Solomon Islands. Regression of sagittal otolith weight vs age.

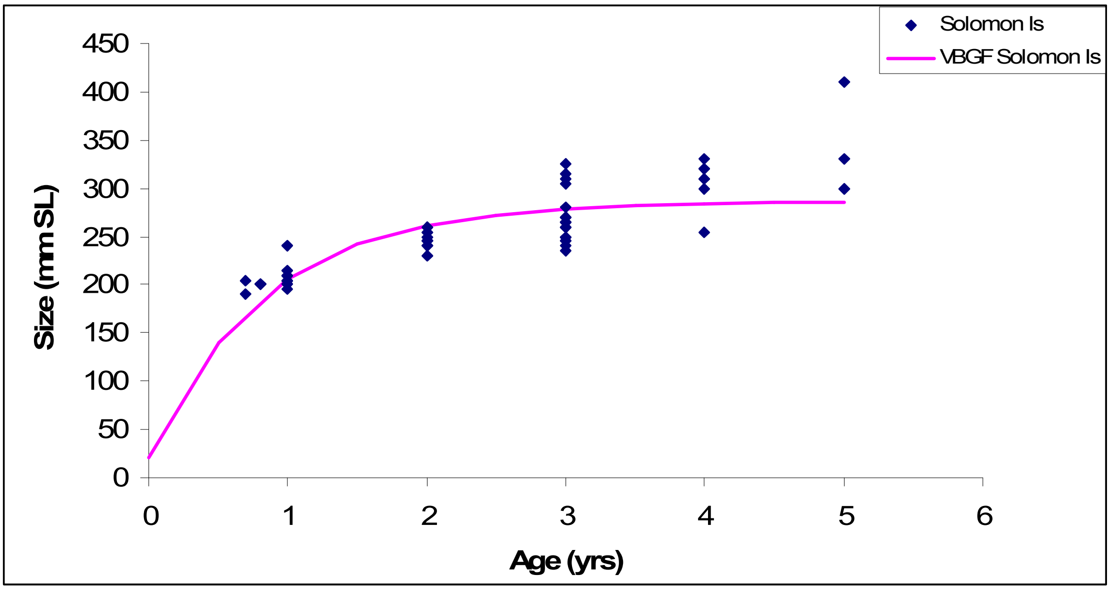

Figure 4.55. Size at age plots with fitted von Bertalanffy growth function curve for *H.longicpes* from Solomon Islands.

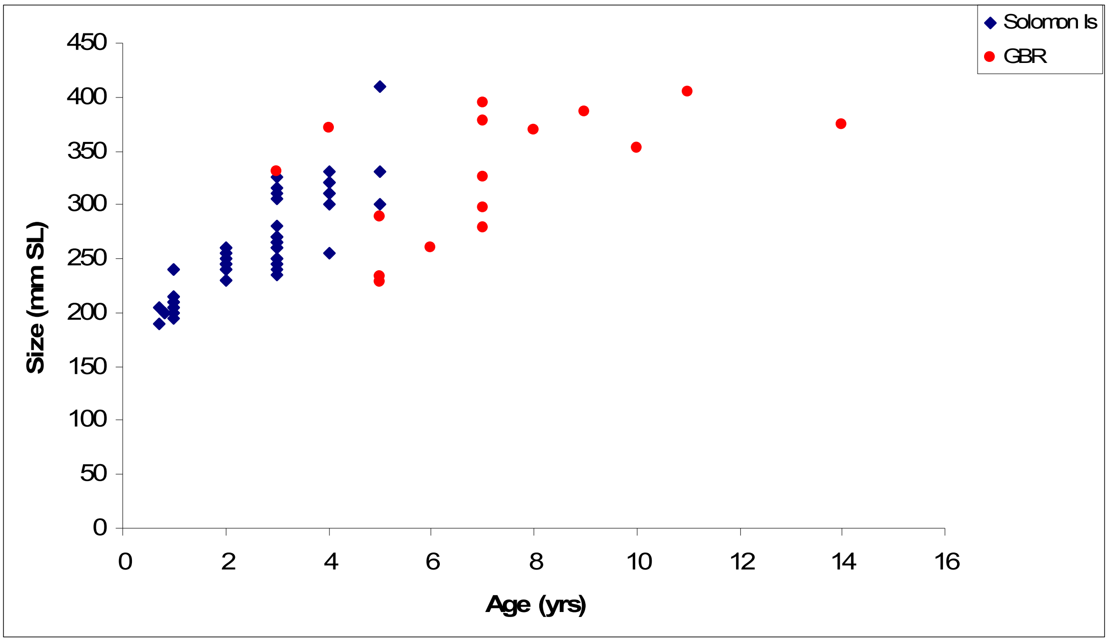

Figure 4.56. Size at age plots of specimens from GBR (n=16) fitted alongside specimens from Solomon Islands (n=67).

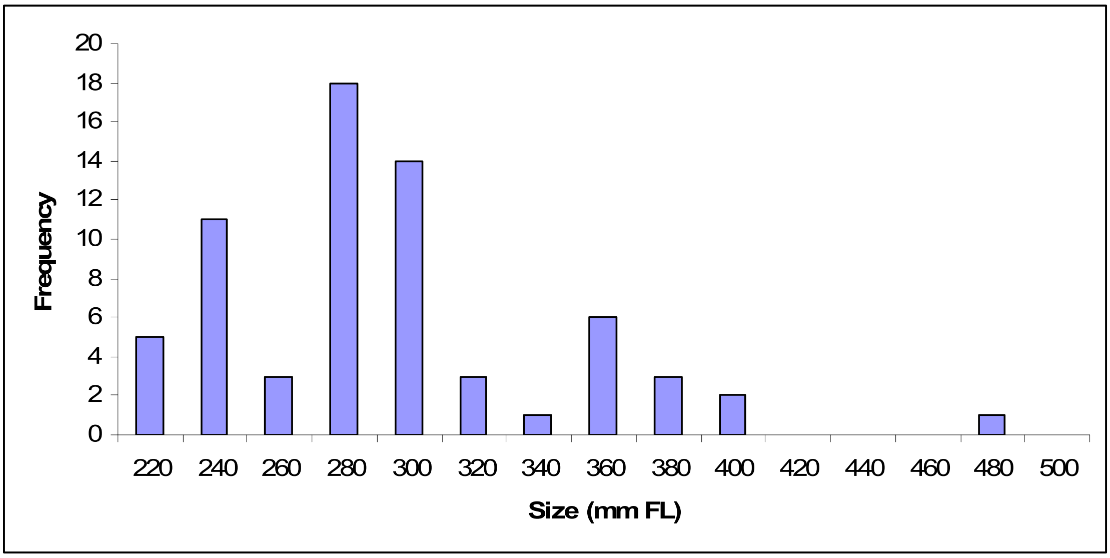

Figure 4.57. Size-frequency distribution for *Hipposcarus longiceps* from Solomon Islands. (n=-67)

Figure 4.58. Age-frequency distribution for *Hipposcarus longiceps* from Solomon Islands. (n=67).

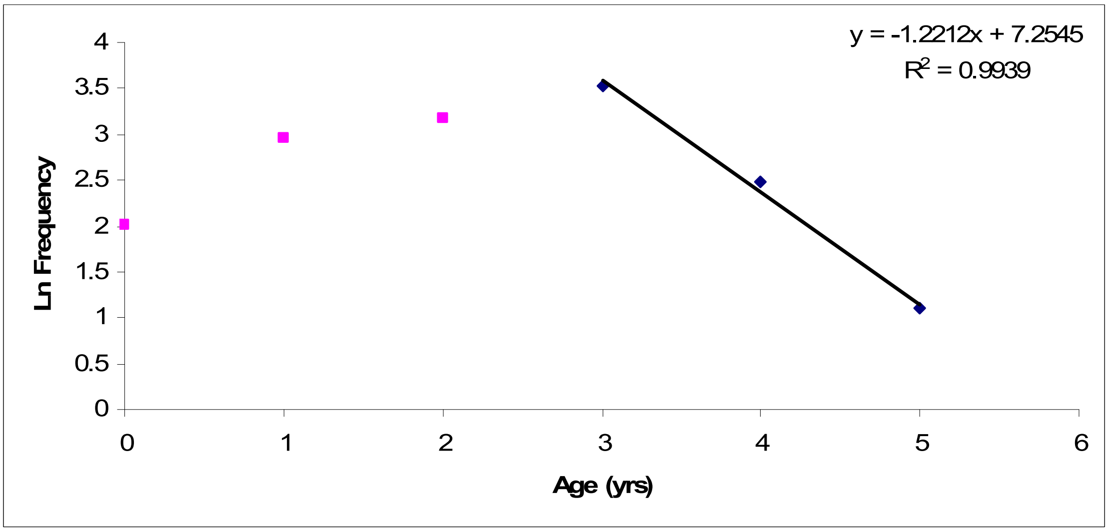

Figure 4.59. Regression of age based catch curves for estimation of *Hipposcarus longicpes* mortality from Solomon Islands.

4.3.4.1 Summary

Although sometimes difficult to ascertain, the matrix of sectioned sagittal otoliths examined for growth increments displayed regular opaque bands when viewed under transmitted light (Figure 4.53). Bands were more pronounced for the older individuals in both Solomon Islands and the GBR. Otolith weight plotted against age displayed a strong linear relationship confirmed by regression analysis (Figure 4.54). Maximum observed age was estimated at five years (330mm SL) while maximum observed length was measured at 410 mm (4 yrs) for the Solomons dataset (Table 4.4). Analysis of the dataset using the von Bertalanffy growth function estimated 286 mm as the size at which asymptotic length is reached (Table 4.4). Mean maximum age and size analysis based on the oldest 15% of the population yielded 3.5 yrs and 297mm, respectively.

Based on these analyses, and also the fact that *H.longiceps* dataset was acquired from an unfished region of the Solomons, it can be tentatively concluded that this species is short-lived and thus has high turnover rates. This assumption is validated by other sources in the WWW.fishbase.org database which point to minimum population doubling occurring of less than 15 months. The importance of these characteristics must not be underestimated, as this dataset goes someway to validating the assumption that parrotfish species in the Solomons are short lived and relatively faster growing compared to their other ocean counter parts.

Comparison of size at age plots against a small sub-sample (n=16) from the GBR reveals two interesting characteristics (Figure 4.56). Firstly, the longevity of *H.longiceps* in the GBR is markedly longer reaching a maximum of 14 yrs of age (SL 375mm) compared to 5 yrs (SL 410mm) for the Solomons. Secondly, growth plateaus at about 400mm for both sample sets, with maximum size reached for the Solomons and GBR measured to a standard length of 410mm (5yrs old) and 405mm (11yrs old), respectively. Clearly the Solomon Islands population sample grew faster and achieved a bigger size at equivalent age compared to the GBR sample. Although a much larger and representative sample from the Solomons is required to definitively estimate longevity, the fact that the younger *H.longiceps* from the Solomons exhibited similar sizes to much older specimens from the GBR can lead us to tentatively conclude that *H.longiceps* does not enjoy extended longevity in the Solomon Islands. Although unlikely, we must not discount the fact that the markedly large differences between ages of the two populations at equivalent size may also be an indication that very old individuals may historically be overfished in western Solomon Islands. Samples were not identified by colour phase or sex, thus size and age frequency distribution were pooled (Figure 4.54 – 4.55). Mortality estimates for the Solomons population yielded a R^2 value of 0.9939. This value could be significant based on which assumptions are adopted. If we assume that *H.longiceps* are short lived in the Solomons, the R^2 value is indicative of this life-history trait. However, if we assume that historical fishing pressure has removed bigger individuals from the Western Solomons, this analysis can be regarded as statistically unreliable (unlikely).

4.4 Discussion

Demographic analysis of parrotfishes from different locations around the world has identified major differences in growth patterns, asymptotic size and longevity. In relation to clear patterns, there appears to be none when datasets are organised into ocean groups. This reinforces the role of environmental and anthropogenic factors in post-settlement processes, such as food availability as proposed by Jones (1986) and Gust et al., (2002), resource availability, competition and density-dependant concepts as proposed by Hixon and Jones (2005) and Choat & Robertson (2002), the effects of temperature as proposed by Conover (1992), and the effects of fishing pressure on mean size and biomass as described by Jennings and Polunin (1996). Furthermore, comparative phylogenetic analysis is revealing that species exhibit distinctly different evolutionary histories with different spatial scales (Bernardi et al. 2001; Bay et al. 2004).

In a comparative demographic analysis on the GBR, Gust (2000) was able to draw a correlation between higher densities of scarids on the outer shelf reef crests and their subsequent slower growth rates. However, more recently Hamilton (2004) found the opposite to be true with the Bumphead parrotfish *B.muricatum* from the Solomon Islands where individuals from the more densely populated *Tetepare* region enjoyed faster growth. Drawing from this study, the *S.rubroviolaceus* dataset provides a good example on the influence of resource availability on growth rate and longevity. Specimens from the Seychelles were extracted from two regions; the granite reefs of the Island of Mahe and the carbonate reefs of the Farquhar region (700 kilometres southwest Mahe). The different food sources available to this scraping scarid may be partly responsible for its markedly different growth pattern and longevity, with the coral rich Farquhar reefs harbouring the bigger and longer-lived individuals. However, this is only a tentative conclusion based on a small sample size. As with the contrasting density-dependant growth analysis of Gust (2000) and Hamilton (2004), it is quite possible that in-depth comparative analysis of diet and resource availability might shed new light on this issue.

Comparisons of VBGF parameters carried out via 95% confidence ellipses revealed that, limitations considered, no two populations enjoyed similar growth trajectories or mean asymptotic length, with every population exhibiting different maximum sizes and ages. Scarids from Solomon Islands appear to be the shortest lived in comparison to other locations. Their growth however appears to be as fast, if not faster, than scarids in other locations, reaching the same equivalent size in much younger age classes. Demographic analysis on *S.ghobban* and *H.longiceps* appear to support this conclusion which is also circumstantially supported by the handful number of *S.rubrovialceus* and *C.microrhinos* individuals opportunistically collected from the same region. In figure 4.42., for example, a two and four year old *S.rubroviolaceus* from Solomon Islands are shown to have reached the equivalent size of a seven and eight year old from Seychelles (Mahe), respectively. This disparity in growth rate is more clearly demonstrated in the *H.longiceps* data set (Figure 4.56), where the Solomon Islands population are undoubtedly growing faster and reaching their asymptotic length years earlier compared to their GBR counterparts. The size reached by the oldest individuals from the Solomons (5yrs, 410mm SL) has not been surpassed by any other individual from the GBR population, including the two oldest at 11 yrs (405mm SL) and 14 years (375mm SL).

However, there is an added significance to the *H.longiceps* dataset. The population sample were sourced from an uninhabited region of the Western province where no fishing activity occurred at the time. This fact adds weight to the short lifespan and faster growing notion of Solomon Islands scarids proposed in this chapter. Although it is entirely reasonable to argue that historical fishing pressure could be responsible for the lack of larger specimens across the entire western Solomons, the same could not be argued with regards to rate of growth and equivalent size at age in comparison with the GBR data. Therefore, while the lack of older individuals may be an issue in the Solomons dataset, their faster growth rate cannot be disputed.

The significance of fishing pressure is two-fold. Not only it helps us to support certain demographic assumptions, it is also an integral part of the mortality formula. The unique status of scarids as a non-commercial species means that in some areas they are at worst subject to recreational fishing pressure (e.g. GBR) or fished very lightly (e.g. Cocos Keeling, Seychelles). This means that some of the mortality estimates presented in this chapter could theoretically be considered as reliable estimates of natural mortality. However, given the skewed age and size distribution of some datasets, it would be unwise to make a definitive conclusion. The most accurate estimate of natural mortality may be awarded to *H.longiceps* from the Solomon Islands based on estimated zero fishing pressure and evenly distributed age. However, as other recent studies have pointed out (Clifton 1995; Gust 2000; Hamilton 2004; Kritzer 2004) mortality can differ markedly between local populations and is dependant on other physical and environmental factors such as predation or refuge availability. Mortality is also demographic and life-history dependant (Caley 1998), which in light of the age and growth plasticity demonstrated in this chapter makes comparisons of mortality rates inappropriate.

Comparative analysis in this chapter has also been able to reinforce some general demographic trends. Sex-specific growth rates have previously been shown to exist in many reef fish species (Appeldoorn 1996; Kritzer 2004), including scarids (van Rooij et al. 1995; Choat et al. 1996). In this study, for example, VBGF growth curves were decoupled by sex or colour phase for *C.microrhinos*, *C.strongylocephalus* and *S.ghobban.* Faster male or TP growth was shown to exist in all cases. Choat and Robertson (2002) attribute the disparity of sexual specific growth in scarids to their protogynous nature whereby during or after transition from female to male, less energy is subsequently required for sperm production, which ultimately means that more energy can be spent on somatic growth (Wootton 1985).

The persistence of females in older age classes is also interesting and requires further examination. In some cases females appeared as the oldest individuals in those populations. These included a 15 year old *C.microrhinos* from GBR, a 15 year old *S.ghobban* from Taiwan*,* a 15 year old *S.rubroviolceus* from Seychelles (Farquhar) and a 12 year old *C.strongylocephalus* from Seychelle (Mahe). According to Begg et al., (2005), a broader age and length distribution, and larger and older females, are indicators of a healthy population which is not being subjected to heavy fishing pressure. On the basis of fishery related information cited in this chapter, it appears that this conclusion can be corroborated with regards to the Seychelles and GBR, but not Taiwan, which is subjected to high fishing pressure (Burk et al., 2002). In fact, the presence of TP individuals in the early age and size classes of Solomon Islands and Taiwan datasets may be indicative of heavy fishing pressure (discussed further in chapter 5). In the light of this analysis, the 15 year old female *S.ghobban* from Taiwan may unfortunately represent a very rare occurrence.

This chapter has reinforced the notion that presumption of demographic characteristics based on physical, environmental or anthropogenic generalisations are unwise and indeed dangerous, particularly when there are management implications. Indeed, this research has validated the conclusion that age-based demography of parrotfishes are much more localised than previously thought. It is likely that a combination of ecological, biological, and fishery related factors influence how scarids grow and age, thus attributing demographic trends to specific factors is unwise. Certainly, variations in demographic parameters may be enhanced by specific differences in post-settlement processes, such as food availability, competition, population density, water temperature or fishing pressure. Drawing a specific example from this chapter, the markedly different growth trajectories and longevity of *S.rubroviolaceus* from two different regions within Seychelles' territorial waters are indicative of spatial demographic plasticity. I have tentatively proposed differences in resource availability between carbonate and granite reefs as responsible for this disparity.

The overall impression of scarids in the Solomons are that they grow fast, have shorter life span, young age-at-maturity (established in chapter five) and enjoy high turnover rates. According to Polunin and Graham (2003) or Dulvy et al., (2003), this r-selected life history trait should be beneficial in terms of resiliency to overfishing. However, quantified fishing pressure (chapter two) and established declining abundance (chapter three), suggest that this particular life-history trait is not able to compensate for parrotfish vulnerability to overfishing. Two key points are evident here; 1) Faster growth, shorter life spans and high turnover rates are evident, even in the absence of fishing pressure (e.g. *H.longiceps*), and 2) Species with short life spans and high turnover rates may still be vulnerable to overfishing. Pauly et al., (2002) states that high fishing pressure induces an expected evolutionary response in terms changes in population dynamics. In order to shed more light on this matter, and also investigate the reproductive parameters of parrotfish, the next chapter undertakes a comparative analysis of scarid life-history.

Chapter Five

Parrotfish reproductive biology; a comparative analysis

Reproductive information on two species are presented in this chapter; Pacific Steephead parrotfish Chlorurus microhinos, and Blue-barred parrotfish Scarus ghobban. The C.microrhinos dataset represents the work of Alec Hughes, acquired as part of a collaborative study on parrotfish demography and life-history with Professor Howard Choat.

5.0 Introduction

The life history of scarine labrids is equally interesting and relevant in fishery management. As protogynous hermaphrodites, they are thought to develop into males via two routes; monandry and diandry (Choat and Robertson 1975). In monandric species, all males are derived exclusively from functional females through sex change (i.e. secondary males), while diandric species undergo two male developmental pathways; some males differentiate directly from juveniles through sexual differentiation (i.e. primary males), while others are secondary males, as in monandry (Liu and Sadovy 2004)

Not all scarids are protogynous hermaphrodites. The Marbled parrotfish *Leptoscarus vaigiensis* (Robertson et al. 1982) and the Mediterranean parrotfish *Sparisoma cretense* (de Girolamo et al. 1999) have been shown to be gonochrostic where each sex develops separately from the other. It now appears that gonochorism may not be such a rare occurrence in scarids. As part of a detailed demographic and reproductive analysis of the Bumphead parrotfish *B.muricatum* Hamilton (2004, 2008) carried out histological analysis on gonadal tissue samples. Analysis of life-history data indicated that this species may indeed be gonochoristic. However, Hamilton does point out that this may be a retaliatory response to high fishing pressure on large adult males by a species that may otherwise be protogynous hermaphrodite.

The majority of scarids are also dichromatic with an initial phase (IP) and terminal phase (TP) colouration during which they change colour from drab colours to more brighter green or blue (Choat and Robertson 1975; Demartini et al. 2005). Example of monochromatic (do not change colour after sex reversal) species include the Bumphead parrotfish *B.muricatum* or the Pacific Steephead parrotfish *C.microrhinos*.

Choat and Robertson (1975) state that colour phase is closely synchronised with sex in scarids where all TP individuals are mature males, while most, but not all IP individuals are females (Hawkins and Roberts 2003). Using phenotypic characteristics as a proxy for allocating gender in larger scale studies where histological gonad analysis is not practical has been previously employed (Hawkins and Roberts 2003). However, this technique has its limitations given that establishment of detailed life-history stages such as sex change or spawning periodicity is not possible without histological analysis.

Increasingly, we are establishing that not all scarids conform to the conventional notion of colouration and sex distribution. For example, Choat and Robertson (1975) point out that in diandric species primary males adopt the initial phase colouration of females and go under normal transition from IP to TP colouration with somatic growth later in life. The bumphead parrotfish *B.muricatum*, is a monochromatic species that adopt the same blue-green colour throughout its life, and apparently has separate male/female development pathways (Hamilton 2004, 2008). The Dusky parrotfish *S.niger*, on the other hand, does change sex, but is also monochromatic (Choat and Robertson 1975).

In monandric species, the growth disparity between initial and terminal phase individuals is more pronounced. According to Sadovy and Shapiro (1987), IP females are generally smaller than TP males. It is a well established fact that females have the dual responsibility of egg production and somatic growth, while males invest relatively little energy in sperm production. In fact, gonads require energy input for their development and will compete with somatic growth for resources (Hart and Reynolds 2002, Pitcher and Hart 1982).

Although certain generalisations have been established with regards to relationship between size, colouration and sexual life-history (Sadovy and Shapiro 1987) it can be confidently stated that scarids do not always conform to these conventional standards. As a result, management initiatives must seriously consult and incorporate life-history parameters such as specific sexual developmental pathways, the age and growth rates pre and post-maturation, and the age at which sex change does occur. However, the role of fishing pressure further complicates ecologically mediated life-history characteristics.

The reason for this lays in the fact that fishing pressure may induce compensated and uncompensated responses (Vincent and Sadovy 1998). In a compensated response the population reacts to fishing pressure through a decline in average size of females and males, while an uncompensated response happens as a result of fishing pressure past the point of fix sexual reversal (i.e female to male) in which case the population moves towards a highly female biased sex ratio (Vincent and Sadovy 1998). The varying life history characteristics of scarids have the potential to complicate management measures as a uniform strategy cannot be employed. As a result, the relevance of reproductive information is two-fold. Firstly it can be used to establish age at maturity, pattern of sex reversal and stages associated with establishing female maturity. Secondly, these parameters can also be used to tailor management measures towards the population's specific regenerative capacity in consideration to ongoing fishing pressures.

Given the holistic fishery, ecological, and demographic approach taken in this thesis, it was logistically difficult to direct the same concerted effort towards conducting large scale comparative life history analysis. As a result, a collaborative effort was sought from Alec Hughes as part of a joint investigation into the demography (chapter four) and life-history of *Chlorurus* species.

5.1 Objectives

This chapter satisfies the reproductive biology aspects of aim two by providing life-history information on two species of parrotfish. In particular, information related to major life history events such as M/F ratio, maturation and sex reversal are provided. Also of interest here is the role of heavy fishing pressure in inducing reproductive phase shifts in sexual development pathways. This aspect is briefly discussed.

The objectives here were to use histological techniques in order to:

1) Confirm the sex of each specimen.
2) Confirm the maturation stage of each specimen.

Having histologically established sex and maturation stage, size and age distribution, along with VBGF parameters were again formulated for each species to provide a complete life-history picture.

5.2 Methods and Materials

5.2.1 Study species

I was able to collect a relatively small sample (n=45) of gonadal tissue for *Scarus ghobban* from the Gizo reefs of Solomon Islands. The significance of the *S.ghobban* dataset is that it will allow me to discuss the possible effects of heavy fishing pressure on scarid reproductive biology.

Gonads from 45 *S.ghobban* individuals were histologically examined. A much larger *C.microrhinos* population sample was collected from the GBR by Alec Hughes as part of a collaborative demographic and life-history investigation of the *Chlorurus* genus. In total, the tissues of 143 gonads were histologically examined for this separate analysis.

5.2.2 Histological analysis

Solomon Islands gonad samples were stored in a preservative solution of 10% phosphate buffered formalin (Acquired from Number 9 Hospital, Honiara, Solomon Islands). The GBR samples were stored in FAACC (4% formaldehyde, 5% acetic acid, 1.3% calcium chloride).

Before sectioning, whole gonads were weighed to the nearest 0.001g. Assuming that no histological differences exists between the right or left lobes, three sections, roughly 3-4mm in thickness, were cut away from the Proximal (P), Medial (M), and Distal (D) regions of the right or left lobe. This was carried out for 1/4 of the total samples at random to determine if the reproductive status along the length of the gonads were consistent. As no evidence of this was observed, subsequent extractions were only made from the medial or central section of each gonad.

Tissue sections were placed in processing cassettes and submerged in 70% alcohol overnight before being transferred to a tissue processor machine where they were dehydrated in ascending grades of alcohol. Tissues were then embedded in paraffin wax blocks, frozen and then prepared for sectioning. Transverse sections were then cut at 5 µm and routinely stained using Mayers' Haematoxlyin and Young's Eosin-Erythrosin stains (Winsor, 1991).

Histological classifications of gonad samples were carried out using descriptions of sexual categories described by Samoilys and Roelof (2000), Adams (2002) and Pears (2005) (see Table 5.1). Transitional individuals were determined according to descriptions in Sadovy and Shapiro (1987). In order to ascertain if the analysed species were diandric protogynous hermaphrodites, the presence of primary and secondary males were determined according to criteria described by Shapiro and Rasotto (1993). The suitability of adapting gonadal description of serranids (refs cited above) in establishing the reproductive stages of scarids has been verified by Hamilton (2004) and Gust (2000).

Developmental stage	**Histological description**
Immature female (IM)	No evidence of prior spawning, ovary small and lamellae well packed with gonia and pre-vitellogenic oocytes.
Resting female (RS)	Ovary dominated by pre-vitellogenic oocytes. Potential indication of prior spawning (atretic vitellogenic oocytes). Possible presence of brown bodies. Thick wall. Ovary larger in diameter than IM.
Ripe female (RP) (i.e, ripening, near spawning & spent)	Ripening or vitellogenic oocytes present. May contain hydrated oocytes. Indication of prior spawning maybe present such as "spent" oocytes undergoing atresia.
Transitional	Previtellogenic oocytes are present. Degenerating oocytes present. Primary and secondary spermatocytes prevalent.
Primary Male	Primary male Proliferating testicular tissue in a gonad similar in appearance to an immature ovary, where spermatogenesis has progressed at least to the secondary spermatocyte stage. Ovarian tissue dominates the lamellae, which are not yet of the typical lobular form of the mature testes, and there are no signs of prior spawning as a female.
Male	Testes dominated by testicular tissue, which may contain gonia and germinal stages at various developmental stages. Includes inactive and active males.

Table 5.1. Histological criteria used to categorize males and females and subsequent female gonad development stages.

Estimates of age and size at maturity were determined by plotting the mean gonad weight of each size class against the relative frequency (%) of immature and mature females. An increase in the proportion of mature females accompanied by an increase in mean gonad weight was interpreted as females reaching maturity. Additionally, mean Gonadosomatic Index (GSI) were decoupled by sex and plotted against size classes of both species to observe any disproportionate increase in GSI with size.

5.3 Results

5.3.1 Scarus ghobban (Solomon Islands)

Sex	*n*	*Size (SL mm)*	*L∞*	*K*	*t_0*	*r^2*
Male	25	205-350	300.50	2.33	-0.02	0.12
Female	19	205-350	260.67	2.58	-0.03	0.21

Table 5.2. Sample sizes, size ranges and VBGF parameters for the Solomon Island *Scarus ghobban* population partitioned by sex. Transitional individuals were not included in analysis.

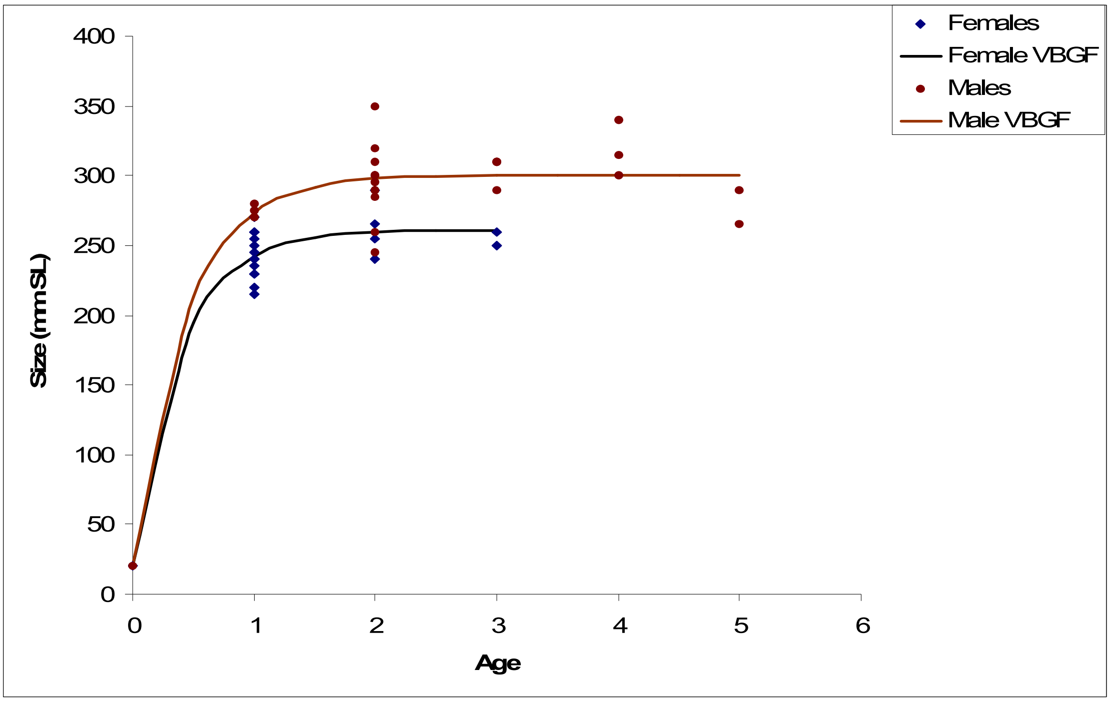

Figure 5.1. Size at age plots with fitted von Bertalanffy growth function curves for Male and Female *Scarus ghobban* from Solomon Islands. (VBGF parameters in Table 1)

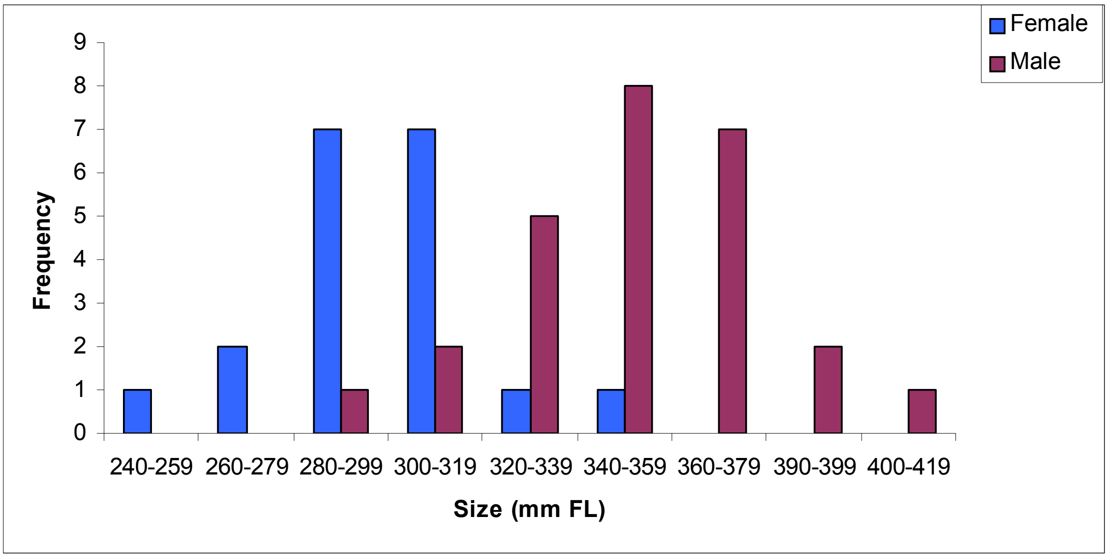

Figure 5.2. Size-frequency distribution of *Scarus ghobban* partitioned by sex.

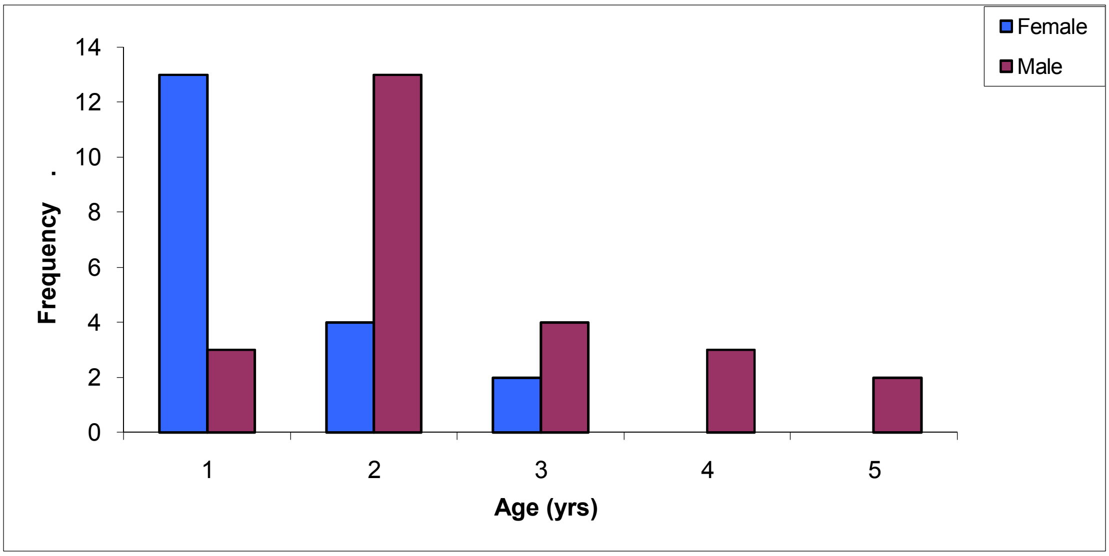

Figure 5.3. Age-frequency distribution of *Scarus ghobban* partitioned by sex.

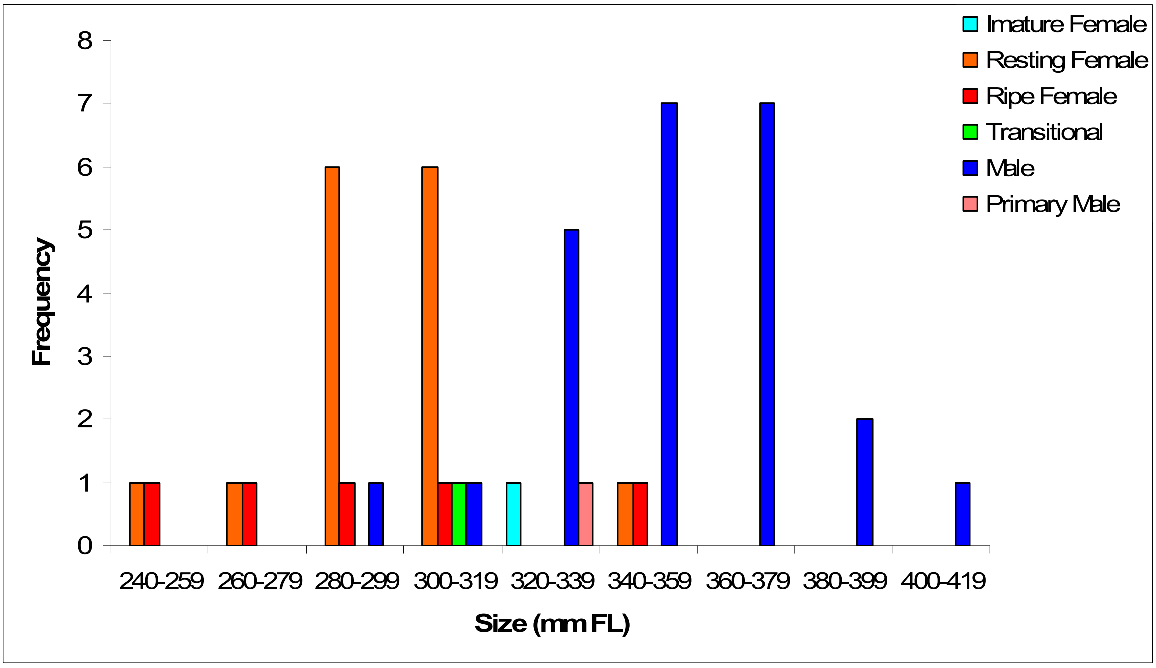

Figure 5.4. Size-frequency distribution of *Scarus ghobban* partitioned by gonad development stage.

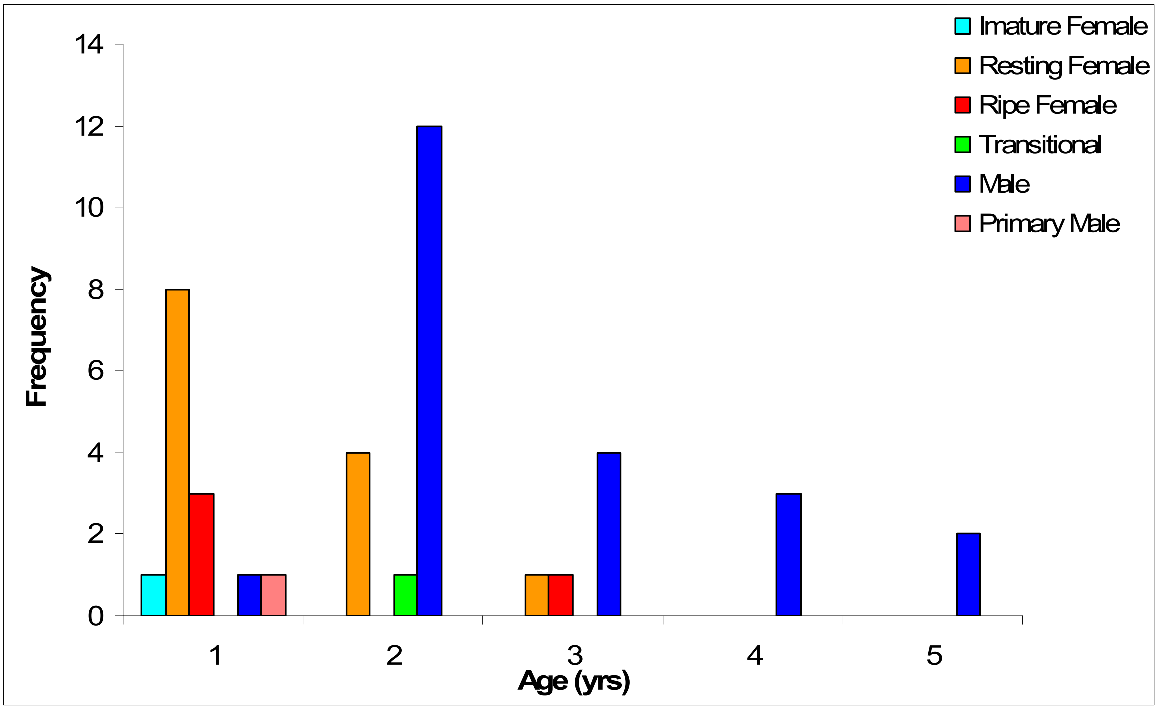

Figure 5.5. Age-frequency distribution of *Scarus ghobban* partitioned by gonad development stage.

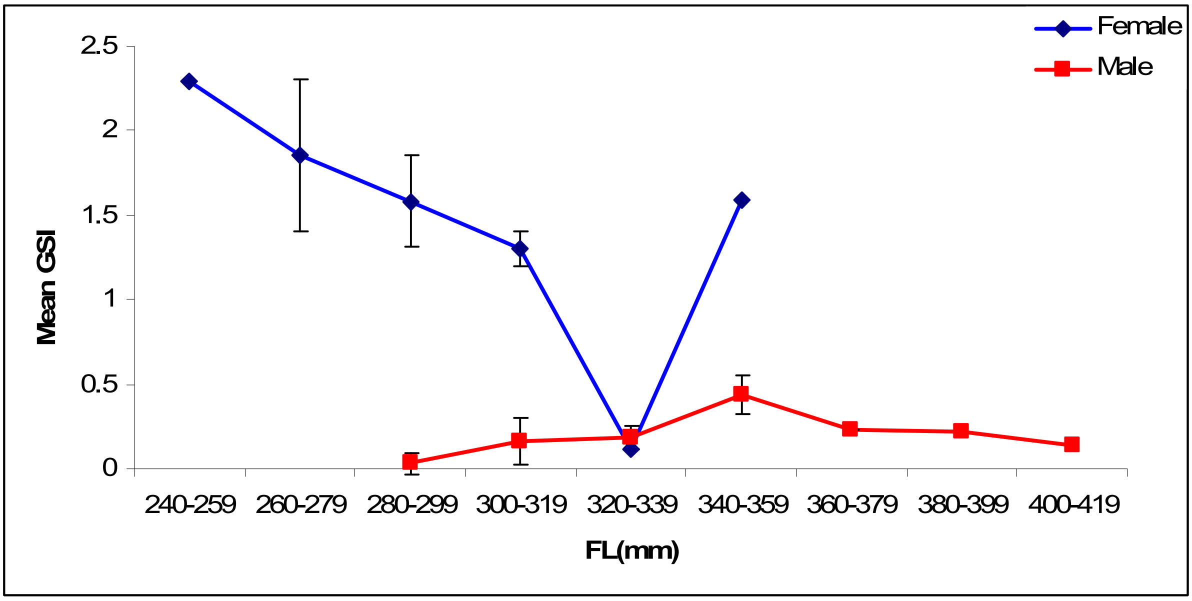

Figure 5.6. Gonad somatic indices for *Scarus ghobban* versus fork length partitioned by sex.

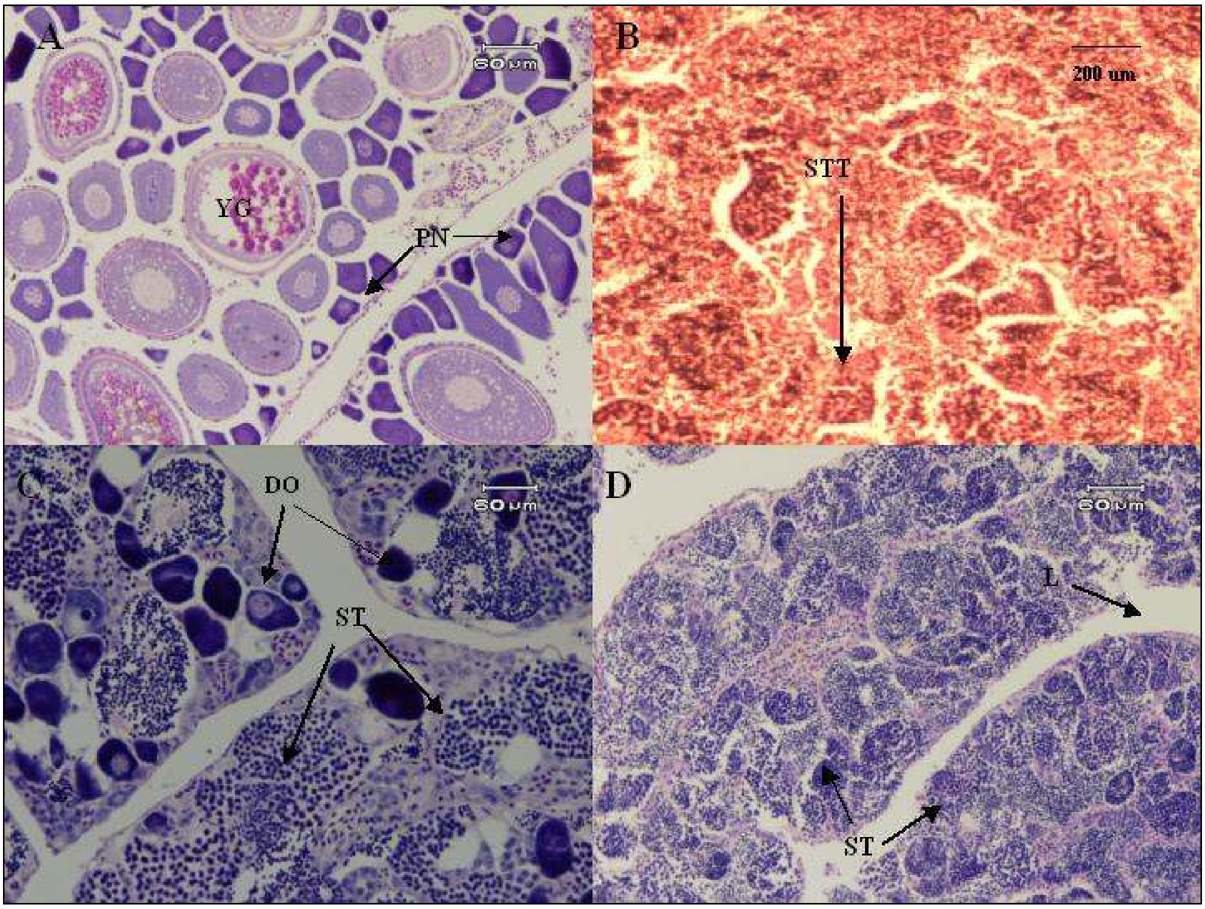

Figure 5.7. Evidence of sex change and dual male development pathway in *Scarus ghobban*. (A) Mature resting female with occyte development, (B) Mature male (IP) with no prior sign of female function, (C) Transitional male (TP), (D) Mature male with ovarian features. STT: spermatogenic tissue, DO: degenerative oocyte, PN: perinucleolar stage oocytes ST: spermatids, L: ovary lumen, YG: yolk globule stage occyte.

5.3.1.1 Summary

Comparisons of estimated growth trajectories partitioned by sex show larger sizes at equivalent age for males (Figure 5.1). The sex ratio was calculated at 1.3 M: 1F with overlap of males and females size distribution occurring within the 280 and 340 mm size categories (Figure 5.2). Age distribution partitioned by sex revealed that both sexes are more abundant in the younger age classes and decline thereafter (Figure 5.3). Males appears very early in the age classes and overlap with females, while no females were estimated to be older than three years or larger than 359 mm (FL).

Mature male and female gonads were relatively easy to identify macroscopically. Ripe ovaries and testes were orange and white in colour, respectively. Histological analysis revealed the smallest mature female to be one year old and 255mm FL, while the smallest mature male was one year old and 290mm FL (Figure 5.4 & 5.5). Given the relatively small sample size, predictions and calculations of life history parameters may be speculative. However, size and age distribution categorized against gonad development stage has produced several interesting points worth discussion. The appearance of a single immature female (1 year old 320mm FL) and transitional male (2 yrs old 310mm FL) is significant on a number of levels. Firstly, the presence of a transitional individual form a relatively small population sample (n=45) may be significant given the fact that protogynous hermaphrodites change sex quickly (Gust 2000). Secondly, the lack of immature females is not only interesting from a reproductive perspective, but also prohibits us from making accurate assumptions on age and size at first maturity. This is an important characteristic where very young and small females are already sexually active and contributing the reproductive output of the local population. This raises the question of how young and at what size does *S.ghobban* first become sexually mature, and is this a reflection of historical heavy fishing pressure or natural life-history characteristics?

The presence of a single transitional male will not by itself enable us to establish patterns associated with sex reversal. In particular, we need a higher percentage of transitional individuals in order to be able to estimate approximate age and size at sex change. However, the presence of a one year old mature IP male with no prior sign of female function, in combination with the transitional male, is indicative of sex change and dual male development pathway (Figure 5.7). Therefore, we can confidently assume that *S.ghobban* has been shown to be a diandric protogynous hermaphrodite. Finally, histological confirmation of gender versus colouration revealed that only the individual mature primary-male exhibited an IP colouration. All other males and females followed TP and IP colouration, respectively. The GSI of females was significantly higher than males ($t=5.41$, $P<0.0001$) with females enjoying higher percentage of gonad to body-weight ratio (Figure 5.6).

5.3.2 Chlorurus microrhinos (Great Barrier Reef)

Sex	***n***	***Size (SL mm)***	***L∞***	***K***	***t0***	***r^2***
Male	47	308 – 587	558.35	0.28	-0.13	0.66
Female	95	197 – 534	465.74	0.34	-0.13	0.56

Table 5.3. Sample sizes, size ranges and VBGF parameters for the GBR *Chlorurus Microhinos* population partitioned by sex. Transitional individuals were not included in analysis.

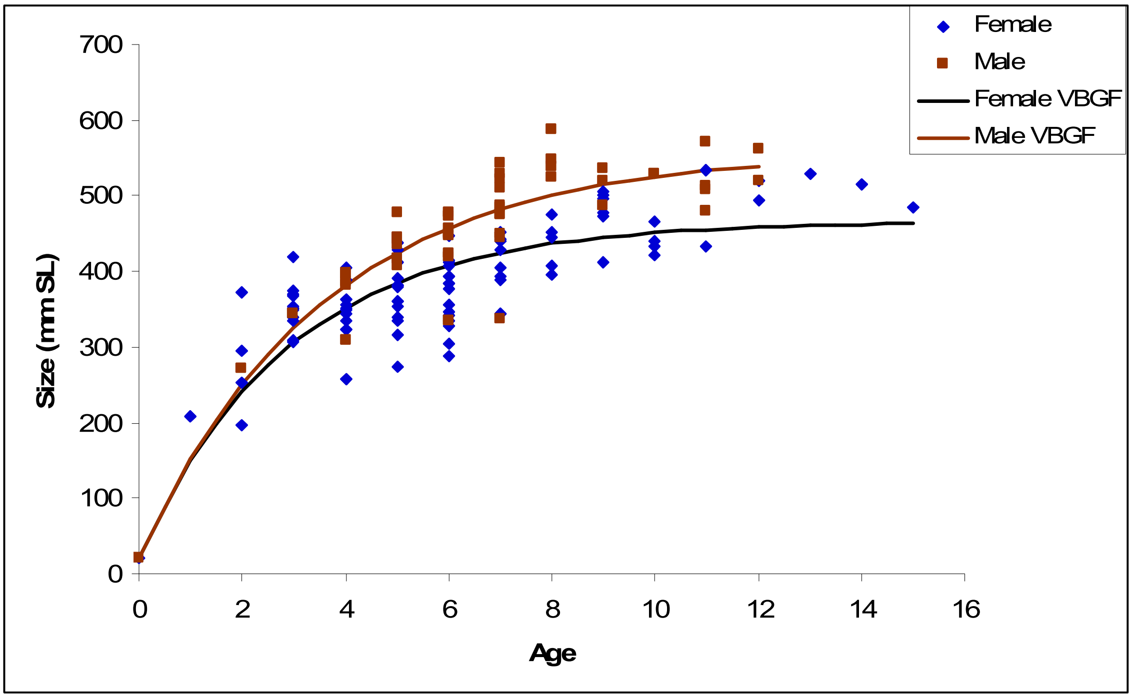

Figure 5.8. Size at age plots with fitted von Bertalanffy growth function curves for Male and Female *Chlorurus Microrhinos* from GBR. (VBGF parameters in Table 5.2)

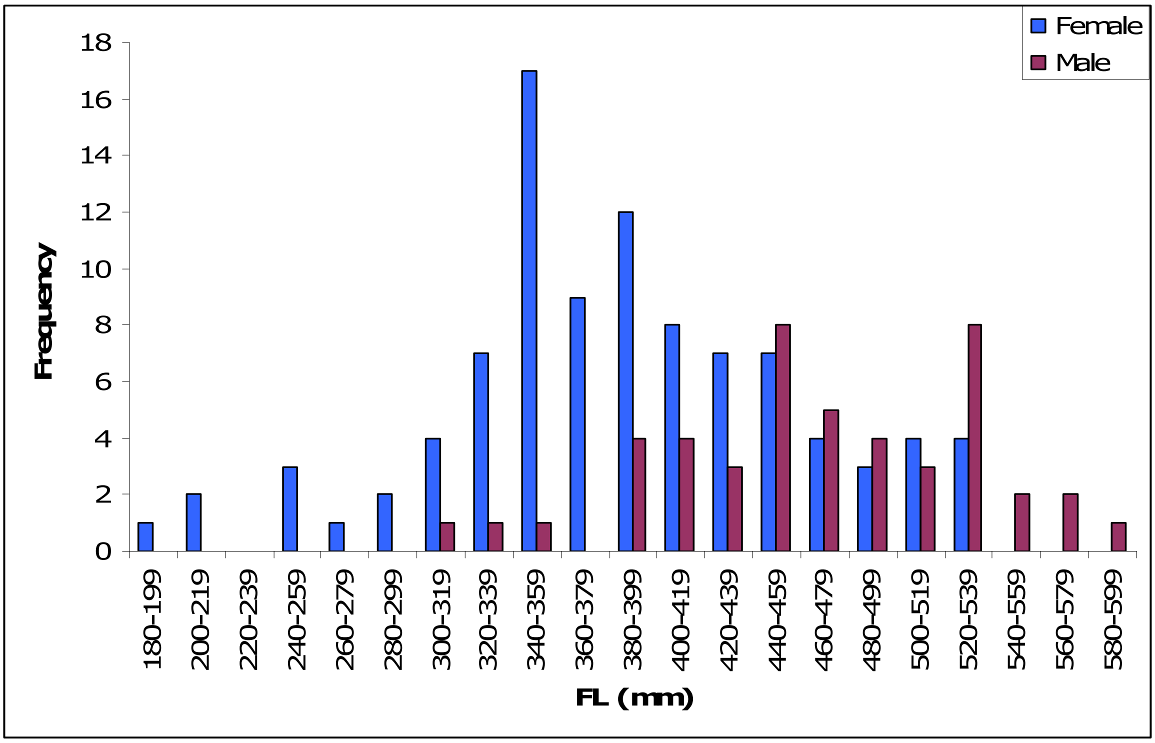

Figure 5.9. Size-frequency distribution of *Chlorurus microrhinos* partitioned by sex.

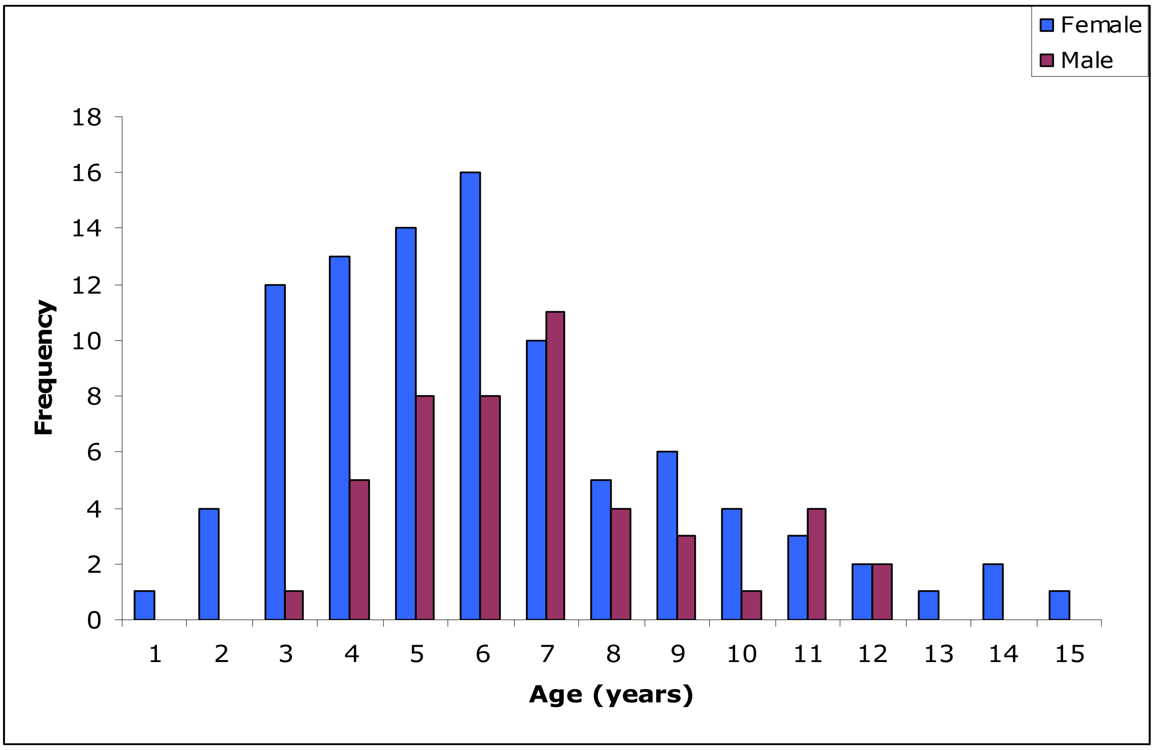

Figure 5.10. Age-frequency distribution of *Chlorurus microrhinos* partitioned by sex.

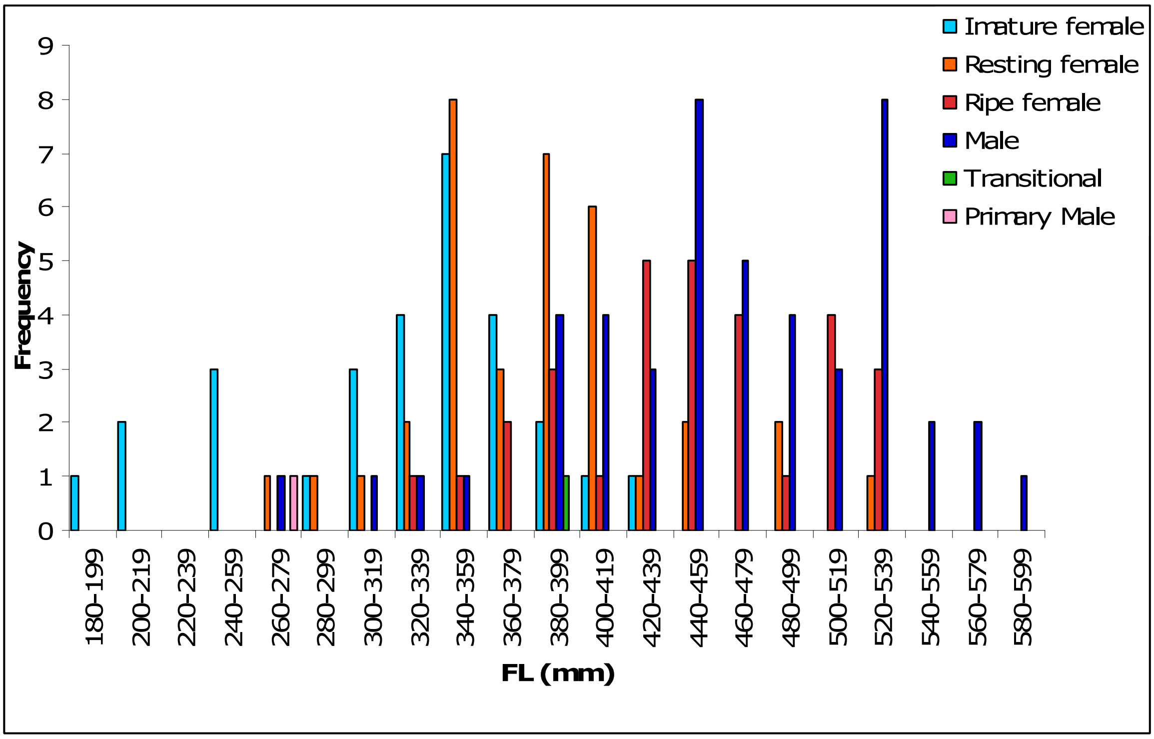

Figure 5.11. Size-frequency distribution of *Chlorurus microrhinos* partitioned by gonad development stage.

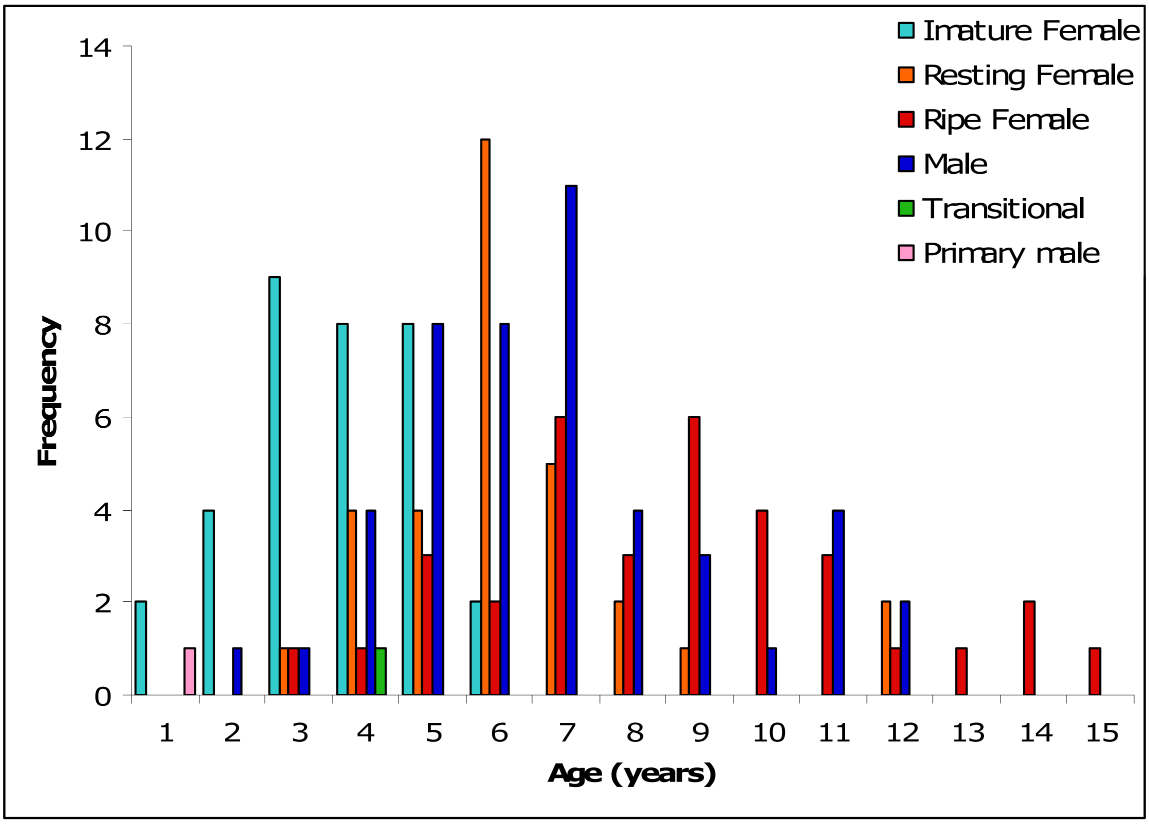

Figure 5.12. Age-frequency distribution of *Chlorurus microrhinos* partitioned by gonad development stage.

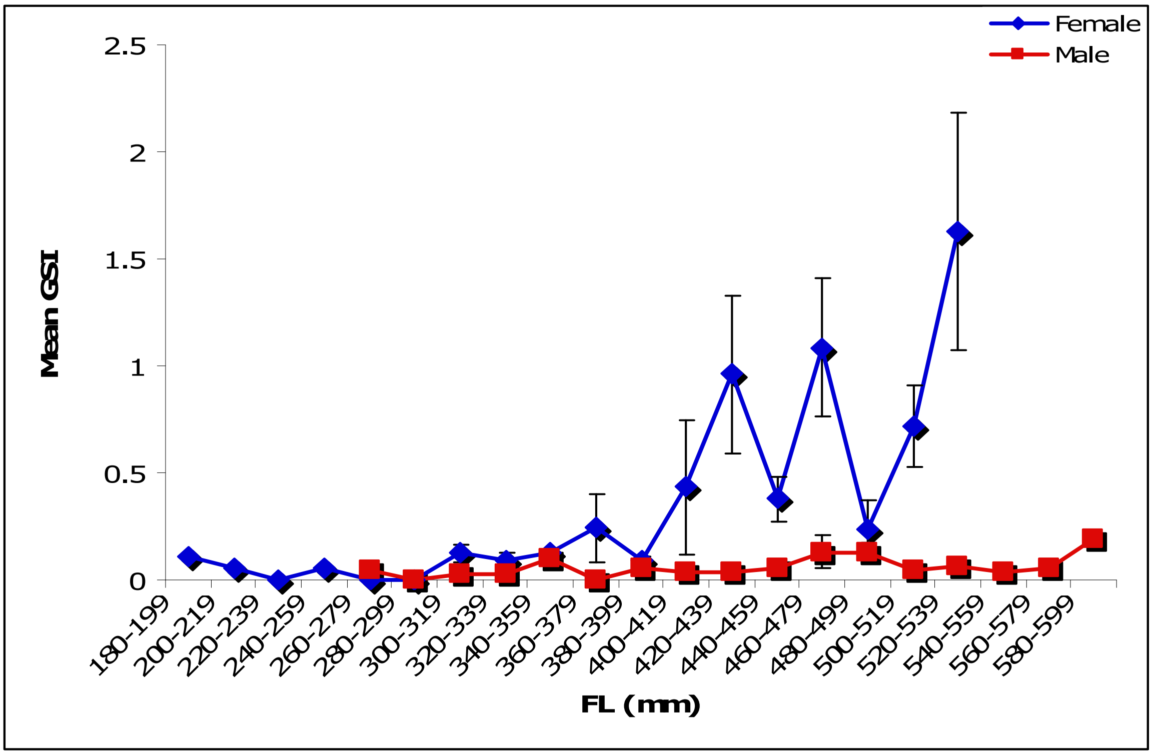

Figure 5.13. Gonad somatic indices for *Chlorurus microrhinos* versus fork length partitioned by sex.

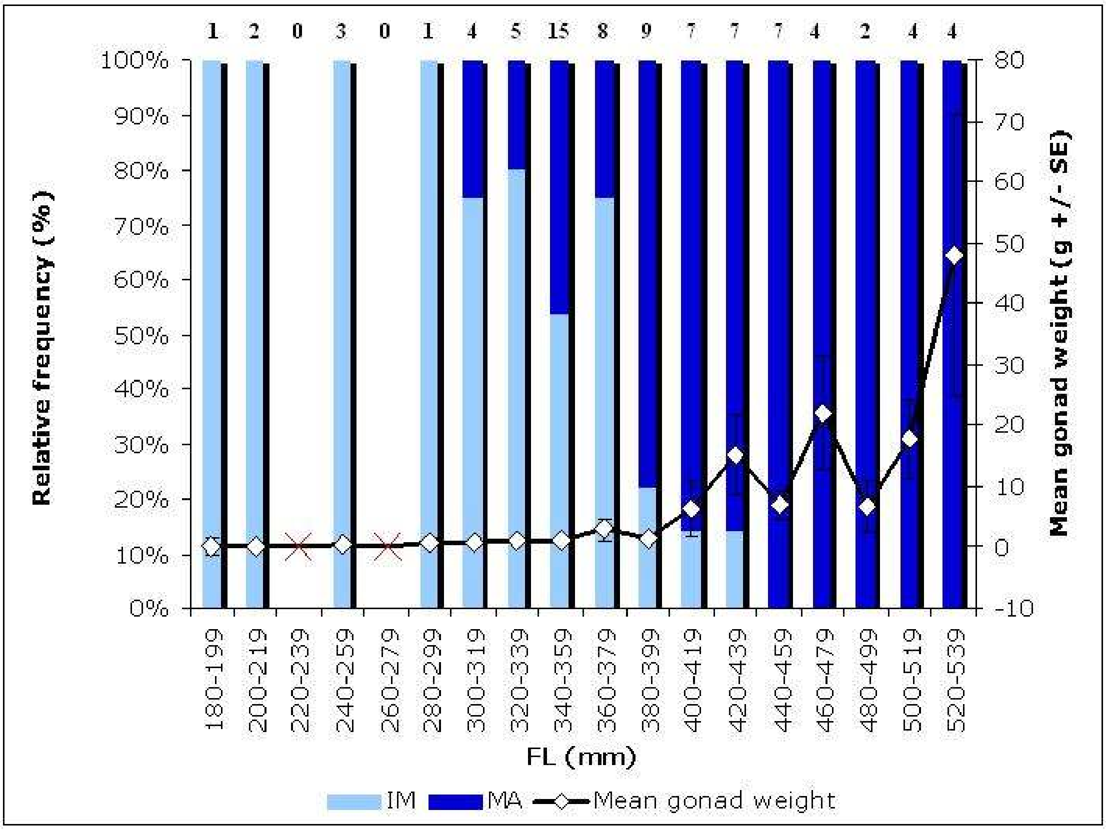

Figure 5.14. Plot of *Chlorurus microrhinos* gonad weight decoupled as immature (IM) and mature (MA) versus size distribution. Crosses indicate no data. Data points with no visible error bars are either made up of 1 sample or their standard error is too small to be registered on the graph. Sample size indicated on top of each bar.

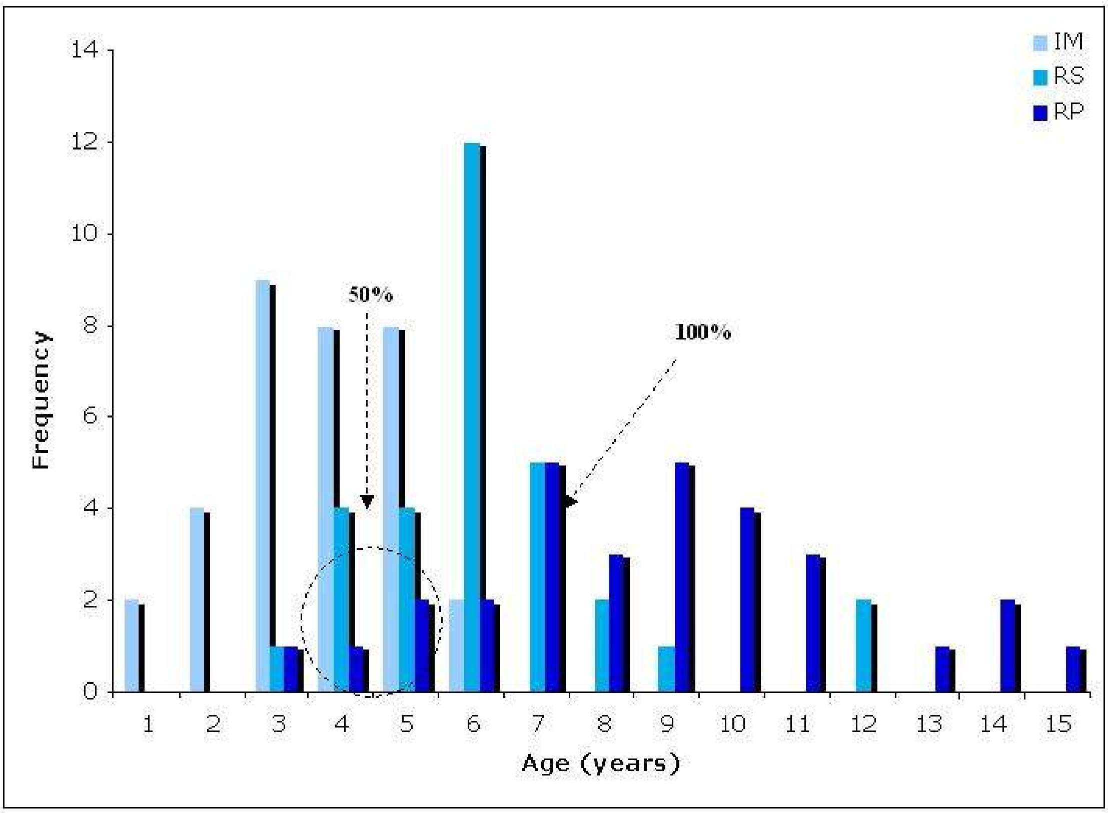

Figure 5.15. Age distribution of *Chlorurus microrhinos* female reproductive stages with estimated 50% and 100% age at maturity indicated.

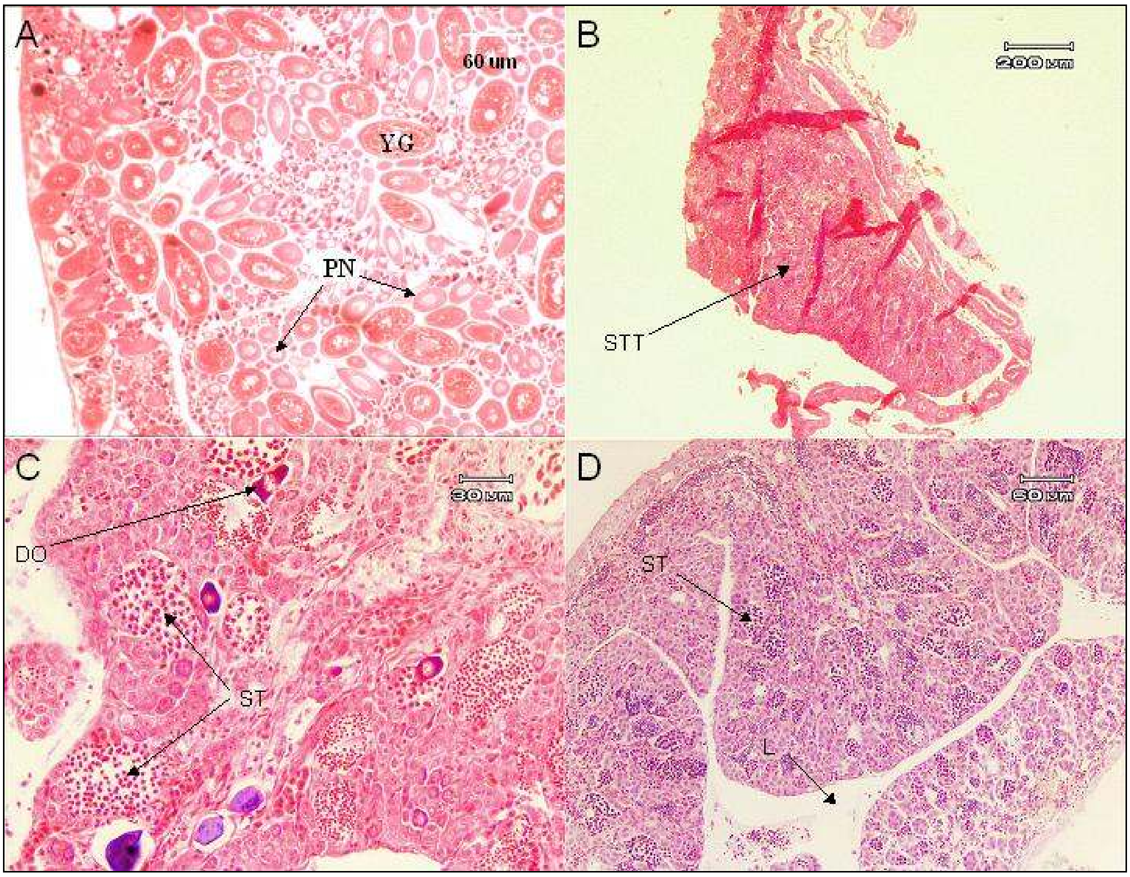

Figure 5.16. Evidence of sex change and dual male development pathway in Chlorurus microrhinos. (A) Mature resting female with occyte development, (B) Immature male with no prior sign of female function, (C) Transitional *male*, (D) Mature male with ovarian features. STT: spermatogenic tissue, DO: degenerative oocyte, PN: perinucleolar stage oocytes ST: spermatids, L: ovary lumen, YG: yolk globule stage occyte.

5.3.2.1 Summary

Growth trajectories partitioned by sex confirmed that males on average had larger sizes at equivalent age (Figure 5.8). The sex ratio was calculated to be female dominated with approximately 2F:1M ratio, while overlap of both sexes occurred between 300-539 FL(mm) and 3-12 yrs, respectively (Figure 5.9 and 5.10). Age distribution portioned by sex reveal that males start appearing in the younger age classes, while mature females persist through all age classes, appearing as the oldest individuals in this dataset. As discussed previously, the presence of mature old females is indicative of a healthy population. As expected, females are the oldest individuals in this population, while males are the largest. The comparatively longer longevity of *C.microrhinos* should theoretically allow more time for each life-history stage. This is reflected in the higher numbers of immature females present in the total sample (n=33; 23%) compared with the *S.ghobban* population (n=1; 2.2%).

An increase in the proportion of mature females in the *C. microrhinos* sample did not have an immediate impact on mean ovary weight. This was due to the higher presence of RS females (developing or resting stages) within the mature stages in the relevant size classes. Based on the ratio of immature to mature individuals, female are estimated to surpass 50% maturity from 360 mm and age range of 4-5 years, while 100% maturity is achieved at a larger approximate size of 400 mm and older age of 7 years (Figures 5.14 & 5.15). Female GSI values markedly increased upon reaching the same size range, indicating an increase in reproductive development (Figure 5.13). The GSI of females was significantly higher than males ($t=3.54$, $P<0.05$)

Mature male and female gonads were relatively easy to identify macroscopically. Ripe ovaries and testes were orange and white in colour, respectively. Gonadal development stage portioned by size and age (Figures 5.11 and 5.12) reveal several interesting characteristics. A transitional individual was estimated to be 4 yrs old and 381 mm(FL) in size exhibiting degenerating previtellogenic oocyte and proliferation of spermatids. The appearance of a single transitional individual in a relatively large sample size (n=143) is significant. Compared with *S.ghobban* (2.22%), the percentage of transitional individuals in the *C.microrhinos* dataset was significantly lower (0.7%). These figures may be significant given the fact that they represent two vastly different regions in terms of fishing pressure.

The presence of an immature primary male, 1 yr and 270mm(FL), with no signs of prior spawning (Figure 5.14B) is in conjunction with the transitional specimen (Figure 5.16C) an indication of the dual male development trait of *C.microrhinos*. We can therefore confidently assume that *C.microrhinos* has been shown to be a diandric protogynous hermaphrodite

5.4 Discussion

Female *C.microrhinos* were estimated to reach 100% maturity at approximately 400 mm(FL) and 7 years of age. It was impossible to calculate the same parameters for *S.ghobban*, as only a single immature female was ever established. Based on the available data, we must assume that *S.ghobban* are 100% mature at 240mm(FL) and 1 year of age. Given the lack of immature females, and subsequent establishment of early maturation, I was unable to ascertain female size and age at first maturity for *S.ghobban*. Histological analysis has produced several interesting points which in conjunction with demographic and fishery information warrant closer examination.

Histological evidence of a single transitional individual, along with a primary male, indicates that *S.ghobban* is a diandric protogynous hermaphrodite. Although mature females and males have been identified as young as one year of age, mature females have been in general much smaller in size. Based on age distribution, indications are that transition from female to males occurs before the third year of life, as no females older than 3 yrs were observed. However, the presence of ripe, young and very small females indicates that female maturity is reached extremely early. Histological analysis for *C.microrhinos* from the GBR also revealed a transitional and primary male individual. The fact that transitional specimens were located in both datasets is a surprise given the rapid speed with which protogynous species change from one sex to another (Gust 2000). Based on the evidence of transitional and primary specimens, *C.microrhinos* is suggested to possess dual male development pathways. In conclusion, both populations are diandric protogynous hermaphrodites based on the presence of both primary and secondary males, and transitional specimens.

In terms of spawning periodicity, estimations are at best speculative. However, macroscopic analysis showed that *S.ghobban* were reproductively active across the duration of the experimental period. It is thus likely that *S.ghobban* are serial spawners. However, this is difficult to accurately estimates as most long-term reproductive studies do not last beyond a season. According to studies cited by Claydon (2004), both *C.microrhinos* and *S.ghobban* aggregate to spawn. However, their spawning periodicity has not been fully confirmed yet. Claydon (2004) highlights four levels of periodicity in spawning aggregation: seasonal, lunar, diel, and tidal. However, he makes an interesting observation that many transient spawning aggregations are formed in association with states of the lunar cycle. Hamilton (2004), ascertained the spawning periodicity of *B.muricatum* in the western solomons as being protracted and limited to “four-five days after the full moon until the early stages of the last quarter”.

The fact that *S.ghobban* is probably a serial spawner has implications from a management perspective. Obviously, regular female reproductive output contributes towards higher fecundity. This characteristic may also be an indication of the short life cycle of this species, which means that reproductive activity must be regular in order to maximize regenerative contribution to the population.

Comparative demographic analysis (chapter 4) has shown that the longevity of *S.ghobban* in the Solomon Islands is relatively short (5yrs) compared with other locations such as Oman (12yrs) or Taiwan (15yrs). However, *S.ghobban* from the Solomons has been shown to achieve relatively faster growth at equivalent age compared with other locations. From this information we are able to ascertain that *S.ghobban* is a fast growing and short lived parrotfish in this region. The short life-span of *S.ghobban* means that the reproductive cycle is extremely compact, indicating that fish do not have extended time as males or females. Further to this, the population sample in this chapter came from a region where fishing pressure is relatively high (chapter 2), which may have some impact on reproductive output and other life-history parameters. However, without comparative data from the same region, this statement is at best speculative.

Males appear very early in the *S.ghobban* population, while *C.microrhinos* exhibit the same trait. Given the higher longevity of *C.microrhinos* (estimate at 15yrs in this dataset), the appearance of mature males as young as two years old appear to suggest that primary males are maturing from immature females. Males either changed sex from mature females or from immature female ovaries in this population. The latter scenario was shown in *B.muricatum* by Hamilton (2004, 2008) This remains the most plausible explanation given the fact that smaller and younger males would obviously not have had the chance of prior function as female.

Both *S.ghobban* and *C.microrhinos* exhibit male-biased sexual size dimorphism. According to Adams (2002) and Munday et al., (2004), this trait could be attributed to faster initial growth rates where differences in growth established during initial somatic development may determine the sex of the individual. Alternatively, Shine (1994) and Choat et al., (1996) put forward the argument that males have more energy to spend on somatic growth after maturation or sex change. The notion that females require more energy input for reproductive development has already been highlighted in this chapter (Hart and Reynolds 2002; Pitcher and Hart 1982).

The *C.microrhinos* dataset was female dominated, indicative of historically low fishing pressure, while the older age classes in the *S.ghobban* population were male dominated, which perhaps is an indication of historically higher fishing pressure. However, the fact that the mature females dominate the older age classes is an interesting and contrasting characteristics between the GBR and Solomons populations. Begg et al., (2005), state that a broader age and length distribution, and older females, are indicators of a healthy population which is not being subjected to heavy fishing pressure. This fact can certainly be corroborated with regard to the GBR, where the persistence of females through all age classes is a reflection of light fishing pressure. From a reproductive perspective it would make sense for a female to grow old and thus increase its reproductive output. If however, the larger and older individuals are being fished out of a population, that is bound to have demographic consequences.

According to (Palumbi 2001) fishing pressure has the ability to force fish to change sex at smaller sizes by favouring such genotype. The actual mechanism which triggers sex change is not known in scarids, however, since they form breeding territories and social harem with dominant territorial males (Streelman et al. 2002), a reduction in larger males is likely to trigger a sex change in smaller females in order to maintain optimum sex ratios (Sadovy and Shapiro 1987). The prevailing consensus on the impact of chronic fishing pressure is that earlier sex change occurs as a result of the removal of larger males leading to a skewed population distribution favouring females (Polunin and Graham 2003). Hawkings and Roberts (2003) studied the effects of fishing pressure on Caribbean parrotfishes and found that the removal of large terminal phase males was triggering initial phase fish to change sex at smaller sizes and thus impact on reproductive success.

The effects of fishing pressure on the reproductive parameters of *S.ghobban* from Gizo are at best speculative. Given the fact that these species are probably short-lived, dictates the time-line when important life-history events can happen. As a result, initial female sexual maturity, sex-change, and male reproductive output are controlled by the natural short longevity of *S.ghobban*. Claro et al., (2001), suggest that sex change at small sizes may be due to genetic factors in the Caribbean hogfish (*Lachnolaimnnus maximus*), excluding fishing related inducement.

Polunin and Graham (2003) state that exploitation can ultimately affect life-history traits. They point to numerous studies which show that chronic intense fishing results in earlier age of maturation, slower growth, skewing of sex ration, reduced egg size, and changes in morphological traits. However, more importantly, Polunin and Graham (2003), along with other authors (Jennings et al. 1999; Dulvy et al. 2003,;Sadovy et al. 2003a) point out that life-history traits also affect vulnerability to exploitation. The central notion here is that species with slow regenerative capacities are more vulnerable to over-fishing than high turnover species. While that may be true in some circumstances, it appears that the apparent high turnover capacity of *S.ghobban* from Solomons Islands is not providing sufficient protection from quantified fishing efforts (chapter 2) which have caused density declines (chapter 3) in parrotfish stock numbers. Having established that parrotfishes are extremely vulnerable to over-exploitation, it appears that not even their apparent high turnover life-history trait can provide sufficient protection.

According to Smith et al., (1998) and Hutchings (2000), age at first maturity, and not fecundity (as proposed by Jennings et al. 1999; Dulvy et al. 2003, Sadovy et al. 2003a), is a strong determinant of resilience factor to over-exploitation, which ultimately controls the rebound potential of species. According to Denny et al., (2002), small body size is associated with early maturity. Regardless of which life-history trait is responsible for rebound potential, none can be applied to *S.ghobban* from Solomon Islands. Firstly, we have shown through histological analysis that this species matures very early and at small sizes. Secondly, this species reaches relatively large body length (350mm in this dataset) within a few short year (chapter 4). Finally, its high fecundity is also supported by its serial spawning periodicity. Theoretically, *S.ghobban* should show some resilience to fishing pressure. This ability has not been observed. As a result, using “surrogates” of life-history traits, as advocated by Polunin and Graham (2003), to generalise certain characteristics must carefully be utilised on a case-by-case basis.

This chapter has contributed to the knowledge-base of parrotfish reproductive characteristics. However, it is imperative that further spatially explicit and comparative sampling are carried out, not only to ascertain reproductive pathways in scarine labrids, but also to determine the magnitude of variation in life-history parameters in relation to ongoing fishing pressure.

Chapter Six

The value of Marine Protected Areas for protecting coral reefs in Melanesia, a case-study from Solomon Islands

The findings of this chapter have been published in: Aswani S, Albert S, Sabetian A, Furusawa T (2007). Customary management as precautionary and adaptive principles for protecting coral reefs in Oceania. Coral Reefs 26(4): 1009- 1021. The above work represents a collaborative investigation of the Nusa Hope/Heloro MPA, as part of a larger marine science and ethnographic monitoring program for the Vonavona and Roviana marine protected areas project. This chapter presents the marine science component of our findings conducted by Simon Albert and myself.

6.0 Introduction

Coral reef fisheries play a vital role in the life of Solomon Islands communities, most of which are coastal. Solomon Islanders have one of the highest per capita seafood consumption rates in the world with over 80% of the population deriving their protein from marine resources (Sabetian and Afzal 2004). Fisheries resources therefore are critical to livelihoods and food security, particularly when a monetary value is assigned. The sale of marine resources collected artisanaly currently provides the second highest foreign exchange earnings for the nation (Sabetian andAfzal, 2004).

The Solomons also has one of the highest annual population growth rates in the world (2.8%) and its current population of approximately half a million is expected to double in less than 20 years (Otter 2002; Sabetian and Afzal 2004). The population distribution across the provinces is very uneven with over 122 thousand inhabiting Malaita representing 30 percent of the Solomon Islands population whereas Rennell-Bellona accommodates approximately 3000 people equating to less than one percent of the population (Sabetian and Afzal 2004). Thus, human related impacts such as over-fishing are variable across the islands, with heavier impacts presumed in areas of higher population and in urban/peri-urban areas. The growing population, especially around urban areas such as the capital Honiara and western provincial capital Gizo, coupled with a heavy dependence on marine resources for subsistence and cash has been shown to put increasing pressure on near shore marine resources (chapters 2 & 3).

Traditionally, fisheries are managed either through regulatory measures which focus on the resource users and owners, and/or through the protection of the targeted species. Johannes et al., (1993) pointed to the fact that centralised government fisheries have had a tainted history in Oceania over several decades. There are many factors that can attribute to failure, including inadequate scientific models, environmental variability, ignorance about natural systems, poor data, non-compliance with management measures, and the complex inter-relationships between biological, economic, and cultural systems (Sabetian and Afzal, 2004). But more importantly, the centralised management of small-scale multi-species multi-method fisheries that are spread over thousands of kilometres is too complex and cost-ineffective for small Pacific Islands nations to carry out effectively. The fact is that coral reef fisheries contribute significantly to low-income fishers with few alternative opportunities for employment, thus conventional controls on catch and effort is difficult to socially justify and also administer (Russ and Alcala 1996). Furthermore, the implementation of conventional western conservation and management systems in areas where customary marine tenure applies is seriously disadvantaged and prone to failure from the outset. Therefore, the approach to setting up conservation and management systems in these regions must be essentially pragmatic, holistic, and integrative. Marine Protected Areas (MPA) have often been a primary tool in integrated coastal management (Christie 2005a-b).

6.1 The case for Marine Protected Areas (MPA)

Marine protected areas or marine reserves are increasingly being advocated as effective management or conservation tools (Halpern and Warner 2002). The effectiveness of MPA as a conservation and management tool is a controversial issue. Halpern and

Warner (2002), conducted a meta-analysis of MPA reports and publications, which showed that the higher average values of density, biomass, average organism size and diversity inside reserves (relative to control groups) improved rapidly. However, this early increase can be misleading as full recoveries can take decades (Russ and Alcala 2004). Although the majority of the reviewed studies by Halpern and Warner (2002) did report positively in terms of initial increased fish abundance, fishery benefits of MPA remain controversial (Roberts et al. 2001). This is mainly based on the lack of empirical data on rates and patterns of increase in density and biomass of artisanaly to commercially target species.

6.2 The Roviana and Vonavona marine protected network

Although discussions around the issue of MPA “fishery benefits" is controversial, their conservation and rehabilitation benefits are less contentious and increasingly being recognised. MPA are getting increasing support in the Caribbean as a primary conservation tool to deal with regions suffering from years of overfishing on herbivorous species which has been directly linked to cascading negative effects on coral reefs and loss of income through tourism (DPNR 2005; Hawkins et al. 2006).

The Roviana and Vonavona marine protected areas project, led by Professor Shankar Aswani from University of Santa Barbara California, is the biggest MPA initiative in the Solomon Islands with over 23 no-take reserves (Appendix 4) having been established since 2003 where resource-harvesting activities are forbidden or restricted. This project will be the most comprehensive integrated resource management initiative in Solomon Islands to date, finally tackling the important question of MPA and their role in ICM systems. I established a collaborative relationship with Professor Aswani, as part of a larger monitoring initiative by a group of marine scientists.

6.3 Objectives

As stated in chapter one, the aim was to investigate the effectiveness of MPA as a parrotfish conservation tool. The specific objectives were to:

1) Collect information on species composition, density and size distribution of fishes within and outside of the Nusa Hope reserve.
2) Assess the rate of herbivory within and outside of the Nusa Hope reserve.

My intention was to acquire information that would complement the herbivory study carried out by another researcher, Mr Simon Albert. This herbivory study contained both grazing and fish abundance experiments. Specifically, algal settlement tiles were used to assess the fish grazing intensity of herbivores both within and outside of an MPA. To supplement the algal indicator data, fish and coral reef conditions were also quantified using a conventional underwater visual census (UVC). For my part, density and size-distribution of several parrotfish species were measured in the same location (outside vs inside) in order to provide another independent dataset. The results of these findings are presented herein. As can be seen, our plan was not to assess the fishery benefits of MPA, rather to examine the suitability of protection as a form of conservation for depleted inner lagoon coral reef ecosystems.

6.4 Methods and Materials

The Inner lagoon reefs of Vonavona and Roviana are generally near large landmasses or villages on isolated raised coral islands. These reefs are periodically affected by rain, which results in large discharges of fresh water carrying sediments flowing over and encrusting coral reefs. The general pattern within the lagoon are coral reefs with a total coral cover lower than the a-biotic cover. This is to be expected where the reefs are subjected to high sedimentation and turbidity levels. High level of sedimentation causes an increase in turbidity reducing the level of sunlight that reaches the coral and increasing the metabolic costs of corals resulting in coral dying. The study sites, the Nusa-Hope community base MPA (Figure 6.1), also suffer from algae encrustation, but contains areas of concentrated collective coral growth descending down to 25 meters at 70 degree reef drops.

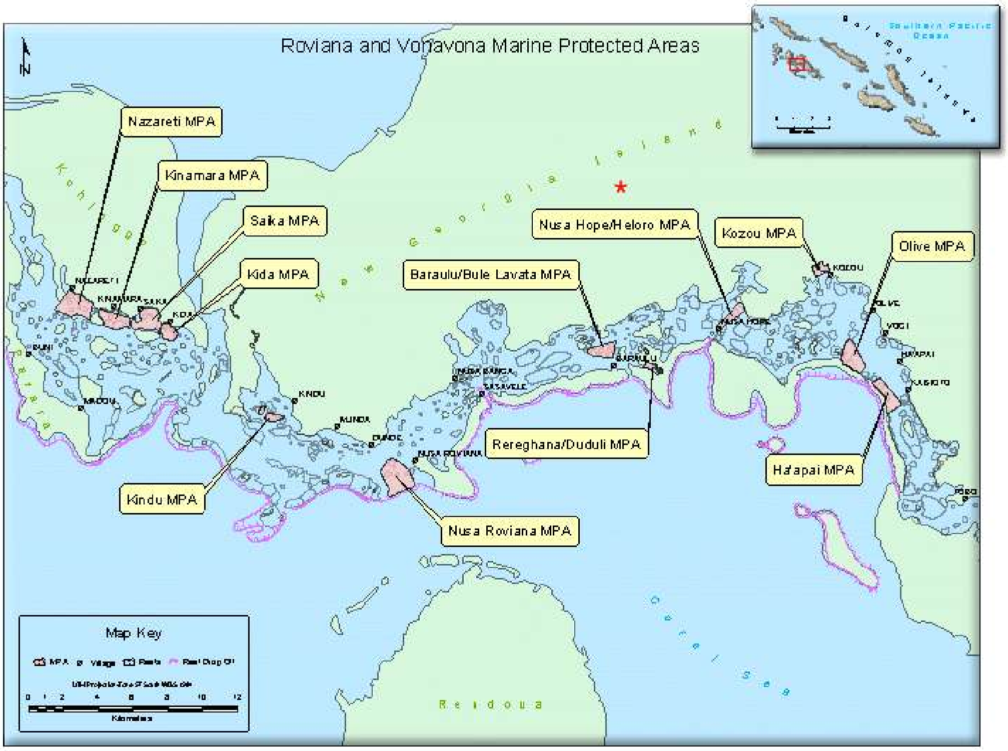

Figure 6.1. Map of 12 Reserves within the Roviana and Vonavona marine protected areas project. Nusa Hope/Heloro MPA (*).

First, algal settlement tiles were used to assess the fish grazing intensity of herbivorous fish. An area of reef (≅50 x 50 m^2) inside the MPA and a second area outside the MPA were used as experiment and control sites, respectively. The protected lagoon reefs at each site are representative of the majority of the MPA area, consisting of coral and algal communities growing on predominantly carbonate bedrock interspersed with sandy areas at a depth of approximately 1 meter. The hydrodynamics and water quality were similar at both sites (in terms of distance to clean oceanic water flushing, distance from terrestrial run-off, and similarity of current velocity). Five replicate settlement tiles (7X7 cm^2 PVC; Smith et al. 2001) were attached to concrete blocks and randomly placed on the reef both inside and outside the MPA and collected after six months. Cover of algal functional groups (Steneck and Dethier 1994) was quantified at 24 random points on a digital image of the tiles using Coral Point Count V3.2 (NCRI). Algal biomass (dry weight) on the top surface of the tiles was quantified following oven drying at 60°C. To verify the algal indicator data, fish abundance (St. John et al. 1990) were quantified along three 50 m transects at 1-m depth within each area. Coral reef condition was analysed along the same three 50 m transects at each site by taking a 1 m^2 digital photo every 2 m and classifying benthos as live coral, dead coral, macroalgae, sponge, substrate (sand) or rock at 24 random points on each digital image using Coral Point Count V3.2 (NCRI). The small number of transects for verifying the algal study resulted from a combination of logistical problems and the participatory nature of this research, which involved local communities in actual research and monitoring.

For the second survey, a strip-transect UVC technique was employed to measure abundance, size-frequency, and spatial distribution patterns for parrotfishes. The decision to focus on this particular family was based on the critical functional role of parrotfish herbivory on the health of shallow water coral reef ecosystems (Hughes 1994; Hawkins and Roberts 2004), their role in maintaining coral reef health within MPAs (Mumby et al. 2006, 2007), due to the fact that they are a main target of Roviana fishermen (Aswani and Hamilton 2004), and to complement the algal study.

Underwater Visual Census monitoring within the Nusa Hope MPA was compounded by the fact that the reserve covered a range of shallow biogeographical regions, which made conducting transects difficult, and the visibility was often unsuitable. Initial investigation revealed that the most representative parrotfish habitat, which occurred at both inside and outside the Nusa Hope MPA, were reef slopes adjacent to the passage. Parrotfish abundance was quantified along 18 separate 100 x 10 m^2 strip transects inside and 18 more outside the MPA at 20 m depth (because visibility was best at this depth). Land reference points were used to ensure transects did not overlap. Parrotfishes were recorded down to species level for excavating and scraping individuals, and comparisons made in this paper concern the differences between parrotfish on the reef slope inside and outside of the MPA.

To meet the assumptions of ANOVA, data from algal tiles, benthic cover and coral reef condition transects, and fish surveys were square-root transformed before conducting ANOVA (Tukey 1977; Keppel 1991; Howell 2002). Exploratory data analysis (Tukey 1977) demonstrated that square root expression of these variables was superior to linear or logarithmic transformations in reducing heterogeneity of variance between groups, and in some instances, ameliorating skew. Data, tables, and figures display untransformed values.

6.4.1 Statistical analysis

Statistical analysis was made using SPSS (Version 11.02, SPSS Inc., Chicago, IL, USA) and Microsoft Excel (Microsoft Corp, Redmond, WA, USA) software, and comparisons were considered to be statistically significant at $P < 0.05$. Values are expressed as mean ± standard error of the mean (SE) unless otherwise noted.

6.5 Results

In the algal study, the top surfaces of the settlement tiles from within the MPA were colonized by a diverse algal community that included cyanobacteria (20% ± 3.6; mean ± SE), filamentous algae (2% ± 1.7), foliose algae (23% ± 2.8), and crustose coralline (23% ± 6.0) (Figure 6.2).

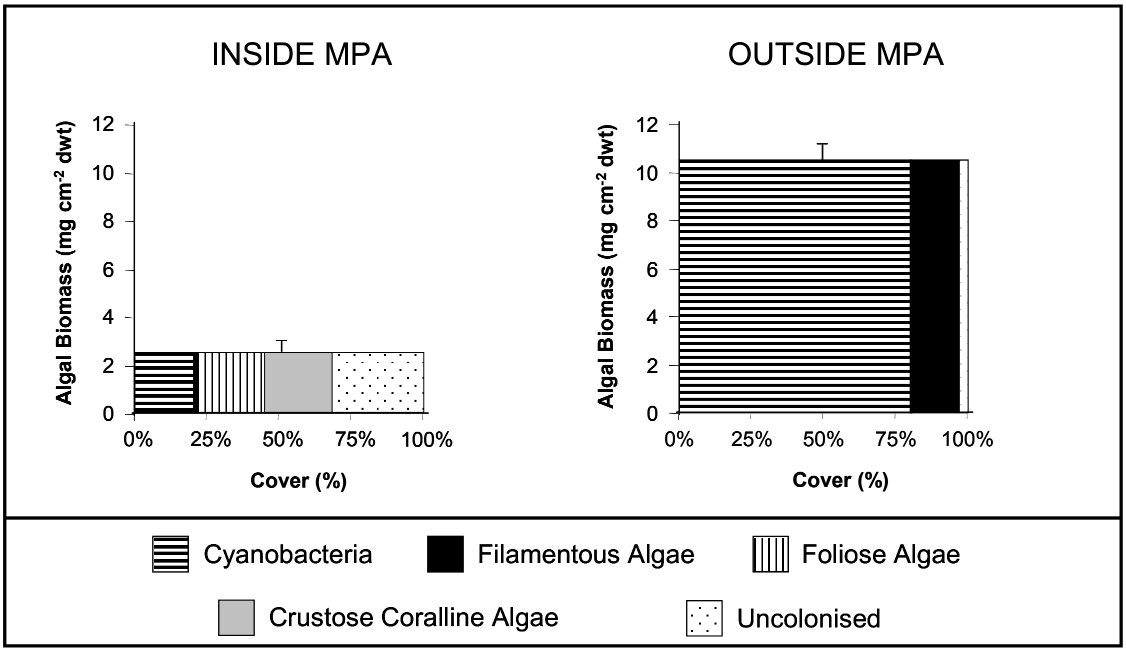

Figure 6.2. Biomass and percentage cover of algal communities on the top surface of settlement tiles.

The remaining 32% (±2.1) of the MPA tiles were uncolonized by algae (visible to the naked eye). Fish grazing marks were evident on the tiles within the MPA, suggesting continual removal of both plants and animals that settled on the tiles. In contrast, the settlement tiles in the adjacent reef outside the MPA were colonized predominately by the cyanobacteria *Oscillatoria* spp. (80% ± 4.0) with a smaller area covered by filamentous algae (17% ± 2.9). An average of only 3% (±2) of the tiles collected from outside the MPA was uncolonized (to the naked eye) (Fig. 2). There was also little evidence of grazing marks on the tiles, which suggests that the frequency of substrate removal was less outside the MPA. Specifically, there was a significant difference between the composition of algal groups covering the settlement tiles inside and outside the MPA, as indicated by a significant location x cover interaction (F(7,64) = 70.2, P < 0.0001). Most apparent, the microalgae (Cyanobacteria) covered 80 ± 4.0% (mean ± SE) of the tiles outside the MPA compared to 20.0 ± 3.6% inside the MPA, F(1,8) = 85.7, P < 0.000001. In addition, there was also a significantly (F(1,8) = 21.3, P < 0.005) higher biomass of algae from outside the MPA (10.8 ± 1.8 mg cm^{-2}) compared with inside (2.5 ± 0.4 mg cm¡2).

The data from the complementary visual census of the herbivorous fish were equivocal. Inside the MPA there were 134 ± 25 fish per 200 m^2 (mean ± SE) compared to 61 ± 18 outside the MPA. However this apparent difference was not statistically significant (F(1,4) = 5.8, P = 0.08) probably because of the small sample size (three transects per group). Total fish abundance inside the MPA was 247 ± 31 per 200 m^2 compared to outside where it was 175 ± 31 per 200 m^2, but this difference was also not statistically significant, F(1,4) = 2.8, P = 0.17. There was significantly (F(1,4) = 12.5, P = 0.02) higher macroalgal cover on the reef area located outside the MPA (6.76% § 2.7) compared with the area inside the MPA (0.06% ± 0.6) (Table 6.1). The macroalgae consisted primarily of corticated groups such as *Caulerpa* spp. growing on dead coral and rocks. There were no significant differences for other benthic communities between the two areas.

	Inside Nusa Hope MPA		**Outside Nusa Hope MPA**		
	MEAN	±SE	MEAN	±SE	p value
LIVE CORAL	5.07	±1.26	4.69	±0.29	0.85
DEAD CORAL	1.33	±0.33	2.09	±1.08	0.69
MACROALGAE	0.06	±0.06	6.76	±2.70	**0.02 ***
SPONGE	1.05	±0.35	2.38	±1.11	0.29
SUBSTRATE	54.8	±8.32	58.2	±3.64	0.69
ROCK	37.7	±7.31	25.9	±2.77	0.21

Table 6.1. Percentage cover of Nusa Hope reef communities within the marine protected area (MPA) and OPEN sites outside the MPA. There was a significant interaction between MPA and benthic cover (ANOVA *F* (5,24) = 4.2, p < 0.01). Significant p values are shown in bold and indicated by a star.

The parrotfish survey revealed that there were more small (<30 cm) parrotfish inside the MPA (38.7 + 3.7) compared to outside the MPA (18.4 + 1.9), F(1,34) = 25.7, P < 0.0001, and importantly, there were >10X more large parrotfish (>30 cm) inside the MPA (9.8 + 1.1) compared to outside the MPA (0.7 + 0.2), F(1,34) = 101.0, P < 0.0001 (Figure 6.3). More specifically, inside the reserve there was a higher number of Pacific steephead parrotfish (*Chlorurus microrhinos*) (4.8 + 1.1 inside, 0.0 outside; F(1,34) = 20.3, P < 0.0001) and bumphead parrotfish (*Bolbometopon muricatum*) (2.4 + 0.7 inside, 0.0 outside; F(1,34) = 14.1, P < 0.001), both of which are excavating species (Fig. 6.4).

Smaller generalist scraping species were also significantly more abundant inside the reserve, including *Scarus ghobban* (1.9 + 0.5 inside, 0.1 + 0.1 outside; F(1,34) = 24.5, P < 0.00002), *Hipposcarus longiceps* (12.0 + 1.7 inside, 6.7 + 1.2 outside; F(1,34) = 8.0, P < 0.01), and other assorted species (*Scarus bleekeri, Scarus oviceps, Scarus globiceps, Scarus quoyi, and Scarus schlegeli*) (14.8 + 2.0 inside, 5.9 + 1.1 outside; F(1,34) = 16.1, P < 0.0005) (Fig. 4).

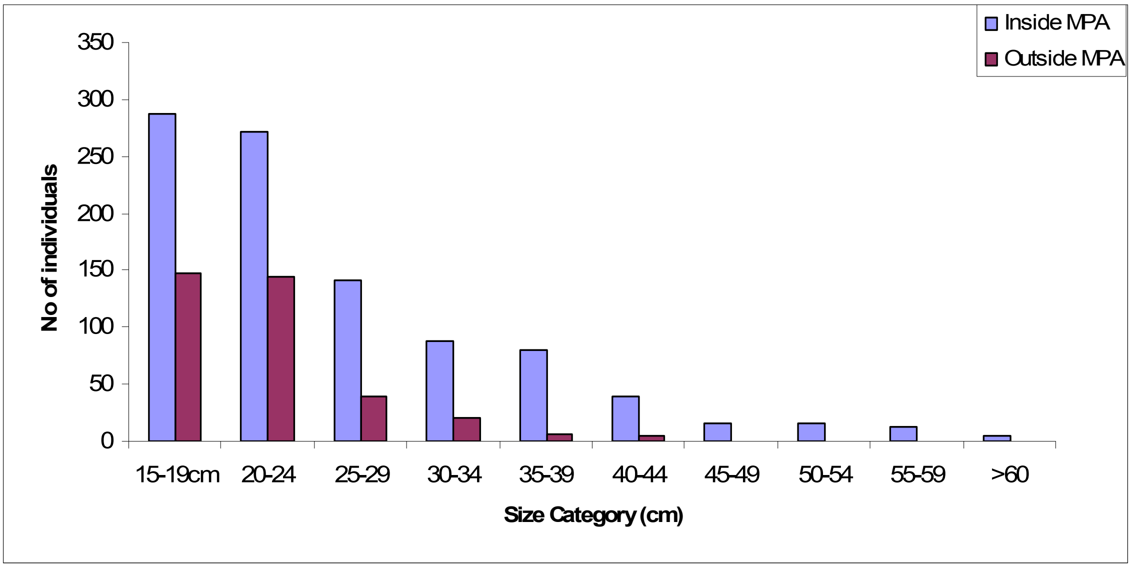

Figure 6.3. Size-frequency distribution for all parrotfishes inside/outside of the Nusa Hope marine protected area.

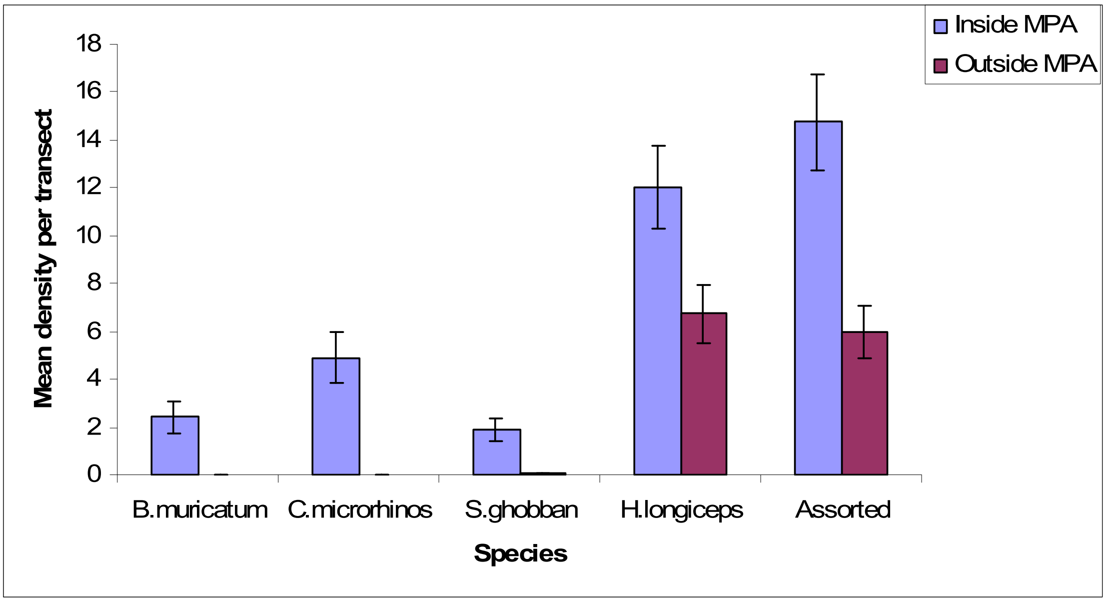

Figure 6.4. Species specific mean density per transect for parrotfish inside/outside of Nusa Hope marine protected area (*error bars represent* ± *1SE*)

6.6 Discussion

The combined results from the algal tiles and the fish surveys suggest a positive trend toward greater abundance of grazing fish, particularly parrotfish, within the Nusa Hope MPA compared to the adjacent area which is open to local fishing pressures. The link between algae communities and herbivore abundance has been well documented in other parts of the world (e.g. Mantyka and Bellwood 2007). Coral reef algal communities are primarily controlled by top-down (grazing disturbance) and bottom-up (nutrient stimulation) processes (McCook 1999; Smith et al. 2001). The balance between nutrients and grazing is essential in maintaining a healthy reef ecosystem. Numerous studies report that reductions in herbivore populations, especially due to fishing, cause an increase in algal biomass, which subsequently leads to (or which has been suggested to lead to) a decline in the health and condition of coral reefs (Stimson et al. 2001; Thacker et al. 2001). At Nusa Hope Village, parrotfish as well as other herbivorous fish species are targeted by fishermen outside the MPA with mass-harvesting techniques, in particular night spearfishing, and to a lesser extent with fish poisoning, netting, and fish-drives (Aswani and Hamilton 2004), increasing the susceptibility of these reefs to algal overgrowth.

The study of algal communities is a first step toward developing a spatially and temporally integrated measure of herbivorous fish at experiment and control sites in this region. Low biomass communities of grazing-resistant algae were present on tiles within the MPA. The foliose and crustose coralline algae that dominated tiles in the MPA site are typical of a healthy reef system with high grazing rates (Steneck and Dethier 1994). In contrast, outside the MPA the algal biomass on tiles was 400% higher than inside the MPA and consisted primarily of cyanobacteria and brown filamentous algae, which are typical of a stressed reef suffering from an imbalance between nutrient availability and grazing pressure (Steneck and Dethier 1994). Because coral and macroalgae compete for space on coral reefs, coral planulae cannot settle on macroalgae, which ultimately reduces coral growth rates, already compounded by the fact that algae overgrowth can trap sediments and smother corals (references cited in Mumby et al., 2007). The higher biomass of algae on tiles outside the MPA is congruent with results from the benthic survey showing a significantly higher cover of algae on the reef outside of the MPA. Changes in algae type and quantity outside the MPA appear to be primarily due to reduced grazing pressure at the site, as other factors such as water quality and reef structure are similar between both areas. These results complement other studies, which have identified a link between herbivore abundance and algal community structure and biomass (e.g. McCook 1999; Smith et al. 2001; Mumby et al. 2006, 2007).

The fish surveys indicate that herbivorous parrotfish were more abundant and were also larger in size inside the MPA. Given that scraping parrotfishes may be more closely tied to their food source this may be an indication of suitable or preferable algal growth inside of the reserve, as indicated by the algal tile study. The justification for higher numbers of excavating scarids within the reserve is less conclusive given that larger parrotfishes have been known to travel longer distances (Hamilton 2004) and may not necessarily be ecologically tied to a specific region. However, during the monitoring survey apparent spawning aggregation of *C. microrhinos* localized within the reserve boundary were observed for a period of approximately 3 weeks, suggesting the maintenance of key ecosystem processes within the MPA. In sum, parrotfish are of crucial functional importance because they are grazers and bio-eroders in the reef ecosystem and their abundance can be used as an indication of coral reef health status (Hughes et al. 2006). As parrotfish abundance increases because of the MPA treatment, so does grazing activity, which reduces macroalgae and enhances coral reef growth (Mumby et al. 2006, 2007).

In spite of the encouraging results presented here, it is impossible to statistically conclude that the observed differences between the experiment and neighbouring control sites are due to the management treatment because of a lack of pre-MPA baseline data (Botsford et al. 2003; Edgar et al. 2004) and replication of this study in multiple-MPA and non-MPA sites across the lagoons. Neither can it be claimed that there is a spillover effect occurring, as the monitoring focused on measuring the effects of reserves on fish within MPA boundaries. Indeed, additional research of other species would have allowed for better scientifically informed decisions when designing the Nusa Hope MPA.

Chapter Seven

General Discussion

7.0 Introduction

This research exercise proposed to investigate parrotfish biology, ecology, and fishery parameters. The information contained within the last six chapters have been able to provide insight into an evolving artisanal fishery from an urbanized perspective, while the ecological and biological data contained herein have enabled me to begin addressing the four specific conservation and management issues highlighted from the outset.

Coral reef fish resources are exploited across thousands of square kilometres. As such, the establishment of central or community-based management are not only burdened by spatial scale, but also by various regionally specific obstacles. In this thesis I argue that targeted management and conservation efforts are urgently required where exploitation is most concentrated. This plea is based on a number of factors. Firstly, the growth of urban populations, particularly in the Western Pacific, is rapidly leading to large concentrations of inhabitants within small regions. In fact, by the end of 2008, the United Nations predicted that for the first time in human history more than half of the world's population will be living in urban areas. The UN estimates that population shift from rural to urban areas will be faster in developing countries where looming economic and social problems will become increasingly difficult to deal with. As highlighted in chapters one and two, the availability of infrastructure in only a handful number of urban locations naturally reduces the option for new urban growth in countries like Papua New Guinea and Solomon Islands. As a result, existing urban areas are continuing to expand. This provides a unique opportunity where the implementation of localized management and conservation measures, and subsequent compliance monitoring, should be less spatially problematic than effort spent across numerous urban or semi-urban areas.

Secondly, urban centres such as Gizo are built on government owned land, and as such are void of customary tenure restrictions which may hamper centralized management initiatives. It appears that the lack of custodial or customary ownership is not only part of the problem, but can also be part of the solution. For example, Gillett and Moy (2006) highlight the fact that centralised management measures in some Caribbean countries may have been more successful given the lack of indigenous ownership and customary marine tenure. In these areas, government backed management and conservation initiatives have been more successful as the majority of inhabitants do not have customary claims to adjacent marine resources and must abide by national laws. Therefore, the justification for fishery management measures in government owned areas such as Gizo should potentially be less problematic. For example, Rhodes et al., (2008), point out that the enforcement of a commercial spearfishing ban in Pohnpei has been successful since “(1) markets are centralized, (2) marine resource enforcement agencies are proximate to markets, and (3) speared fish are readily identified”.

Finally, given the fact that expansion of fishing effort beyond the immediate depleted urban parameters are increasingly being documented in the Pacific, concerted management and conservation efforts are now a priority. Although in certain areas, such as Gizo and Alotau (discussed in chapter 2), expansion of fishing effort is relatively easier given the vast areas of outer coastline available to artisanal fishers, in smaller countries the limited natural resources will inevitably be exhausted faster, to the point where urban human survival may become exceedingly difficult. However, even in resource rich countries such as Solomon Islands and Papua New Guinea artisanal fishing activities are restricted by cost-effectiveness, as pointed out by Liese et al., (2003) in the open access fisheries of Indonesia (Discussed in chapter 2). Furthermore, the increasing cost of fuel will inevitably impact on overall costs which fishers will have to take into account when considering overall cost-effectiveness. In fact, the higher cost of fuel has already influenced fisher behaviour around Gizo, with spear fishers camping overnight on nearby islands in order to fish longer (Pers Com., Alec Hughes, 05/04/10).

In recent macro-analysis of global coral reef fisheries, Newton et al., (2007) and Mora (2008) concluded that the majority of coral reefs are overexploited. Using island-specific human population projections, Newton et al (2007) estimate that “an extra 196,000 km^2 of coral reef may be required by 2050 to support the anticipated growth in human populations”. Mora (2008) has confirmed that population density and the associated activities of concentrated urban populations, such as coastal development, pollution, and changes in land use, are profoundly exacerbating impacts on coral reef already under severe stress from overfishing and climate change. In fact, Mora (2008) advocates the use of human population density as the best surrogate of anthropogenic impacts related to the loss of biodiversity in coral reef ecosystems.

As highlighted in this thesis, the importance of coral reef fisheries are not due to the quantity of landings, which amount to between 2%-5% of global fisheries catches (Pauly et al. 2003), but to their contribution towards food security and supplemental employment for millions of coastal inhabitants across the planet.

There is an increasing realization that human populations living on coastal coral reef dominated regions cannot solely rely on the sea for either their nutritional needs, or income generation (Mora 2008; Newton et al. 2007). Not only diversification in resource use is necessary to shift dependency away from reef fisheries, but there is an urgent need to identify and support alternative income-generation initiatives. However, this is easier said than done given the lack of key economic or logistical infrastructure in many tropical nations. In relatively more affluent American Samoa, Craig et al., (2007) showed that the subsistence fishery of outer islands have been sustainable over the millennia, partly due to depopulation, and partly due to increasing socio-economic opportunities “which translates into less reliance on fishing”. Ironically, the depopulation of remote outer islands in the Pacific is a direct consequence of the urban drift by people looking to improve their socio-economic status in burgeoning cities.

Following I have partitioned the summation of my findings to initially conclude on the major ecological, biological and fishery-related information, after which I proceed to use these findings to address the four cornerstone coral reef fisheries management and conservation issues highlighted at the beginning of the thesis. The three specific aims drafted to answer the above four issues were; 1) To evaluate the impacts of local small-scale fishing on parrotfish assemblages in Gizo, Solomon Islands, 2) To compare the demographic and life-history characteristics of parrotfish species from Solomon Islands with other locations, and 3) To investigate the effectiveness of Marine Protected Areas (MPA) as a parrotfish conservation tool.

7.1 The Gizo fishery

The first long-term study of an urbanised artisanal fishery in Western Solomon Islands has been presented in this thesis. CPUE data from the Gizo artisanal fishery indicated that fishing pressure was high and probably unsustainable. Herbivorous fishes were shown to be important to the local artisanal fishery where scarids and acanthurids made up a large proportion of night-time spearfishing landings. In particular, these two groups proved especially vulnerable to spearfishing, making up the largest percentage of total CPUE catch (kg) in 2004 and 2005. Size distribution analysis showed a sharp decline in larger scarids (maximum TL>50cm) from 2004 to 2005, and the eventual prominence of smaller scraping scarids (maximum TL<50cm) in 2005. The notion that fishers target larger individuals was validated from this analysis.

Creel data analysis confirmed the general trend exhibited by CPUE data, where smaller scarids appeared to fill in the gap left by the decline of larger species. Size distribution analysis for *Scarus ghobban* and *Hipposcarus longiceps* depicted declines in larger size-classes from 2004 to 2005, which are supported by evidence of their decline in total catch figures (kg). Gizo spearfishing CPUE was calculated at approximately 5.8 kg hr^{-1} in 2004 and 4.5 kg hr^{-1} in 2005, which are slightly larger than spearfishing CPUE from Pohnpei (Rohdes et al., 2008), estimated to be at 3.6 kg hr^{-1}, and also in Papua New Guinea (see Chapter two). Scarids contributed on average 1.67 kg hr^{-1} or 29% of the hourly catch in 2004, and 1.596 kg hr^{-1} or 36% of the hourly catch in 2005. Parrotfish creel data showed that the Gizo reefs yielded approximately 4.44 kg hr^{-1} in 2004 and 4.34

kg hr^{-1} in 2005. This confirms the notion that in the face of declining size-distribution, catch figures remain relatively similar as effort shifts on to smaller individuals.

Overall, the rapid transition with which catch and size-distribution of parrotfish species changed is an indication of their vulnerability to even low levels of artisanal fishing pressure.

7.1.1 Fisher survey

An interesting insight was gained into fisher behaviour and perception from the interviews. Detailed ecological knowledge pertaining to parrotfish harvesting methods were demonstrated. There was unanimous agreement that spearfishing at night was the only productive method to exploit parrotfish schooling and sleeping behaviour. Although fishers were aware that general abundance had declined around Gizo, they did not perceive it to be a problem as expansion of fishing effort was still cost-effective. In fact, interviewed fishers communicated a strong desire in acquiring independent logistical capabilities, such as outboards, nets, fibre canoes and SCUBA, in order to allow for larger expansion capacity in fishing effort.

CPUE, Creel, and fisher data collected as part of this research suggest that the Gizo reefs are suffering from localised fish declines, most noticeably in important functional and predatory species such as parrotfishes and groupers, respectively. Given the large size of Western Solomon Islands widespread fisheries crashes are unlikely. However, ramifications from localised removal of trophically important fishes in conjunction with associated changes in land use and pollution form Gizo's increasing population is of concern and will inevitably lead to overall ecosystem changes.

7.1.2 Abundance and Distribution

Abundance and distribution data collected as part of this research was consistent with the general stock status of scarids portrayed by the CPUE data. Furthermore, UVCs were able to confirm that the decline of parrotfish numbers are localised to the open access Gizo fishery, as contrasted by the relatively healthy Vavanga subsistence fishery.

Parrotfish CPUE (including creel) and UVC estimates appear to confirm that (1) density had declined, (2) large bodied fish numbers had declined, and (3) mean body size had declined. Although density estimates significantly decreased across most size classes for all species, larger bodied *Bolbometopon muricatum* and *Scarus ghobban* (>40cm TL) were clearly the most affected. Total parrotfish numbers sighted declined across most size classes from 2004 to 2005. Gizo's overall density and size-class distribution portrayed a parrotfish assemblage that was in decline under quantified fishing pressure. Overall, analysis of specific abundance indicators, in combination with fishery related data, have enabled me to assess the relative impacts of fishing pressure on Gizo's parrotfish assemblages. Based on this investigation I can confirm that fishing pressure from the Gizo fishery appears to have induced; (a) declining population densities, (b) shifts in size-structure, and (c) reduced average body size. It is therefore reasonable to assume that such transitions will have ecological implications given the functional roles of herbivores within coral reef ecosystems.

Overall, the findings of studies cited in chapter 3, such as Ruttenberg (2001) from the Galapagos Islands, Bruckner and Bruckner (2003) from Netherlands Antilles, Kramer et al., (2003) from Bahamas, Hawkins and Roberts (2004) from the Caribbean region, and Tuya et al., (2006) from the Canarian Archipelago, are consistent with the above findings.

7.2 Population dynamics

7.2.1 Age and Growth

This study identified plasticity in parrotfish demographic trends between different locations, including within the same ocean system (Australia vs Solomon Islands). Growth patterns, asymptotic size and longevity were vastly different, even within a single region (e.g. Seychelles) or the same ocean system (e.g. The Pacific). The considerable variations in demographic parameters cannot be attributed to the effects of latitude or resource availability alone. Furthermore, the effects of historical fishing pressure on longevity are at best speculative without detailed comparative analysis. In addition, comparative phylogenetic analysis has revealed that some species exhibit distinctly unique evolutionary histories within different spatial scales (Bernardi et al. 2001; Bay et al. 2004), which further complicates the generalisation of demographic trends. However, such regionally specific traits reinforce the need for the understanding of localised population dynamic patterns.

Comparison of VBGF curves revealed that no two population samples enjoyed similar growth trajectories or mean asymptotic length. The population samples of each examined species exhibited different maximum sizes and ages. However, there are some general trends. Parrotfishes from Solomon Islands appear to be the shortest lived and among the faster growing in comparison to other locations. For example, in chapter four I draw attention to a two and four year old *Scarus rubroviolaceus* from Solomon Islands which have the same size of a seven and eight year old from Seychelles, respectively. This disparity in growth rate is even more evident in *Hipposcarus longiceps* where the size reached by the oldest individuals from the Solomons (5yrs, 410mm SL) has not been surpassed by any other individual from the GBR population, including the two oldest at 11 yrs (405mm SL) and 14 years (375mm SL). A trend which is apparent in all population samples is that most growth occurs quite quickly and certainly within the first half of their respective life-span. Interestingly, the presence of old females was evident in populations where fishing pressure was low or negligible (e.g GBR). In this case, it appears that essentially there is no gender imbalance which would force females to change sex to males. A trend that was apparent across all species and population samples was in growth trajectories decoupled by sex, which showed faster male growth in all datasets (discussed in chapter 5). Assessment of mortality estimates were generally speculative given the size and age-biased nature of some datasets, not to mention other physical and environmental factors such as predation or refuge availability.

Iconic coral reef fish species such as *B.muricatum* and *C.undulatus,* which are known to be large and relatively long-living, are susceptible to modest fishing pressure (Hamilton 2004, Choat and Davies 2006). However, this thesis has demonstrated that smaller fast growing parrotfish species, which have shorter life span and high turnover rates, are also not immune to modest fishing pressure.

7.2.2 Life history

While age and size at 100% female maturity was estimated for *Chlorurus microrhinos* from the GBR, it was impossible to calculate the same parameters for *Scarus ghobban* from the Solomons, as only a single immature female was histologically confirmed. Based on the available data, we must assume that *Scarus ghobban* are 100% mature at 240mm(FL) or 1 year of age. Given the lack of immature females, and the presence of very young and small mature females, the possibility that such traits are induced from heavy fishing pressure cannot be discounted. However, without comparative data it is impossible to make any solid assertions, and having already established the short longevity of scrarids from Solomon Islands, it is entirely possible that such compact life-history traits are entirely normal.

Histological evidence of transitional individuals, in addition to primary males, indicates that *S.ghobban* and *C.microrhinos* are diandric protogynous hermaphrodites. As such, both species possess dual male development pathways. However, there are indications in the *S.ghobban* dataset that some males may have changed sex from immature females, given the lack of gonadal evidence pointing to previous spawning. Histological analysis also point to the fact that scarids in Solomon Islands grow fast, have shorter life span, are very young at first female maturity, and thus enjoy high population turnover rates. Such characteristics may be related to a short life cycle, which means that reproductive activity must be regular in order to maximize regenerative contribution back into the population. While that may be true, it appears that the apparent high turnover capacity of *S.ghobban* from Solomon Islands is not providing sufficient protection from quantified fishing efforts which have caused density declines in parrotfish stock numbers.

Overall, three specific traits have been identified with regards to *S.ghobban*. Firstly, females mature very early and at small sizes. Secondly, large body length is reached relatively quickly. Thirdly, it's a high fecund species. As such, *S.ghobban* should theoretically show resilience to fishing pressure around Gizo. However, established catch rates and density estimates indicate otherwise.

7.3 Marine Protected Areas

The role of parrotfishes in maintaining or rehabilitating coral reef health within marine reserves and their capacity to respond to protection are increasingly being investigated (Mumby et al. 2006, 2007). As the link between herbivory and the health of shallow water coral reef ecosystems is afforded more attention (Hughes 1994; Hawkins and Roberts 2004) the use of marine reserves as a conservation and rehabilitation tool is also being advocated (TNC 2002; Hawkins and Roberts 2003; DPNR 2005; Hawkins et al. 2006). Hawkings and Robert (2003, 2004) found that not only after 5 years of protection the average size of parrotfish species increased but biomass also increased four-fold within reserves and doubled within nearby fishing areas.

Similarly, Mumby et al., (2006) state that protection from fishing pressure significantly increases parrotfish size and density, which in turn increases grazing capacity, resulting in a four-fold reduction in the cover of macroalgae. Interestingly, Mumby et al., (2006) point to the fact that because parrotfishes are often overexploited, especially through spearfishing, "reserves will almost always increase the level of fish grazing within their boundaries", "a process that is key to the ecosystem functioning of coral reefs". Mumby et al., (2007) draw on their research in the Caribbean to point out that protection from fishing induces trophic cascade on targeted coral reefs, where the number of grazing herbivorous fish increase, leading to greater grazing rates, which subsequently leads to reduced macroalgal cover, which ultimately enhances the process of coral recruitment.

Ecosystem based approaches to coral reef fisheries management are increasingly being advocated, especially through precautionary management tools such as marine protected areas (Aswani et al. 2007; Mumby et al. 2006/2007). However, marine reserves are not a panacea, given that reserves effects are especially complex to predict, depending on a range of complex interactions. Russ *et al.*, (2003) point to the fact that the rate of recovery periods for exploited species depends on a number of factors, including initial population size, intrinsic rate of population increase, life-history characteristics, the success of individuals recruitment events, rate of immigration into reserves, and the extent of reduction of fishing mortality in a reserve (Quoted from Jennings 2001). Simply put, rate of recovery is taxon and demography specific.

For example, the investigation by Nardi *et al.*, (2004) in to the effects of protection on the abundance of two exploited reef fish; the subtropical wrasse, *Choerodon rubescens*, and the coral trout, *Plectropomus leopardus*, discovered that after eight years of protection abundance of the coral trout had significantly increased within two protected areas while the *C. rubescens* did not appear to respond to protection. However, they also showed that the abundance of *P. leopardus* did not respond to protection for three years. This further re-enforces the “long-term” approach to protection advocated by Russ & Alcala over the past decade (see Russ and Alcala 2004 or Russ 2002).

What these cited studies demonstrate is the complexities involved in protecting a multi-species ecosystem where functional species coexist alongside apex predators. They also give possible reasons for these responses ranging from prey-predator relationships to natural fluctuations, demographic and life-history traits, spatial ecological distribution, and unregulated heavy fishing pressure close to MPA boundaries. Marine protected areas are “built upon opportunity or necessity” (Roberts et al. 2001). It is not often possible to create reserves or networks of reserves driven by a primary focus on covering all critical biogeographic regions. In reality however, a balancing act must be played in order to reach a consensus between the social, biological and economic factors that drive conservation. In practical terms, the expectations that MPAs can manage or rehabilitate adjacent fisheries through the ‘spillover effect’ may not be realistic in every situation. Thus the “Spillover effect” should not be reason alone to protect coral reefs.

Roviana and Vonavona lagoons are dominated with small patchy areas of coral growth often encrusted with algae or sediments. Therefore, the opportunity to protect and conserve rare areas of high collective coral growth and biodiversity, such as Nusa Hope, should be seized upon and encouraged. Our algal and fish surveys in Nusa Hope demonstrated higher grazing activity and parrotfish abundance within the MPA, respectively, which appear to suggest that parrotfish do benefit from protection. Although the lack of replication across multiple MPAs within the two lagoon systems prohibits me from making definitive assumptions as to the benefits of protection from fishing pressure, the findings of a parrotfish survey (Aswani and Sabetian 2010) conducted in the Kida MPA (Vonavona Lagoon) also portray similar higher parrotfish abundance within reserve boundaries.

7.4 Conclusions

In chapter one I proposed that information gathered as part of this research program will be used to address four specific conservation and management issues. The following tentative conclusions have been arrived at with respect to each of these issues. However, it must be pointed out that given the baseline nature of this study, the working hypotheses arrived at here need to be replicated and re-evaluated.

1) *The capacity of exploited species to respond to non-subsistence levels of fishing pressure*

Based on the findings of fishery and abundance surveys, it appears that the shallow water behaviour of parrotfishes make them more susceptible to fishing pressure. The results appear to suggest that artisanal and small-scale commercial fishing in Gizo have had a negative effect on density and size-structure of scarids. We can therefore tentatively conclude that the capacity of parrotfishes to respond to modest levels of fishing pressure is weak. This conclusion is consistent with the findings of Ruttenberg (2001), Bruckner and Bruckner (2003), Kramer et al., (2003), Hawkins and Roberts (2004) and Tuya et al., (2006), which concluded that coral reef fish assemblages, including herbivorous fishes, are susceptible to low fishing pressure.

2) *Lack of detailed comparative biological and ecological information*

The wide range of comparative demographic and life-history data collected as part of this research program have provided important insight into age, growth, and reproductive characteristics of scarids. Comparative analysis has clearly shown that the population dynamics of parrotfishes are spatially-specific. Given this fact, regionally specific management and conservation efforts will have to incorporate detailed biological and ecological information.

3) Over-exploitation and possible links with life history traits.

Demographic analysis of scarids subjected to little or no fishing pressure (e.g. GBR, Cocos Kelling) show the persistence of females through older age classes. Reproductive analysis of scarids subjected to heavy fishing pressure (e.g. Solomon Islands) show sexual maturity at an extremely early age, and no older females. However, no comparative data is available to link fishing pressure as the causative factor. The early maturation of scarids in the solomons, combined with their relatively faster growth and short longevity appear not to provide any protection from fishing pressure. As such, their regenerative capacity from fast growth and high fecundity appears to be insufficient in the face of over-exploitation. Finally, it is not clear that all scarine labrids are protogynous hermaphrodites, as shown by Hamilton (2004, 2008). As such, further research is required to address the paucity of data in this field.

4) The capacity of coral reef ecosystems to respond to protection.

Our MPA surveys appear to suggest that parrotfishes do benefit from the removal of fishing pressure. Given that scarids are ecologically tied to coral reefs, the absence of human activity allows for faster coral regeneration and consequent healthier coral reef ecosystems through increased herbivory. This is an added advantage of protection, especially given the amount of coral destruction which occurs either by trampling or other human activities.

7.4.1 Conservation and management implications

The findings of this study have significantly contributed to the knowledge-base of parrotfish fisheries conservation. By triangulating fishery, ecological, and biological data, I have been able to help address baseline knowledge deficiencies in all three fields, validate important facts, and also identify areas of future research priority. It is clear that coral reef fisheries are most in need of protection around high concentrations of human populations, thus a shift of research focus from village to urban fisheries is urgently required. Although relatively little attention has been given to the effects of urbanization on coral reefs in the Pacific, there are examples that can be cited. Zann (1994) summarized the status of coral reefs adjacent to urban regions in Fiji, Tonga and Western Samoa. His conclusions were that human impacts on coral reefs adjacent to urban areas were much higher than outer remote villages. He warned that rapid population growth and unplanned development in urban areas result in significant loss of coastal habitats, overfishing, pollution and eutrophication of adjacent reefs. Some of the trends observed were growth and stock overfishing of several fish species, which are assumed to have been responsible for observed outbreaks of crown-of-thorn starfish. Zann's conclusions were based on previously unpublished results, but much more empirical research is needed to verify, validate and better understand data associated with urban fisheries parameters.

Although precautionary ecosystem management is being touted as the future of coral reef conservation, they are still dependent on relevant scientific knowledge. Simply forbidding access to an area through establishing marine reserves, or banning certain fishing methods (spearfishing in this case) will not be effective if we don't know the ecology and biology of the species being protected, do not fully understand the ecological parameters within which they exist, or manage activities that can affect coral reefs outside of reserve boundaries. For example, WWF Solomon Islands in conjunction with the western provincial government has embarked on the development of marine reserves around Ghizo island. I believe this initiative to be an important steppingstone towards the conservation of Ghizo's marine resources, which will have to be complimented by management measures such as a ban on spearfishing. However, these measures will not be effective if other issues such as monitoring and enforcement of rules, adjacent land zoning and land-based sources of pollution, artisanal fishing patterns, and socio-economic reliance on coral reef fisheries are not addressed at a governance level.

There is enough scientific evidence out there which suggests that coral reefs, and associated species, do suffer from human activities. The challenge is to identify species that suffer disproportionately due to specific anthropogenic (e.g. Pollution), biological (e.g. Grouper spawning aggregation) or ecological (e.g. Parrotfish shallow water sleeping) characteristics. It is only then we can begin to study the relative impact of their loss within the reef ecosystem.

The information presented in this thesis provides a platform for further discussion on the significance of parrotfish fishery conservation as advocated by the 2008 International Coral Reef Symposium in ensuring the long-term survival of coral reefs around the world.

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Appendix 1. Summary of responses to fisher survey interview questions

For the fisher survey a semi-informal interview format was adopted. Between 2004 and 2006, fifteen “expert” artisanal fishers from the collaborating crew were asked a range of questions.

In total, seven questions were asked of each informant, in a semi-informal manner, where the direct responses to the questions were used to initiate further conversation.
The main reason for choosing this format was to conduct questioning in a more relaxed atmosphere where the respondent felt comfortable in sharing their knowledge and experiences. The interviews were conducted in Solomon Island Pijin and began by each member stating their name and location of residence. Other details such as the date, and duration of the interview were also recorded.

The responses of 15 interviews are summarised below.

Question One) How would you rate your knowledge of Parrotfish behaviour, and ecology? Give examples.

Excellent = 9; Good = 6;

Feeding

All respondents (n=15) mentioned the fact that Parrotfish are herbivors. All respondents made reference to the fact that *B.muricatum* breaks off chunks of live and dead coral with its head and consumes whole. Twelve fishers made specific reference to the fact that some parrotfish species do not eat coral and appear to scrap or pick at algae or detritus. Five fisher had personally observed parrotfish feeding on coconut flesh disposed off after or during Copra operations, potato or cassava pudding, taro, and other cooked plant or vegetable materials. The other fishers had heard of this behaviour. Nine fishers (n=11) claimed that parrotfish actively feed just after dawn and just before dusk, while the remaining respondents said that parrotfish feed at all times of the day. Five fishers (n=5) claimed that rising tides appears to be the time when parrotfish are more actively feeding, the remaining where not sure (n=5 could not answer that question).

Ecology (Swimming, Sleeping, Spawning)

Nine fishers (n=9) made specific reference to the claim that not all parrotfish display schooling behaviours. The species *H.longiceps, B.muricatum, C.microrhinos,* and *S.oviceps* (in no particular order) were the top four nominated schooling species. Interestingly, four fishers from the nine respondents distinguished the fact that *B.muricatum* and *C.microrhinos* did not appear to swim in large schools (>30). The remaining six respondents gave various answers, an interesting one of which was the statement that some species of parrotfish (e.g. *C.Bicolor*) swam in pairs, and some only start gathering in large numbers prior to spawning. Two fishers referred to having personally observed sleeping parrotfishes encased in a mucus bubble. All respondents (n=15) stated that parrotfish sleep in coral trees, holes and crevices. Nine fishes claimed to know where exactly to look for sleeping parrotfish. They describe refuges where protection from current movement is provided. They made specific reference to coral trees for smaller scarids, where they wedge themselves between branches. Larger scarids (e.g. *C.bicolor*) were mentioned as solitary sleeping parrotfishes, while most other species where referred to be group sleepers. Four fishers (n=4) described parrotfishes as the deepest sleepers of all fish.

When asked about spawning periodicity, there was no clear consensus. Three fishers (n=3) claimed that numerous species of scarids mass spawn together during the new moon period. Four fisher (n=4) stated that parrotfish appear not to have specific lunar or diel spawning periodicity. This may be an important response, even though consensus has not been reached. This response appears to concur with other informal conversations that I have had with fishers over the past 5 years. Two fishers claimed to know of specific spawning locations for *C.microrhinos* near neighbouring Rarumana Isalnd where aggregations last for several days. Both these fishers stated that spawning aggregations seems to happen in sandy shallow areas. Again, no consensus was reached on spawning locations, as 13 fishers (n=13) could not associate specific habitat type with spawning. All of the following were mentioned; passages, shallow reefs, reef drop-offs, sandy beds adjacent to mangroves, sea-grass beds, inner lagoon areas, and exposed outer lagoon areas.

Question Two) What is the best method with which to target Parrotfish? Why?

Spearfishing is best = 9
Netting (including drive netting) is best = 1
Spearfishing and netting = 5

The answers to this question turned out to be very interesting. The first part of the question was answered very directly by the respondents. The fisher response was directly related to which method they thought best represented the easiest technique to catch parrotfish. However, when asked "why"? the responses became more interesting. It appeared to me that the fishers response was directly related to their proficiency as free divers, with those who were not experts nominating netting as the best method.

Furthermore, there was a distinction made when fishers elaborated on their answers. From a commercial perspective, a number of factors, such as fuel, manpower, carrying capacity, and location were considered before the fishing method (spear vs netting) was decided upon. There was unanimous agreement that parrotfishes are most easily targeted with night-time spearfishing. This method is the cheapest. However, "if" conditions are right, netting can be more productive and less laborious. But this method is more costly (i.e. fuel) and can some time be unfruitful. Six fishers for example, claimed that rising tides just before sunset or sunrise provided the best opportunity to place nets for schooling parrotfish.

Poisoning and line-fishing baited with potatoes were also mentioned as techniques with which parrotfish can be caught. These methods are not used artisanaly.

Question Three) Are parrotfish popular with customers? Which species?

Most popular = 6; Very popular = 10; Moderately popular = 6; Not very popular = 0

The bumphead parrotfish *B.muricatum* was voted unanimously as the post popular species with customers, regardless of its high fetching price. This was followed by the other big parrotfish, *C.microrhinos*. The $60 price tag on the *Epinephelus spp* Grouper (Figure 1 C) puts in to perspective the sort of prices for similar sized scarids (>50cm FL). One fisher claimed to have the Gizo Hotel (the biggest in Gizo) as a regular customer for *B.muricatum*, willing to pay $10 a kilogram. The prices of other scarids were all size based with individuals greater that 350mm FL fetching prices of >$7 each. The pictures below are a range of scarids and prices at the Gizo fish market.

Figure 1. Gizo fish market, 2005.

Question Four) Which fishing grounds have you targeted over the past year?

Gizo reef, Hapu Hapu reefs; North New Georgia, n=15
Gizo reef, Hapu Hapu reefs, Rarumana Is., Vela Lavela Is.; n=8
Hapu Hapu reefs, North New Georgia Is., Rarumana; n=6
Gizo reefs, Njari reefs, Hapu Hapu reefs, Rarumana Is, Vela Lavela Is., Kolombangara Is.; n=4

Question Five) How would you rate the Parrotfish stocks around Gizo?

Declining = 15

The elaborated responses to this question was informative as all fishers blamed increasing fishing effort as responsible for declines across all reef fish families. Nine fishers (n=9) claimed that fisher pressure had significantly increased in the past few years. The ethnic Malaitan fishers attributed the increase in fishing pressure to the civil violence in the country since 1999. Widespread lawlessness and intimidation drove many Malaitans back to Honiara and other regions during that period. In their absence, the non-indigenous Gilbertese fishers took advantage of the employment gap by meeting the supply shortage in the fish market, thus taking up the fishing trade which the Malaitans have become known for in Gizo. Upon the establishment of a Multi-national peace force in 2003 (RAMSI) the Malaitans returned to Gizo, only to find that the artisanal fish market had become more competitive. The fishers tell me that establishing large crews in the only way to cut costs and improve profit margins.

Question Six) What do you think of past, current, and future fishing trends around Gizo?

There was unanimous agreement that fishing productivity had declined historically. There was also unanimous agreement at parrotfish species had become more prominent in the market. Spearfishing and the close proximity of the Gizo reefs (within paddling distance) were nominated by nine fishers as contributing factors. Eight fishers had claimed that large-sized parrotfish numbers had declined significantly in 2005, with five fishers claiming to have noticed absence of specific species such as *S.ghobban* (TP) and *C.microrhinos.*

Although fishers agreed that parrotfish stocks were declining, only three fishers (n=3) perceived this to be a problem as the others pointed to the fact that demand was being met by an expansion of fishing area on to more productive reefs. Four fishers claimed personal knowledge that fish stocks, parrotfish in particular, are still very healthy around the uninhabited areas of neighbouring Rarumana Island, but did concede that the most productive and cost-effective way to target the area was through mobilizing large crews and utilizing very large nets.

Question Seven) Do you intend to continue fishing as a main source of income? What would you like to do or have to increase your fishing capacity?

Twelve respondents (n=12) indicated that they would continue with fishing as a main source of income, while the remaining three fishers held some reservations and would not commit long-term. Interestingly, these three fishers cited their lack of transport and logistical ownership as the reason for wanting to find other employment alternatives. This maybe an indication that artisanal fishing is increasingly difficult to conduct as a sole trader, thus crews are required to share costs and increase effort. Given that fact that petrol prices were at the time nearly $9 a litter, a round trip to Rarumana would cost a fishing crew at least $400 dollars ($80 AUD). Thus operating costs are increasingly being considered in expeditions. Furthermore, all fishers pointed to the close proximity of the Gizo reefs as a reason why they are disproportionately targeted. In particular because fishing expeditions are significantly less expensive. Nine fishers indicated their desire for independence conceding that their inability to acquire fishing gears (e.g. nets), boats and outboard motors on their own was the major reason why they still persisted with working in crews. One very interesting revelation from six respondents was their desire to learn how to SCUBA dive and use air compressors.

Appendix 2. Underwater size-estimate exercise

The aim of this exercise was to gaining sufficient real-time proficiency in the estimation of fish size. Instead of estimating and then measuring coral heads (as used in Sabetian 2003) to achieve an acceptable error in estimated and actual size, the use of live fish were employed for this exercise. Slow moving, feeding, or stationary parrotfish were observed underwater and their TL estimated. After this, I approached the location and measured the length of the identified background substrate.

Between 14/06/04 and 16/06/04, three separate dives were conducted at Vavanga, at approximately 10 meters depth where I randomly identified parrotfish and proceeded to estimate their size. The distance from which observations were made ranged from five to ten meters, after which the actual length of the background substrate was measured to the nearest centimetre. Twenty estimates were recorded during each exercise.

After conducting this exercise, I was sufficiently confident about my ability to accurately estimate fish size within a certain category. Although, the data shows that I could not accurately estimate the actual length of all fish, the error was consistent to within 5cm of the actual length. The size of this error was deemed to be acceptable and was consequently used to set the 5cm size range in the UVC. Finally, because the majority of scarids encountered were below 50cm TL, it was decided to award a single size category for fish above this size, '50+ cm', which represents all scarids equal to or above this size. Although large Bumphead parrotfish *B.muricatum* were regularly sighted during this exercise, it was impossible to verify their size as they always kept their distance to beyond 20 meters. On the opposite side of the size categories, it was deemed appropriate to ignore fish below 15cm TL as their size estimation would not only be somewhat difficult, but also problematic given the schooling nature of some scarids. Furthermore, it would have been too time consuming to pursue very small individuals. Visibility was consistently at 20 meters or better during the dives.

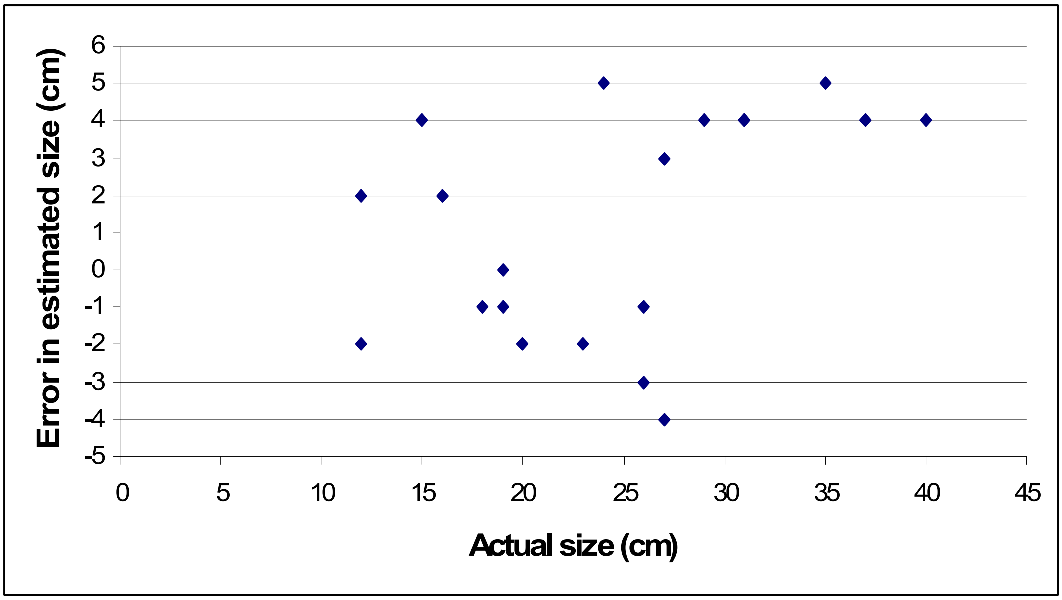

Figure 1. Error in the estimated size (cm) of fish, carried out on the 14/06/04 at Vavanga (n=20).

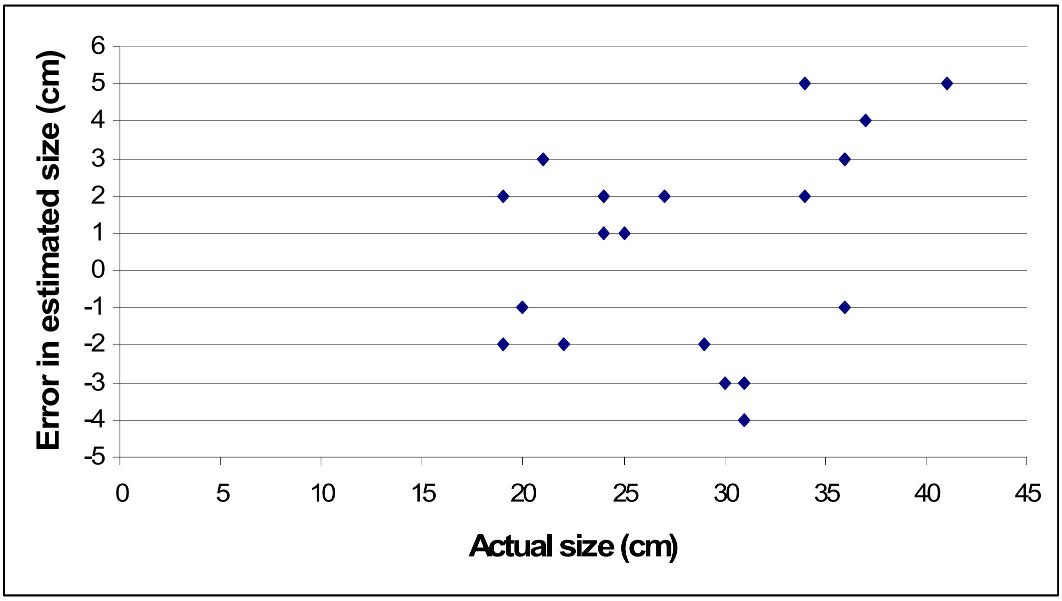

Figure 2. Error in the estimated size (cm) of fish, carried out on the 15/06/04 at Vavanga (n=20).

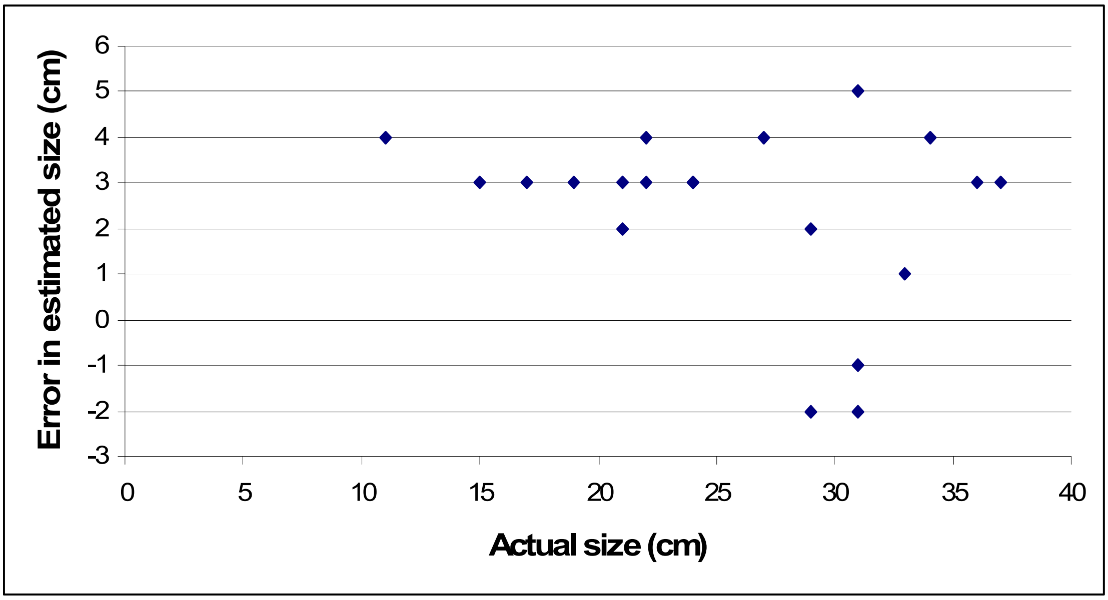

Figure 3. Error in the estimated size (cm) of fish, carried out on the 16/06/04 at Vavanga (n=20).

Figure 1-3 show that error was estimated to within 5 cm of actual size in most occasions. As a general trend, I underestimated the size of smaller scarids while overestimate the size of larger one. In order to apply size estimation training to real fish, calibrations were also carried out *in situ*. This process involves estimating the total length of a fish, spearing it, and then recording its actual length (Bellwood and Alcala, 1988; Kingsford, 1992).

Date	Species	Estimate size	Actual size
15/06/04	*Scarus Ovicpes*	20 cm	23 cm
15/06/04	*Scarus Ovicpes*	21 cm	20 cm
15/06/04	*Scarus Ovicpes*	24 cm	26 cm
15/06/04	*Scarus ghobban*	27 cm	29 cm
16/06/04	*Scarus schlegeli*	30 cm	33 cm
16/06/04	*Hipposcarus longiceps*	21 cm	19 cm
16/06/04	*Hipposcarus longiceps*	29 cm	25 cm

Table 1. Estimated and actual total length of scarids speared in Vavanga. Average observer error was 2.4 cm.

Appendix 3. Transect swim-time exercise

I employed the timed swim technique for two major reasons, firstly it is extremely laborious and time consuming to lay transect lines. Secondly, I wanted to be able to descend to the marked depth and swim a transect without having previously disturbed the fish community through laying measurement lines. Using GPS coordinates 21 pre-measured 100m transects were replicated at 15 meters of depth during which the size and frequency of the six targeted scarid species were observed. At the completion of this exercise I was confident that a 5-minute swim accurately represented a 100 meter transect and also give enough time for observation.

Transect	Date	Time	GPS coordinates
1	17/06/04	3 min 54 sec	X
2	17/06/04	4 min 45 sec	X
3	17/06/04	5 min 25 sec	X
4	17/06/04	5 min 04 sec	X
5	17/06/04	4 min 59 sec	X
6	18/06/04	5 min 19 sec	X
7	18/06/04	4 min 02 sec	X
8	18/06/04	5 min 39 sec	X
9	18/06/04	5 min 51 sec	X
10	18/06/04	5 min 12sec	X
11	21/06/04	5 min 19 sec	X
12	21/06/04	5 min 28 sec	X
13	21/06/04	5 min 34 sec	X
14	21/06/04	5 min 42 sec	X
15	21/06/04	5 min 16 sec	X
16	22/06/04	5 min 04 sec	X
17	22/06/04	4 min 53 sec	X
18	22/06/04	5 min 09 sec	X
19	22/06/04	5 min 01 sec	X
20	22/06/04	5 min 11 sec	X
21	22/06/04	5 min 05 sec	X

Table 1. Timed transect swims at Vavanga. (note: I was unable to extract the GPS coordinates from the handheld GPS recorder due to product failure caused by excessive humidity damage in the field.)

Appendix 4. Roviana and Vonavona Marine Protected Areas

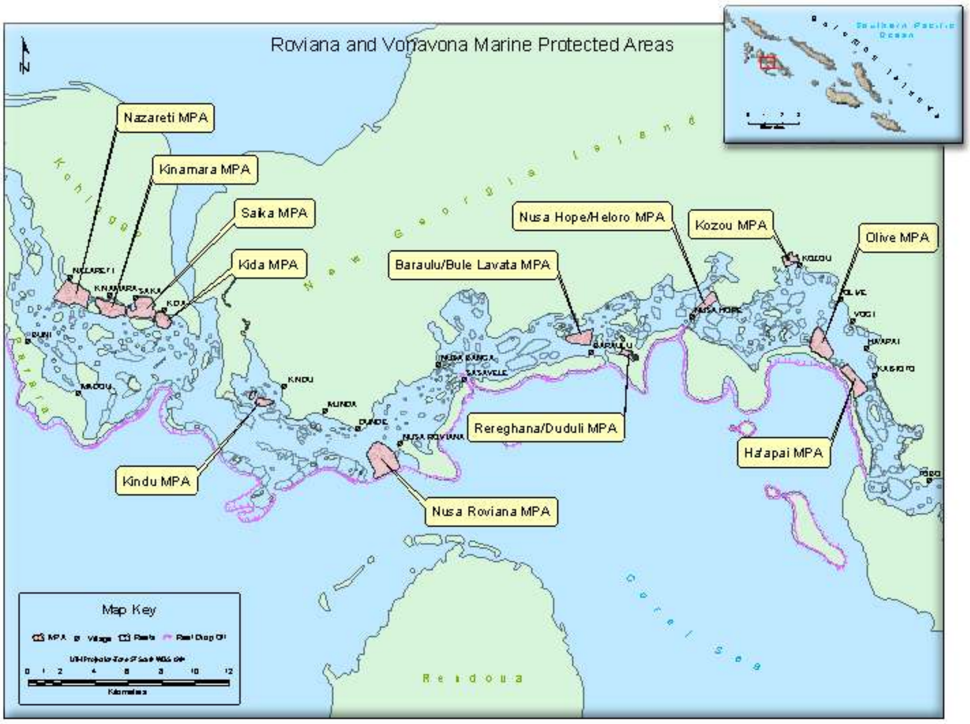

Figure 1. To date, 23 protected areas have been established. A further 11 sites have been added since this map was developed.