REEF FISH SURVEY METHODS AND APPLICATION TO POPULATION DYNAMICS OF PARROTFISH WITHIN THE KINGDOM TONGA

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DEDICATION

This thesis is dedicated to the memory of my Opa, Walter. A man whose love of science has passed through the generations, and who I hope to make proud.

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Just keep swimming….. – Dory, Finding Nemo

ABSTRACT

With environmental and anthropogenic impacts coral reef communities worldwide are predicted to decline. This is of particular concern for the Pacific Island nations' such as the Kingdom of Tonga. This thesis aimed to identify a method that could be applied to the collection of baseline data for shallow water coral reef fish communities, in particular to parrotfish. Parrotfish have been identified as a key ecosystem species on inshore coral reefs, and potentially indicators of reef health. A comparison of stationary and swim video Underwater Visual Census methods revealed similar results, but for both methods the stationary method overall had greater benefits. This method was then applied to the second aim of this thesis, investigating the population dynamics of parrotfish within the semi-enclosed lagoon of Tongatapu. All parrotfish were counted, and their behaviour and the size of the harem they were seen in were recorded. Terminal phase and, when possible, initial phase individuals were identified to species level for species richness. This identified 14 parrotfish species and one key species, *Chlorurus spilurus* (identified previously in the Pacific as *Chlorurus sordidus).* Very few harems contained a male, instead comprosing mainly of juveniles and initial phase individuals. Factors influencing these dynamics were identified, such as site factors; coral reef health, distance from mainland, anthropogenic and environmental influences. Through providing temporal baseline and understanding of population dynamics these results will assist in future management of inshore reef fisheries of parrotfish, with potential influence on the resilience of the coral reef health within the semi enclosed lagoon of Tongatapu.

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1 CHAPTER 1 - INTRODUCTION

The rapid growth of the human population has led to an increasing demand on the Earth's resources such as food, space and water, and the demands continue to increase (Ehrlich and Holdren 1971, Kronen *et al.* 2010, Limburg *et al.* 2011). Food resources and food security for primary production economies is of particular concern. For example, in the Pacific nations this demand falls primarily on their major source of protein, fish (Bell *et al*. 2009, Kronen *et al.* 2010, Cruz-Trinidad *et al*. 2014, Albert *et al*. 2015). These fish resources are used by multiple end-users with different levels of pressure; subsistence living (local), artisanal fisheries (regional), or industrial fisheries (global) (Kronen *et al*. 2010, Kittinger 2013, Albert *et al.* 2015). To supplement the demand of the local fisheries people are travelling further afield to collect fish, where one stock runs out they move to the next area (Albert *et al.* 2015). Forecasts predict that even the well managed fisheries of the Pacific Islands and territories will not meet the demand for fish in the future 15 years or even maintain the current consumption (Bell *et al.* 2009). Sustainability in the future is key, understanding fish populations and their dynamics through monitoring and research programmes is essential (Ludwig *et al.* 1993, Pauly *et al*. 2002, Kittinger 2013).

Coral reef and impacts

Coral reef ecosystems are renowned for their biodiversity, beauty and productivity (Hoegh-Guldberg 1999, Moberg and Folke 1999, Bellwood *et al.* 2005). They provide ecosystem services such as coastal protection, fisheries and eco-tourism (Moberg and Folke 1999, Hughes *et al.* 2003, Bellwood *et al.* 2004, Harley *et al.* 2006, Hoegh-Guldberg *et al*. 2007, Doney *et al*. 2012). They also play an important role culturally (Bellwood *et al*. 2004). The coral reefs and their inhabitants are in a vulnerable situation with the continual rise in anthropogenic effects, such as global climate change, ocean acidification, rising sea temperatures, over exploitation and pollution (Wilkinson 1999, Hodgson and Liebeler 2002, Bellwood *et al.* 2004, Harley *et al.* 2006, Hoegh-Guldberg *et al.* 2007, Winters *et al*. 2010, Doney *et al.* 2012, Wagner *et al*. 2015). Negative effects come from situations such as overfishing, siltation from land dredging, sewage pollution and toxic wastes (Done 1992, Hughes *et al*. 2003). These can also be placed into groups such as chronic disturbances, generally anthropogenic in nature, and short term, such as storms (Done 1992, Spalding and Brown 2015). Short term disturbances can involve bleaching events due to the change in sea level causing some corals to be exposed, from as small as 1 degree in temperature increase, with consecutive occurrences causing greater and more severe loss (Hoegh-Guldberg 1999, Harley *et al*. 2006, Hoegh-Guldberg *et al.* 2007, Muller *et al.* 2007, Doney *et al.* 2012, Spalding and Brown 2015). The rising of carbon dioxide with human fossil fuel combustion, reduces the oceans pH causing a change in the oceans chemistry of carbonate (Harley *et al.* 2006, Doney *et al*. 2009, Spalding and Brown 2015). This changes the calcification rates of the coral reefs, causing their degradation, smothering of the reefs by algae, and shifts in ecosystem dynamics (Doney *et al*. 2009). Direct effects of these changes may alter the physiological function of organisms, their behaviour and demographic traits. Conclusively leading to shifts in species interactions, trophic pathways and ecosystems functions and services (Doney *et al.* 2012).

The sensitivity of the coral-algal symbiosis to minor changes in temperature, for example, is what puts this as a more pressing issue (Doney *et al*. 2012). Bleaching can also create vulnerability to disease and delayed mortality can occur after a bleaching event (Doney *et al*. 2004, Spalding and Brown 2015). Coral reefs provide both a shelter from predators and a food source in terms of algae, in return the algae growth is managed by the herbivorous fish feeding beahaviour (Hixon 1997, Fulton and Bellwood 2002, Doney *et al.* 2012). Studies completed by Jones *et al.* (2004) and Idajadi and Edmunds (2006) identified that a loss in fish diversity and also invertebrate diversity may occur with coral habitat loss (Doney *et al*. 2012). The density of reef fish may decrease also through post settlement mortality, due to a reduction in hiding places, and lack of food for juveniles (Hoegh-Guldberg *et al*. 2007), which effects its services as a nursery. These effects on reef fish will have outreaching effects on the available fisheries resources (Hoegh-Guldberg *et al.* 2007).

Coral reefs worldwide yield at least 6 million metric tonnes of fish catches (Munro 1996, Hoegh-Guldberg 1999), which provides numerous jobs, and is a fundamental source of protein for developing countries (Hoegh-Guldberg 1999). Fishing yields, however, are likely to reduce with declining reef viability (Carte 1996, Munro 1996, Hoegh-Guldberg 1999). Coral reefs and their neighbouring ecosystems, such as mangrove and seagrass beds, together buffer the land from prolonged erosion and inundation, providing protective services (Hoegh-Guldberg 1999, Doney *et al*. 2012). This nature based protection can also act as a nursery for many commercial or recreationally important species, while also filtering and settling sediment and carbon storage, all working together to support the entire ecosystem (Doney *et al*. 2012). Without protection, and with increased sea level rise from melting and warming of the oceans, it is likely reef accretion rates will increase, intensifying the impact of storms (Hoegh-Guldberg *et al.* 2007). Coral reefs in the Pacific have already declined approximately 50% in the past hundred years, mainly due to local human impacts (Jackson 2010, Doney *et al.* 2012).

Anthropogenic effects are adding to the pressures of natural disturbances, many of which are chronic, and although they may be reversible they are not being reversed (Hughes *et al.* 2003, Spalding and Brown 2015). Overfishing has affected most of the world's coral reefs, leaving large areas of the reef without functional groups (Spalding and Brown 2015). The biomass of herbivores on reefs not accessible for fishing is on average double that compared to fished reefs; similar to the top predators such as sharks (Spalding and Brown 2015). With human population growth and alteration of coastal habitats there is a change in suspended sediment and nutrients with increased terrestrial runoff, this relates to direct and indirect effects on coral and algal growth, and the dynamics between them (Nyström *et al.* 2000). It has been noted that these more recent catastrophic effects have diverted attention from the fact that the reefs have been degrading since 1900 (Pandolfi *et al.* 2003), this earlier decline has been likely effected by the interaction of overfishing and land derived pollution (Pandolfi *et al.* 2003). Different species do have different tolerances to climate change and bleaching events, but regardless of tolerances, the services provided by the ecosystem will be altered (Hughes *et al*. 2003). Concerns for the reefs have continued to grow with further research; Marine Protected Areas (MPAs), no take zones and baseline research are being completed to provide an understanding of the current situation, and adaptive management and strategies for the future.

The impact of two stressors such as climate change, primarily loss of coral reef habitat, and also overfishing will have negative effects on fish diversity and leave reef fish vulnerable (Graham *et al.* 2011, Doney *et al*. 2012). Complex feedbacks of fishing effort, stock size, and climate can affect humans too (Harley *et al.* 2006). Combining the influence of fishing pressure and environmental changes caused a collapse of the Caribbean coral reef ecosystem, removing herbivores and leaving only sea urchins to control macro algal growth (Bellwood *et al.* 2004, Scheffer *et al*. 2005, Harley *et al*. 2006). The human response to a collapse of fisheries it to shift efforts to another resource lower down the food chain (Hamilton *et al.* 2000), this shows that it is likely that our response to the effects of climate change will impact the ecological outcomes with shifting direct pressures (Harley *et al.* 2006). Instead of altering our efforts to become more sustainable we often just move our efforts, eventually leading to similar outcomes just on another species.

An ecological community with a wide array of diverse species is known to increase both the productivity and stability of ecosystems through the roles they may play and interactions such as mutualisms, functional redundancy and facilitation (Zann *et al.* 1984, Spalding and Brown 2015). Fish communities play an important role in this species diversity and also for human societies (Morrison *et al.* 2013, Bonaldo *et al.* 2014). Seafood has been a primary source of protein, in particular for many coastal and island nations (Zann *et al*. 1984, Hughes *et al.* 2012, Morrison *et al.* 2013). They provide economic gains for both local and major fisheries, also for tourism (Zann *et al.* 1984). A key component in this is herbivorous fish and the role they play for the coral reef ecosystem.

Herbivorous fish

Herbivorous fish are key components in bioerosion and recycling of sediment on the coral reef. They feed on algae, through scraping part of the substrate where the algae is attached, which is then digested and excreted as fine sediment (Adam *et al.* 2015). Encrusting algae helps to consolidate the fine sediment as a platform for coral polyps to settle and recruit (Adam *et al*. 2015). Removing herbivorous fish, such as parrotfish and surgeonfish, not only alters the size of the stock but also the dynamics of the reef (Hughes *et al.* 2007). Reduced herbivory can lead to increased risk of coral disease, which prevents reef recovery after events such as bleaching, and therefore, the coral reefs' overall resilience (Hughes *et al*. 2003). The removal of macro algae by herbivorous fish is a key process on the coral reefs (Streit *et al*. 2015) that moderates the competition for benthic space and facilitates the recovery of coral populations following macro algal dominance (McCook 1999, Hughes *et al.* 2007, Streit *et al.* 2015). Populations of herbivorous fish may also play an important role in the resilience of coral reefs to temperature changes and weather storms, as the grazers provide time and space for damaged and dead corals to regrow (Edwards *et al*. 2011, Bonaldo *et al.* 2014). Maintaining herbivore species richness appears critical for preserving coral reefs, as complementary feeding by diverse herbivores produces positive, but indirect, effects on corals the foundation species for the ecosystem (Burkepile and Hay 2008).

Pollution is a serious anthropogenic influence on the coral reef ecosystem, such as terrestrial run off of sediment and urban pollutants, including agricultural waste products (Global Environment Facility 2013). These can have damaging effects and have continued to increase as urbanisation increases, with greater mangrove removal and river catchments being altered (McCulloch *et al.* 2003). When the water becomes enriched in nutrients there becomes interference with the coral growth, influencing the fish and other organisms that rely on the coral (Spalding and Brown 2015). Nutrient enrichment can lead to the changes, such as macro algae dominant reefs, as the increased nutrient levels also support algal growth through eutrophication. Eutrophication can affect the physiology and also the behaviour of fish, such as their anti-predator behaviour, which in turn alters their survival (Domenici *et al*. 2007). Grazing by herbivorous fish has been identified as having the ability to prevent these macro algal phase shifts (Ledlie *et al*. 2007, Holbrook *et al.* 2008, Cheal *et al.* 2013), which is a common sign of loss of resilience (Dudgeon *et al.* 2010, Cheal *et al*. 2013). The fishing removal of herbivorous fish and excessive nutrients from land based activities has led to ecological shifts causing these alternate states (Nyström *et al.* 2008, Welsh and Bellwood 2012). This phase shift from coral reef to macro algae state, referred to by Hughes (1994) as an algal bloom still persisting, occurred in the Caribbean with the removal of herbivorous fish through fishing pressure (Hughes 1994). When a coral reef has gone through this phase shift to macro algae dominated it has been suggested that herbivorous fish would need to be reintroduced into an area to attempt to reverse the state (Mumby *et al.* 2007). However, the amount of algae can be too excessive and established for herbivorous fish, such as the parrotfish, preventing them from coming into an area or being capable of removing large mature macro algae (Bellwood *et al*. 2006, Streit *et al.* 2015). With this understanding of difficulty in reversing phase shifts the best course of action is therefore prevention in the first place, as returning to the desired state is more work and may not be possible (Hoey and Bellwood 2008). Shifts in abundance of macro algae, or the population dynamics of herbivorous fish may be a potential measure of resilience in the coral reef ecosystem.

Parrotfish

Parrotfish (Labridae, formerly Scaridae), particularly larger bodied species, are economically valuable in fisheries and tourism (Smith 1993, Page 1998, McClanahan *et al.* 1999, Howard *et al.* 2013). Parrotfish are a prominent clade of 99 species that shape coral reef communities worldwide through their actions as grazing herbivores (Kazancioglu *et al.* 2009, Parenti and Randall 2011); approximately 80% of all the species are coral reef associated while the remaining 20% reside in seagrass (Streelman *et al.* 2002). Parrotfish have high species diversity, a conspicuous nature, and considerable abundance of many species (Sale 1991, Streelman *et al.* 2002, Taylor *et al.* 2015). Most parrotfish species are generalists in both habitat use and feeding and have multiple centres-of-origin. For example, genera such as *Scarus* and *Chlorurus* probably originated in the Indo-Pacific (Streelman *et al.* 2002, Kazancioglu *et al.* 2009), compared to *Sparisoma* which originated mainly in the Caribbean, a region with a very different history such as a recent mass extinction (Jackson *et al.* 1996, Kazancioglu *et al.* 2009). Of the 80% that are coral reef associated species, the *Chlorurus* and *Scarus* genera comprise approximately 76% (Streelman *et al.* 2002, Michael 2009). Phylogenetically nested within Labridae, the ecological impact and high species richness of parrotfishes suggest that their diversification and ecological success may be linked (Kazancioglu *et al.* 2009). They can inhabit a variety of habitats such as mangroves, seagrasses and coral reefs (Ogden and Buckman 1973, Feitosa and Ferreira 2014). The food availability within these habitats alters how they feed and the species found in each habitat (Wolf 1985, Nash *et al.* 2012, Feitosa and Ferriera 2014).

They are generally a long lived species, with larger species tending to live longer, the blunthead *Chlorurus sordidus* and bridled parrotfish, the *Scarus frenatus,* may live to 20 years (Michael 2009). Currently, despite widespread fishery-induced pressure on Pacific coral reefs, the structuring of parrotfish on a spatial scale is largely due to habitat (Taylor *et al.* 2015). Parrotfish typically demonstrate a weak response to fishing in areas where fisheries are aimed at higher trophic levels (Russ and Alcala 1998, Taylor *et al*. 2015), but they have strong species level responses to fishing pressure (Clua and Legendre 2008). These responses are driven by differing life histories among species, suggesting a high capacity for fishing induced changes in parrotfish assemblages. The artisanal and traditional fisheries can be highly focused on the grazers, and even though they are small, their removal can have a large localised impact on the ecosystem function (Bellwood *et al*. 2004). Similarly, growth in the aquarium trade has focussed fishing on parrotfish for supply to many countries; for example, where they are being sold in London after being collected from the Seychelles and Persian Gulf (Bellwood *et al.* 2004). The additional fishing pressure on the Indo-Pacific is growing with the greater demand coming from Asia, and herbivorous fish are now ranked as the second largest group targeted for exploitation based on their biomass (Bellwood *et al.* 2004). Tens of millions of people depend on the coastal coral reef fisheries for their livelihood and protein intake (Salvat 1992, Moberg and Folke 1999). For example, in the Caribbean many populations of fish, including parrotfish and surgeonfish, were fished to functional extinction as the fisheries moved from one species to the next (Koslow *et al.* 1988). Decades of overexploitation of herbivorous fish have been implicated in the 1983/4 region wide shift to alternate macro algae dominated states (Hughes 1994, McManus and Polsenberg 2004).

Most parrotfish species are protogynous hermaphrodites that change colour and sex, from initial phase females or males (IP) to terminal phase males (TP) (Koltes 1993, Bonaldo *et al.* 2006). Many fish living in reef environments display remarkable flexibility in sexuality with social interactions determining their sex either during juvenile development or in adulthood (Godwin 2009). Hermaphroditism is the expression of both male and female reproductive function in a single individual (Mitcheson and Liu 2008). Hermaphroditism is taxonomically widespread among teleost fishes (Frisch 2004), and takes on many forms including simultaneous, protogynous, and protandrous hermaphroditism, bidirectional sex change, and androdioecy (Erisman *et al.* 2013). Functional hermaphroditism is noted in 27 teleost families in seven orders, mainly in the tropical species (Mitcheson and Liu 2008). A study by Hawkins and Roberts (2003) has provided evidence that protogynous hermaphrodites are particularly sensitive to overfishing and, therefore, may make them more sensitive to other anthropogenic and natural impacts. Fish reproduction and maturation are regulated by endogenous substances and exogenous factors, such as photo period and temperatures, so that reproduction occurs at an optimal time (Arukwe and Goksøyr 1998). Within the parrotfish, protogyny has been confirmed in five genera; *Chlorurus, Caltomus, Cryptotomus, Scarus* and *Sparisoma* (Mitcheson and Liu 2008). A diandric protogynous hermaphrodite has two ways in which a fish can become male. One way is as a primary male maturing directly to a male, or being a secondary male that first reproduces as a female and then changes sex to become a male (Munday *et al.* 2006). The differences between the primary and secondary males are due to the environmentally induced differences of their timing to male development (Munday *et al*. 2006).

What does cause confusion in parrotfish is that the body colour, testes development and reproduction function do not always match (Micheson and Liu 2008). The classical theory states that the largest female will change sex (Ghiselin 1969, Warner 1988, Yamaguchi *et al.* 2013). An example of this is within the genera *Chlorurus, Scarus* and *Sparisoma*, where the male's testes can be showing ovarian structure but can be smaller than other females in the same local population; they may not have derived from functional females (Choat and Roberston 1975, Robertson and Warner 1978, Mitchseon and Liu 2008). This supports the fact that the largest female doesn't always have to change when the male is removed, therefore, the secondary male may be smaller than other females in the population. A study on triggerfish opposed this classical theory with smaller females changing sex (Yamaguchi *et al.* 2013). They found that with field research and models they can predict the females that are less fertile change sex earlier (Yamaguchi *et al.* 2013). Other papers support the classical theory, noting that the larger female does change (Robertson 1972, Kuwamura 1984). Sex changes without males being removed or leaving has also been reported in a couple of studies (Moyer and Zaiser 1984, Aldenhoven 1986, Sakai 1997). Hermaphroditism in coral fish is often due to monopolization of mating by larger males (Warner 1984). Often small males will 'sneak' in to another larger male's territory to reproduce, this has also been shown to occur in wrasses (Warner 1984). The social control of reef fishes has been documented from the 1970s (Fishelson 1970, Roberston 1972, Warner and Swearer 1991).The mating system of reef associated species can vary with habitat and territorial behaviour, whereas seagrass species are generally not territorial. In comparison, on the reef some are permanently territorial and others only during the mating period (Michael 2009). Buckman and Ogden (1973) noted that the striped parrotfish, *Scarus iseri,* had different social classes in Panama; territorials, stationaries and foragers. A territorial is a male with a dominant female and other females defending an area, they make up the harem. Stationaries are site attached initial phase males and females and terminal phase males that do not defend an area. Foragers roam the reef generally in large groups, many initial phase females, this species breeds in pairs and groups. Other fish without permanent territories can develop lek-like spawning territories where females visit and they usually pair spawn, although this has only been observed in Puerto Rico

(Michael 2009, Munoz *et al.* 2014). In the spawning period, the colour of males often intensifies with aggression behaviour, as they chase conspecifics from their territory (Michael 2009). If a female is responsive to males in their territory then they will begin a spawning ascent together (Michael 2009). The initial phase males may try streaking in these environments in an attempt to join in on the group spawning (Munoz and Warner 2004). Spawning occurs year round but may peak in the summer months, typically this occurs in the afternoon and may be effected by the tide (Yogo *et al.* 1980, Michael 2009).

Feeding activities make parrotfish some of the ecologically most important fishes on modern coral reefs (Bellwood 1995, Hughes *et al.* 2007, Hoey and Bellwood 2008, Kazancioglu *et al.* 2009); it is their feeding apparatus that gives them their name due to its beak like appearance. Parrotfish species are generalist feeders, which means they can depend on food types other than algae such as detritus, crustaceans, sponges, gorgonians and dead or live coral (Randall 1967, Bruggemann *et al*. 1994, McAfee and Morgan 1996, Ferreira and Goncalves 2006, Feitosa and Ferreira 2014). Adults tend to be more abundant in the shallower regions and this is where they have greatest effect. The size of the fish and their feeding behaviour are important factors in their effect on erosion. Larger parrotfish produce more sediment, and excavators will produce more than scrapers, which in turn produce more than browsers (Michael 2009, Feitosa and Ferreira 2014). They are a major producer of the sand around the reef. It has been estimated that one scarid may deposit up to 2700kg of sediment in a year (Michael 2009). Early modifications of the pharyngeal and oral jaws in the evolutionof parrotfish may have contributed to their high diversity by establishing parrotfishes as unique reef herbivores (Kazancioglu *et al.* 2009). A series of functional innovations in their feeding mechanism allow parrotfishes to scrape algae from the surface of hard substrates and to pulverize and digest the mixture of algae, bacteria, detritus, benthic invertebrates, dead coral skeletons and sand (Clements and Bellwood 1988, Bellwood 1994, Wainwright *et al.* 2004, Kazancioglu *et al*. 2009). There are different types of feeding parrotfish, such as the bioeroding parrotfish the *Bolbometopon muricatum,* where each individual can ingest up to 5000 tonnes of coral annually (Bellwood *et al.* 2004). Another feeding type is the scraping parrotfish such as the *Scarus flavipectoralis*, which removes the epilithic algae and sediment (Bellwood *et al.* 2004).

The Kingdom of Tonga

The Kingdom of Tonga is located in the central South Pacific, and almost three quarters of the entire population is located on one of the four main island groups, Tongatapu (Lovell and Palaki 2000). Tongatapu is a raised coral platform; here, the major coastal water body is Fanga’uta Lagoon

(Morrison *et al*. 2013). The resident population in the lagoons catchment was estimated at approximately 40,000 in 2013 (Morrison *et al.* 2013). Over the years the lagoon has suffered many anthropogenic impacts including coastal modification, such as the removal of vegetation, runoff from the roads and farmland, sedimentation especially with the removal of mangroves, polluted groundwater inputs, overfishing and dumping of waste (Chesher 1985, Kaly and Morrison 2005).

Pacific islanders have a large dependency on coastal resources for food, income, culture and recreation (Muller *et al*. 2000). Challenges these pacific communities face are due to expanding populations, creating greater income and food needs leading to over exploitation of these coastal resources (Muller *et al*. 2000). As seen in the Solomon Islands, there is increased migration of the population to the urbanized areas, posing greater threats to the adjacent coral reefs (Aswani and Sabetian 2010). Many of these Small Island Developing States (SIDS) are vulnerable to anthropogenic climate change (Mavrogenis and Kelman 2013). Tonga has 172 named islands of which 36 are inhabited (Mavrogenis and Kelman 2013), here the main source of livelihood is agriculture and fishing, much of this is subsistence based. Most of Tonga's islands are low and flat with the average altitude of 2-5 metres, such as Tongatapu, making them vulnerable to storm surges (Mimura 1999, Mavrogenis and Kelman 2013). And with forecasted increases in global sea level and increased frequency of storms this means greater damage to the island and the coral reefs with less recovery time (Mimura 1999). Climate change effects in Tonga are expected, with moderate confidence, to be an average increase in sea surface temperature of 1.9°C, and an average sea level rise of 39 cm, compared to the 1990 baseline (MESCAL 2013). With this vulnerability, Tonga is also suffering from impacts of environmental degradation from both internal and external anthropogenic pressures (Mavrogenis and Kelman 2013). Common pollution problems in Tonga have been described by Morrison and Munro (1999) as lack of government priority; support, finance, long term planning; poor landfill position; lack of awareness of waste management and overall poor handling of waste.

There is no central sewer system in the Kingdom of Tonga (National IWRM Diagnostic Report 2007, Global Environment Facility 2013). Therefore, wastewater is managed by the community with permits from the Ministry of Health. This commonly leads to poorly constructed systems, which leak pathogens and nutrients into the groundwater (Naidu *et al.* 1991, Morrison 1997, Global Environment Facility 2013, Morrison *et al*. 2013). These nutrients have been noted to be affecting the health of the near shore reefs in Nuku'alofa area and the lagoon with increased algal growth (Global Environment Facility 2013). Concerns then spread to the fish that are harvested in these areas, including the shellfish. In the Fanga'uta lagoon there has been concern around the metal contamination in the lagoon, this is due to dumping and the groundwater flowing into the lagoon from terrestrial contamination sites (Morrison and Brown 2003). Shellfish from the lagoon were studied in 1988, and again in 2003, and in both cases they found low abundances where high contamination was measured (Naidu *et al.* 1991, Morrison and Brown 2003). Contamination of organochlorine pesticides and polychlorinated biphenyls (PCBs) at low levels were found in 1992 in lagoon sediments (Harrison *et al*. 1996, Morrison *et al.* 2013), close to a known illegal dumping site. In a complementary study in 2000, low concentrations of chlorfluazuron and flusilazole were found in sediments, probably originating from use in nearby agricultural fields (Chen *et al.* 2000, Morrison *et al.* 2013). In contrast, Morrison and Brown (2003) identified the metal concentrations in sediments were not at significant concentrations.

Urban development and illegal cutting have largely decreased the stands of mangroves during the past decades (Global Environment Facility 2013, Morrison *et al.* 2013). Pigs are commonly found living on wastes in the mangroves and cause local areas of erosion and mangrove loss (Global Environment Facility 2013). These pigs may also increase the nutrients into the water ways through their waste (Prescott *et al.* 2007). There are high levels of human impact on the mangroves of the lagoon system (Global Environment Facility 2013, Morrison *et al.* 2013). Major habitat loss may be reducing the abundance of juvenile fish due to loss of nursery habitats connected to the reef environment (Morrison *et al.* 2013). A study in one village of Nukuhetulu in the Kingdom of Tonga, by Prescott *et al.* (2007), found there was organic rubbish being put into the lagoon and mangrove areas, this is a typical problem for most communities and threatens the lagoon and fish ecosystem. Pit latrines are also common in low socioeconomic villages, and are polluting the ground water (Prescott 2001, Prescott *et al.* 2007). Many farmers also use pesticides, which can contaminate groundwater (Prescott *et al*. 2007). Combining the influence from the land on the coral reef environment with overfishing of the coral reef inhabitants leaves the reefs and associated species vulnerable.

Tonga’s fishing pressure

The Fanga’uta Lagoon, in Tongatapu, has been identified as having approximately 96 species of fish (Zann *et al.* 1984, Prescott *et al.* 2007, Global Environment Facility 2013). Recent scientific studies in Tonga have shown that, in many places, coastal areas are already exploited at, or beyond, their maximum capacity (Forth report: Review of Tonga National Biodiversity Strategy and Action Plan, Nuku'alofa. 2010, Global Environment Facility 2013). Historically, the inshore parts of Tonga have been subject to heavy fishing (Global Environment Facility 2013), as the entire coastline is open for fishing to everyone. Prior to the Tongan constitution of 1875 for nearshore marine areas, coastal fisheries were under control of community chiefs (Kronen 2003, Sun *et al.* 2011). After 1875 all people had the right to fish or gather marine resources, there were no exclusive fishing rights (Kronen 2003, Sun *et al.* 2011). Traditionally Tongans only fished inside the reef (Bell *et al.* 1994). The commercial fishing is only 3% of Tonga's gross domestic product (GDP) (Gillett 2009, Sun *et al.* 2011); however, subsistence fishing is practiced by a majority of people (Zann 1994, Sun *et al.* 2011). Methods include the use of handlines, spears and gill netting (Kronen and Bender 2007, Sun *et al.* 2011), although a combination is generally used (Sun *et al.* 2011).

Since the early 1970s, to meet the increased demand for fresh fish, traditional subsistence fishing techniques were replaced by more efficient monofilament gillnets, arrowhead fish fences, and a trawl fishery for penaeid prawns, and the use of explosives was common (Zann, n.d., Global Environment Facility 2013). In 1975, commercial fishing in the lagoon was banned in an attempt to reduce fishing pressure, but without strict enforcement (Global Environment Facility 2013). Currently, the lagoon fishery is broadly called subsistence, although many fishers sell some of their catch at the local markets (Global Environment Facility 2013). The decline noted by fishermen in size and abundance of many species in the catches of the lagoon appeared to be due to a combination of increased fishing pressure, loss of habitat area and quality, and coastal developments (Ellison 1999, Global Environment Facility 2013). Information gathered during household surveys in 2001 reveals that quantity and quality of fish and shellfish catches in the lagoon had declined over the years and were continuing to decline rapidly (Prescott *et al.* 2007, Global Environment Facility 2013). Fish stocks are now significantly reduced in mangrove areas, the lagoon and bays, and on near-shore coral reefs. Overfishing resulted in the closure of the lagoon to commercial fishing during 1975-1981 (Global Environment Facility 2013). However, there is limited effective management and conservation of inshore fishery resources with no action taken to implement minimum harvest size, or impose controls and closed fishing seasons (Global Environment Facility 2013).

Tongan local fisheries have included the use of nets over the coral reefs. There has also been use of dynamite to blow up reefs and stun fish, allowing fishers to collect them at the top of the water as they float (Tuiano, R. T. pers. comm. 2015). Chemicals such as cyanide have also been used, which not only effects the target fish but also all fish and organisms in the vicinity (Dee *et al*. 2014). Parrotfish cannot be caught through the use of line and hook, as they cannot be caught through baiting. Many night fisherman dive or spear gun these fish, when they are asleep in their cocoons they are seen hiding amongst the coral, caught using nets or guns (Tuiano, R. T. pers. comm. 2015). Cyanide has been used to stun and collect targeted aquaria fish species (Noyes 1976, McAllister *et al.* 1999, Wood 2001, Vaz *et al.* 2012). Cyanide is a toxic chemical, given to the fish at a sublethal dose, and can incur mortality and wastage (Randall 1987, Rubec 1998). As many of the fish hide in burrows, and amongst rocks and coral, trying to remove them may damage the fish and also the coral; therefore, the use of tranquilisers and anaesthetics is often applied across coral reefs for easier capture and to increase the catch (Wood 2001). Hand nets and barrier nets were most commonly used to persuade fish from their refuge in the reef (Wood 2001). The five major species caught through spearfishing and diving included the parrotfish, both in day and night (Vaikona *et al*. 1997, NSFO 2010). During night diving the parrotfish *Scarus* species made up the highest percent of the catch at 29%, followed by rabbitfish and unicornfish (Vaikona *et al*. 1997). They were the second highest fish captured during day dives, at 13%, following behind surgeon fish at 38%.

The parrotfish fishery has long been part of the subsistence fishery. Parrotfish are the most common group of fish within the Labridae family, sub family Scaridae, in Tongatapu (Lovell and Palaki 2000) and they are the main finfish resource from the shallow-water reefs, providing the primary source of protein for local people (Lovell and Palaki 2000). Fish demand in Tonga based on 2007 Food and Agriculture Organization (FAO) Food Balance Sheet is 35 kg per capita consumption per year (FAO 2009, NSFO 2010). In 2010 this translates to a demand for 3627 tonnes of fish (NSFO 2010). This demand may be influenced by emigration, fish price increase, cost of fish substitutes and changes in dietary preferences (NSFO 2010). The ability to collect this much fish will be affected by overfishing. Fish is an important source of protein, with FAO Food Balance Sheets (2007, FAO 2009) showing that fish constitutes 13.5% of protein in the diet; higher in the rural areas. Any substitutes are generally imported fatty meats which are negative for health (NSFO 2010). Tongan Fisheries Project 2005 found that of 64% of households on Tongatapu fished for their own seafood supply and for gifts to others, whereas the more rural further afar places such as Ha'apai and Vava'u were around 80%. A report from the Tonga Statistics Department (2004, NSFO 2010) gave results from employment in 2003, showing that the fisheries represented 3% of employment in the country.

Although resilience of Pacific Ocean reefs has commonly been noted, this has largely been due to the high abundance of herbivores which are absent on other worldwide reefs (Bruno *et al.* 2009, Adam *et al*. 2011). With the effects of the anthropogenic impacts increasing in affiliation with suspected negative influence on these herbivorous fish, such as the parrotfish, there is a real possibility that there could be a state change to become macro algae dominated. There is unfortunately not a lot of scientific study completed on the Tongan coral reef ecosystem and its herbivorous reef fish (but see, Zann 1994, Randall *et al.* 2003, Clua and Legendre 2008, Friedman *et al.* 2009, Adjeroud *et al.* 2013). From these studies, there has been a comprehensive list of fish in the area; however, this is only presence and not abundance of these fish (Randall *et al.* 2003). Clua and Legendre (2008) identified fishing pressures on parrotfish, and that shifts in dominance within the scarid family due to overfishing pressure may mean smaller bodied species replacing the targeted larger bodied species, therefore influencing the role they play in the reef environment. Fishing pressure caused a decline in average size of some species due to genetic loss, and was shown to vary across sites dependent on fishing pressure (Clua and Legendre 2008). This study was completed at sites across the entire Tonga islands group and not only in the locations with the greatest anthropogenic influences. Another comprehensive study with a similar issue was completed by Friedmen *et al.* (2009), it provides a baseline for fisheries species indicating health of the reef system is poor without explicit testing, but was completed Pacific wide therefore had a limited number of sites across island groups. Further study across a variety of sites and years, will provide greater knowledge for the parrotfish composition and dynamics in the near shore coral reefs of Tongatapu.

Research and monitoring of reef fish

Recently, greater focus has been put on measuring and monitoring resilience for conservation and management of coral reef ecosystems, especially in the wake of increasing anthropogenic effects (Bachtiar *et al.* 2011, Graham *et al*. 2013, Heenan and Williams 2013). Examples of this include the South Pacific Islands, with continually more MPAs and no take zones being implemented (Adjeroud *et al*. 2009, Halpern *et al.* 2013, Heenan and Williams 2013). These are implemented, to enhance reef resilience to anthropogenic disturbances, such as fishing pressure, pollution, damage and climate change; and to natural disturbances, such as Crowne of Thorne (*Acanthaster planci*) outbreaks, hurricanes and storms (Done 1987, McClanahan *et al.* 2002, Berumen and Pratchett 2006, Cheal *et al.* 2010, Carassou *et al.* 2013).

Many baseline data sets have been taken before the implementation of management to have a comparison for the management benefits to be seen against. Baseline data often includes all aspects of the coral reef ecosystems, such as benthic coral and algae cover, benthic organisms and pelagic organisms. Baseline data sets are taken before management is implemented to identify any changes that occur under the management protocol (Edgar *et al.* 2004, Bergseth *et al*. 2015). These are necessary to identify if there is success of the management that is being implemented (Edgar *et al*. 2004), this can be reflected in the reef fish communities in terms of species composition and size/age structure for example (Bohnsack *et al.* 1999). Reserves have been put in place with data-less management through the use of the precautionary principle (Johannes 1998, Lauck *et al.* 1998). This means acting without any data, to manage an ecosystem before there are severe effects on the resources (Johannes 1998). Even though this can be applied, where possible a baseline data set is optimal (Dayton *et al.* 2000, Edgar *et al*. 2004, Stewart *et al.* 2008).

Baseline data is essential for understanding the ecosystem and giving a reference point for management (DAI and MERF 2003, Edgar *et al*. 2004, Bergseth *et al*. 2015). This baseline data focuses on indicators of reef resilience, which herbivorous fish have been identified as, such as the parrotfish (Mumby and Steneck 2008, Green and Bellwood 2009, Cheal *et al.* 2010). Fish base line data, and general observations on the reef often involve scuba diving, recording videos and collecting physical, chemical and biological data (Hsiao *et al*. 2014). Underwater visual census (UVC) techniques are the most accepted and widely used method for scientific reef surveys on fish since the initial development of the approach by Brock (1954) (Colvocoresses and Acosta 2007). The logistics of sampling every site and ecosystem for baseline data is often difficult, this means that the precautionary principle does have to be applied in some cases (Johannes 1998). However, there is an alternative with the use of citizen science and community management (Mumby *et al*. 1995, Edgar and Stuart-Smith 2009). Volunteers and organisations are available to collect this baseline data, and the monitoring data necessary for management regimes (Mumby *et al.* 1995, Gillett *et al.* 2012).

The methods used for data collection is important. Previous works have assessed methods used to monitor, regulate and manage reef communities, with varying strategies being identified as better suited for different research aims (Dee *et al.* 2014). When assessing the suitability of either method, availability of time, funding, and relevant expertise should be primary considerations (Holmes *et al.* 2013). Researchers need to consider carefully the target species and employ the appropriate method to achieve the research goals (Willis and Babcock 2000).

1.1 THESIS AIMS

There are two aims of this thesis: 1) to investigate shallow reef methods for fish counts and behavioural studies. This is completed through a comparative assessment of two UVC methods, and 2) to investigate parrotfish population dynamics, such as diversity, abundance, harem structure and behaviour to create a spatiotemporal baseline data set.

The following chapters provide information on achieving these aims. Chapter two compares two methods of underwater visual census, using a territorial fish species (damselfish) and a pelagic fish species (parrotfish). Chapter three uses the method from Chapter 2 most appropriate for investigating parrotfish and their population dynamics. The final chapter forms the discussion over the results of these aims and the contribution this research has made. Also the further research opportunities and applications of the data collected here.

2 CHAPTER 2 – METHODOLOGY

2.1 INTRODUCTION

Underwater visual census (UVC) is a common tool for detecting and quantifying range shifts of tropical marine fishes (Booth *et al.* 2007, Booth *et al.* 2011, Wernberg *et al*. 2012, Afonso *et al.* 2013, Assis *et al*. 2013, Nakamura and Yamaoka 2013, Nakamura *et al.* 2013, Beck *et al.* 2014). The development and expansion of this non-destructive method was established by Brock (1954) in response to limitations of diver-based methods and an increase in the use of non-experts for reef monitoring programmes (Barry and Coggan 2010, Lowry *et al*. 2012). Now UVC is one of the most widely used methods and is highly regarded for studying coral reef fish communities (Samoilys and Carlos 2000, MacNeil *et al*. 2008, Bernard *et al*. 2013, Lindfield *et al*. 2013, Pita *et al*. 2014). The move to UVC began with its use alongside some of the older fisher based methods. For example, mark recapture was used on coral trout on the Great Barrier Reef and involved hook line fishing, marking fish with a freeze brand number and later using UVC for resighting of the marked fish (Zeller and Russ 1998, Zeller and Russ 2000). Population dynamics and fisheries impact studies have also relied heavily on data from fisherman, however, the reliability of these have often been in question (Wood 2001). In the collection of data, fisheries have used multiple methods, which have involved chemicals such as cyanide (Noyes 1976, McAllister *et al*. 1999, Wood 2001, Vaz *et al*. 2012) and quinaldine (Randall 1987). Other methods for stock assessments include hand and barrier nets (Wood 2001), traps, and tag-recapture (Randall 1961, Randall 1963, Miller and Hunte 1987). These fisheries methods are in general more invasive with negative impacts occurring on the reef fish and habitat, hence the advancements towards video and diver sampling techniques. The early studies employing only UVC methods were largely used to assess patterns of species abundance and distribution (Sale and Douglas 1981, Samoilys and Carlos 2000, Edgar *et al.* 2004). Increasingly, UVC are being used to assess fish populations (Ward-Paige *et al.* 2010, Pelletier *et al.* 2011) as they can census reef fish in complex habitats or shallow reef areas (Murphy and Jenkins 2010). Underwater Visual Census methods have also been used to assess mobile benthic macroinvertebrate populations (Stuart-Smith *et al.* 2008). The use of UVC as a non-invasive and non-destructive observation method is particularly important for surveying in and around Marine Protected Areas (MPAs) (Pelletier *et al*. 2011).

Instead of using people in the water for surveying, there has begun to be a shift to remote camera and video control out of the water (Murphy and Jenkins 2010). Cameras and videos can be operated in multiple ways to best capture the target, such as; those operated by divers (e.g., Harvey *et al*. 2001, Cappo *et al*. 2003), cameras towed behind boats (e.g., Giddens *et al*. 2014), remote underwater video (RUV)) (e.g., Mallet *et al.* 2014, Pita *et al.* 2014), autonomous underwater vehicles (AUV), remotely operated vehicles (ROV) (e.g., Willis *et al.* 2000, Willis and Babcock 2000, Westera *et al*. 2003, Watson *et al.* 2005, Watson *et al*. 2007), and unbaited or baited remote underwater video (e.g., Willis and Babcock 2000, Willis *et al.* 2000, Harvey *et al.* 2004). Video method are therefore useful to collect data in aquatic environments that are complex or that include inaccessible habitats (Colton and Swearer 2010, Unsworth *et al.* 2014). Manned submersibles, ROVs, AUVs, and towed camera platforms have been successfully used for abundance assessments and to investigate species habitat associations (Seiler *et al*. 2012). Video transect surveys are faster than UVC surveys as the diver does not have to identify and measure species while conducting the surveys (Harvey *et al.* 2001, Murphy and Jenkins 2010), however, more work is required outside of time spent in the field (Pita *et al.* 2014). Video techniques and their use are generally bulky, heavy and costly in terms of ROV and RUV compared to divers alone (Pita *et al.* 2014), although in terms of accuracy they have been regarded as effective methods, such as Baited Remote Underwater Video (BRUV) with their use on carnivorous reef fish (Cappo *et al.* 2007, Gardner and Struthers 2013). The BRUV uses a bait to either attract specific species or a community of fish (Pita *et al*. 2014) has been used worldwide, for example in the Philippine islands, American Samoa, Fiji and on the Great Barrier Reef (Borard *et al.* 2014). It is a useful tool that can be used at depths of 300-400 metres (Borard *et al*. 2014). Factors involved in the use of bait include the concentration of the chemical scent moving into the surrounding waters which is virtually impossible to quantify (Borard *et al*. 2014). Baited Remote Underwater Videos are used to decrease the amount of zero counts, while they increase similarity and repeatability between surveys (Cappo *et al*. 2007, Murphy and Jenkins 2010). This method complements UVC as it provides information of diver adverse species; therefore, it may be most beneficial when used in combination for specific species (Lowry *et al*. 2012). Non baited cameras have also been used to capture the natural distribution and behaviour of organisms, such as the crab *Carcinus maenas* (Sheehan *et al*. 2010).

An autonomous underwater vehicle (AUV), can give estimates of abundance, size and habitat, and has been used to quantify fish such as the ocean perch (Seiler *et al*. 2012). Autonomous underwater vehicle sampling has been used to complement fishery independent surveys of rocky reef species (Seiler *et al.* 2014). Similarly ROVs allow for high replication, but their usefulness is restrained by a restricted field of view and requires time to analyse the videos after collection of data (Murphy and Jenkins 2010). Stoner *et al.* (2008) observed that their largest bias when sampling fish with an ROV was associated with fish attracted to, or avoiding, the ROV outside the field of view (Lindfield *et al.*

2014). As previously mentioned, SCUBA diving is recognized as an intrusive activity (Lobel 2001, Schmidt and Gassner 2006, Dickens *et al*. 2011, Lindfield *et al*. 2014), and the sound it produces has been identified as a significant contributing factor to fish avoidance (Chapman and Johnstone 1974, Lobel 2005, Cole *et al*. 2007). Stoner *et al.* (2008) showed that biases can be caused by movement, lighting, noise of the motors, electric and magnetic fields of the ROV, whereas the use of RUV does not have this impact (Cappo *et al*. 2003, Pita *et al*. 2014). The effects of these biases can influence the abundance count. An example of this is underestimations of abundances recorded by ROV and diver methods, also the variability from RUV collected abundance data, showing that each method has its biases (Pita *et al*. 2014).

The use of underwater cameras is not restricted to remotely controlled video, but they can be fixed and continually monitored to have a set point and area of research for temporal variation. Fixed underwater cameras can be used to continuously record the reef environment, which necessarily avoids some disadvantages of data collected by divers or snorkelers as the fish are not influenced by the sampler's presence (Boom *et al.* 2014). For example, a combination of methods mentioned previously come together in the Station Video Rotative method (STAVIRO) technique used by Murphy and Jenkins (2010) which is an autonomous, remote, unbaited video that rotates by 60 degrees each 30 seconds. Videos are recorded without any external disturbance for around 10 minutes at each site, with at least three complete rotations (Murphy and Jenkins 2010). When STAVIRO was compared to UVC no significant differences were observed between the methods for larger species, but UVC detected a greater number of smaller species. The Station Video Rotative method identified more target species and only UVC detected differences in fish assemblages according to reef type (Murphy and Jenkins 2010). Therefore, using both STAVIRO and UVC methods in combination may be beneficial.

Similarly, handheld underwater video cameras can be used by divers during transect surveys, so divers with limited survey skills can conduct monitoring surveys (Cappo *et al*. 2003). Handheld stereo-video cameras significantly improve the accuracy and precision of fish-length measurements compared with divers visually estimating lengths (Harvey *et al.* 2001). Another method similar to this using video and SCUBA divers is a diver-operated stereo-video transect (DOV), which is regarded as a more precise method than UVC for some data collection, such as fish length estimates (Harvey *et al.* 2007). Both of these methods allow the researcher to manoeuvre through more complex habitats to collect data on more cryptic and smaller species then other video methods (Watson *et al*. 2005, Hui *et al*. 2015). In a comparison of the DOV and BRUV methods, it was identified that the BRUV captured greater numbers in areas such as species richness which may be due to the use of bait and lack of diver presence (Willis *et al.* 2000, Harvey *et al.* 2007, Watson *et al.* 2010). However, DOV recorded greater abundances of Pomacentridae, Labridae and Scaridae species at Ningaloo reef, Australia, demonstrating the need for sampling technique to be taken into account when interpreting fish assemblage structure (Watson *et al*. 2010). Both the handheld and DOV methods put less demand on the divers in the field than using the UVC technique, as the data is collected on the video (Hui *et al*. 2015), unfortunately neither of these methods remove the diver bias (Watson *et al.* 2010).

The sampling plan used with UVC, and the above mentioned methods often involve the use of transects. Transect counts are the most frequently used method for counting fish within a corridor of known length and width (Cappo and Brown 1996, Edgar *et al.* 2004, Kulbicki *et al.* 2010). The width of the transect strip can be dependent on visibility (Pita *et al*. 2014). Two types of transect surveys that are often used in UVC include strip-belt transects and line transects (Edgar *et al*. 2004). In a strip-belt transect, a diver or snorkeler swims along a predetermined length and width of a reef; commonly used for fish surveys where species identification and length are determined by these divers or snorkelers swimming slowly and constantly (Russ and Alcala 1996, Willis *et al*. 2006). This is best in high clarity water, for conspicuous species, ages and sexes, and species with neutral behaviour towards divers (Côté and Perrow 2006). The strip transect method with UVC is particularly successful and widely recognised as the most accurate UVC technique (DeMartini and Roberts 1982, Kimmel 1985, Guidetti *et al*. 2005, Murphy and Jenkins 2010). Linear transects can be completed through continuous recording or sampling along the transect (Stoddart 1972). In contrast, line transects involve swimming along the transect and recording all target species and estimating their distance and direction from the transect line (Kulbicki and Sarramegna 1999). The divers are not restricted by width, but this method relies on them accurately estimating distances (Loya 1972). Line transects are commonly used for coral community structure (Loya and Slobodkin 1971, Loya 1972, Porter 1972, Wallace 1974), and can be split into different categories: modified linear transect (LTM) (McClanahan and Shaffir 1990), the line intercept method (LIT) (Loya 1972) and linear point intercept method (LPI) (Obura 1995). Line point transects are noted as the quickest monitoring system, whilst being as precise and accurate as strip and line transects (Nadon and Stirling 2006).

Another form of visual census is point counts, censusing in a circle or half circle (MacNeil *et al*. 2008). Generally point counts yield higher diversity and density estimates but there is no explanation for the difference (Kulbicki *et al*. 2010). Point counts are also often used for estimating fish densities, this technique is faster than transects, requires only one diver, and the area sampled depends on visibility to a 15 metre radius (Murphy and Jenkins 2010). The point count method involves counting fish within a circular area, while also taking consideration of the boat interference by swimming a number of beats from the boat as explained in the transect method (Samoilys and Carlos 2000).

Citizen science role

Underwater Visual Consensus is the most common method of surveying and monitoring shallow water fish assemblages (Caldwell 2011). The UVC is a popular technique as it requires minimal equipment. However, UVC generally requires a high degree of observer expertise, due to the knowledge required to rapidly identify fishes in the environment (Holmes *et al.* 2013). Despite this, a number of studies have shown that data collected by volunteers are of a similar quality to that collected by researchers, suggesting that citizen-science initiatives can contribute to monitoring objectives (Mumby *et al*. 1995, Darwall and Dulvy 1996, Edgar and Stuart-Smith 2009).

Monitoring programs often have limited funding and limited access to experienced UVC observers (Hassell *et al.* 2013). The use of non-expert observers, predominantly volunteers, to collect data in marine ecological monitoring programs has increased and is becoming a common approach (Pattengill-Semmens and Semmens 2003, Silvertown 2009). Using volunteers to conduct UVC surveys is occurring worldwide (Silvertown 2009), such as; the Philippines (White and Calumpong 1993, Beger 2002), Tanzania (Darwall and Dulvy 1996), the United States (Schmitt and Sullivan 1996, Shuman *et al.* 2008), Australia (Musso and Inglis 1998), Fiji (Léopold *et al.* 2009), and Eastern Mediterranean (Bodilis e*t al*. 2014), through many organisations such as reef check (Gillett *et al*. 2012) and coral cay conservation (Mumby *et al*. 1995). These initiatives attempt to empower locals to become stewards of their marine resources by training them in these UVC techniques (Beger 2002). The use of non-expert volunteers to collect ecological data has been criticised on the basis that the information may be unreliable, inaccurate, and inconsistent due to insufficient observer training, experience and large between-observer biases (Darwall and Dulvy 1996). Studies comparing reef fish survey data collected by experts compared to non-experts or volunteers (Darwall and Dulvy 1996, Pattengill-Semmens and Semmens 1998, Williams *et al*. 2006) have shown that some information is of a sufficient quality to supplement data collected by experts (William *et al.* 2006, Gillett *et al.* 2012), whereas other studies suggest the opposite (Beger 2002, Uychiaoco 2005). However, these different outcomes can be attributed to semantics: the term non-experts in work by Hassell *et al.* (2013) was used to describe Masters students, whereas the term was used in Fiji to describe volunteers, so in comparison to work completed in Fiji they have greater experience. In Fiji, volunteers were snorkelers from nearby villages using their colloquial names for five fish species and non-resident volunteers using a target list of fish with nine local taxa (Léopold *et al.* 2009). The scientifically comprehensive survey in Fiji included all fish identified to species level from twelve families (Léopold *et al*. 2009). Léopold *et al.* (2009) found some inconsistancies between the scientific and volunteer based data, but suggested that the calibration of the community based UVC was partiucalrly necessary.

Bernard *et al.* (2013) searched the influence of observer type, volunteer vs experienced, and fish community structure on the detection probability and count dissimilarity between two observers simultaneously surveying a transect. Community data collected by volunteers were of a lower quality than that collected by researchers. However, this was not the case for the dominant species for which researchers and volunteers produced similar data (Bernard *et al.* 2013). Volunteers were not totally unqualified as they were scuba divers from 1-30 years' experience diving in the region; they also attended standardised two day training courses (Bernard *et al*. 2013). One of the benefits of volunteer programmes is the increased manpower. However, the results from this study suggest that most volunteers produce data that of a lower quality than that from experienced researchers. Bernard *et al.* (2013) identified that certain species were easier to survey than others (Kulbicki 1998), suggesting that volunteer programmes can reduce the observer bias through focussing on a subset of the community.

Field measurements completed by trained volunteer divers were assessed for reliability in artificial reefs by Halusky *et al.* (1994). The conclusion was that volunteer divers offer a credible capability for field sampling and measurement of certain environmental variables when proper training and support services are provided. Volunteers can enhance the scientific observations of research organizations, and reef management (Halusky *et al.* 1994). This is particularly important for projects and research limited funding. The best outcome will occur when they are working with a scientific organisation, as they can provide a credible capacity for field collection of environmental variables when they have the proper training (Halusky *et al.* 1994, Mumby *et al.* 1995, Harding *et al.* 2000).

This dissimilarity sometimes found between the data sets may have a simple fix with alterations to the amount of training and practice completed by the volunteers prior to the research. With sufficient training and practice there was no significant difference found, or it was comparable to the expert data collected (Darwall and Dulvy 1996, Harding *et al.* 2000). Initial training by experienced team members from one study ensured all divers were consistent in recording a range of variables (Pink and Fulton 2015). Through the training of divers the accuracy and precision of data collection was improved substantially (Zeller and Russ 2000). Training includes practicing for size estimates with the use of a range of sizes shown at different depths, distances, and visibility so for those taking a visual census they can then reliably estimate the size of the fish (Mille and Van Tassel 1994, Stuart-Smith *et al.* 2008, Yulianto *et al*. 2015). According to Kadison *et al.* (2002), training a new observer improved the accuracy of the diver's length estimates over 100 percent. A trained diver can therefore improve their precision and learn to achieve an accurate size estimate (Ross 1965). Practice and regular experience will allow for a coordinated group to execute the underwater research techniques and potentially even extend this to long term reef monitoring. Overall, it appears with sufficient training and continual growth of experience that non experts can provide comparable estimates of variables for both target and non-target fish species.

Current research

The aim of my research was to compare the UVC method using snorkelers and hand held videos along transects, to a novel method that to my knowledge has only previously been used by a student at the University of Canterbury (van Lier 2013). This novel method involves using a stand with videos set to record a 360^{o} field of view at set points along a transect. A stand allows the removal of the person, so fish can return to their normal behaviour and activity as they become accustomed to the video stand. This methodology comparison was completed with the use of two target fish, a pelagic fish: the parrotfish and a territorial fish: the damsel fish. Both of these fish are common within the Kingdom of Tonga, are easily identifiable, and due to their territorial and pelagic behaviours, they are likely to vary in response to these methods.

Parrotfishes (Labridae) have been widely recognized as a key component of the coral reef herbivore community in terms of their impact on the ecosystem as grazers and bioeroders of the reef substratum (Bellwood *et al.* 2003, Mumby *et al.* 2006, Fox and Bellwood 2007). A methodology used to demonstrate the overall importance of herbivores in marine ecosystems could yet prove instrumental in furthering our understanding of the balance between corals and algae on reefs and the taxa responsible for maintaining that balance (Fox and Bellwood 2007). Parrotfish are particularly good indicators of fish diversity as they are taxonomically well documented, are a conspicuous core reef fish with diurnal behaviour, and utilize a wide range of habitats in the reef (Allen and Werner 2002). Underwater visual census has been completed on parrotfish previously, with a few of the species being identified as some of the most diver-negative species, being highly mobile and displaying skittish behaviours. This may be because vision is the main stimulus for the fishes encountered on the transects and the divers could be interpreted by fish as predators (Dickens *et al.* 2011). The biomass of roving herbivores recorded by divers was greatly dominated by three species, two of these being parrotfish, and indeed those identified as diver-negative *Scarus rivulatus* and *Chlorurus microrhinos* (Fox and Bellwood 2007). The use of the video method on stands may be particularly beneficial for gathering data on parrotfish due to their dominance and responses to divers.

Damselfish are an abundant fish in the Kingdom of Tonga and are easily identifiable in terms of their appearance and behaviours. These behaviours change with environment and conditions (White *et al.* 2013). For instance, chasing behaviours have been noted to be rare during the daily activities of many reef fish (Pink and Fulton 2015), and the behaviour is usually less than 3 seconds, or less than 1% of the daily time budget of an individual (Layton and Fulton 2014, Noble *et al*. 2014, Pink and Fulton 2014). Therefore, in studies comparing methods for damselfish, very few chasing events were recorded using video UVC compared to manual UVC, partly due to only tracking a few individuals a day, where manual census can rapidly view multiple individuals (Pink and Fulton 2015). On a video the recording of chasing activity is often poorly tracked due to the distance being covered as too rapid (Pink and Fulton 2015). Video-based observations by Ebner *et al.* (2009) found chasing behaviour to be rare, which in their study may have be related to the fixed field of view of their stationary camera, but also this inability to capture the rapid movements. A previous study found roaming surveys were significantly more accurate and precise than belt transects in estimating densities of their focal damselfish, *Abudefduf vaigiensis*, while density estimates did not significantly differ between methods for the rarer species, *Abudefduf whitleyi* (Allen and Robertson 1994). Comparing stand and swim videos may identify which method is best for use in collecting behavioural or abundance data on these territorial fish.

This present study sought to expand on previous work comparing methods for collecting data on reef fish (Cappo and Brown 1996, Samoilys and Carlos 2000, Nadon and Stirling 2006, Colvocoresses and Acosta 2007, Kulbicki *et al*. 2010, Pelletier *et al.* 2011, Beck *et al.* 2014, Pita *et al*. 2014, Pink and Fulton 2015). With one of the aims being identifying which method may be best when researching pelagic or territorial fish. This is important for future research so that through understanding of reactive behaviours and their capacity to collect a variety of data then the correct methods can be used for the target species. The importance of this study lies with the future potential this method may provide and potential improvements of this technique. Here, this research was used to identify which method is best for analysing parrotfish population dynamics within the semi-enclosed lagoon of the island of Tongatapu, of the Kingdom of Tonga (Chapter 3).

I hypothesise that measures of species richness, abundance and behaviour of pelagic and territorial fish using underwater video, will differ between a stationary video stand method and a swimming video transect method.

2.2 METHODS

Swim vs Stand Experimental comparison

Site description

Tongatapu is a raised coral platform with a coastal water body, the Fanga'uta Lagoon situated to the north of the island (Morrison *et al.* 2013). There are many nearby offshore islands and offshore reefs forming a semi-enclosed lagoon structure, and some of these are where sampling took place. The sampling locations used were similar to those used by van Lier (2013), with continuous coral reef flats or patch reefs at <2 metre depth and with varying anthropogenic and environmental factors acting on them. Each of these sites was divided into two stations. These were located approximately 500 metres apart or a fourth of the distance of the reef apart where the reef was smaller, such as in the offshore site around Pangaimotu and Manima where the patchy reefs were not consistent distances apart.

Sampling

Sampling occurred over a two month period in April – June 2015 in daylight hours, sampling 2-3 hours around the high tide. Over this period multiple GoPro cameras were used of varying series (Hero2, Hero 3, Hero3+, Hero4); all set up with 1080 pixels to remain consistent. Both snorkelers came from scientific backgrounds, being Masters students, but did have support from locals, in particular to understand the local names of the species we were identifying and also for their knowledge of the reef fishes spatial variation.

Three crest swims were completed at each location. The GoPro cameras were in a water proof housing mounted on top of approximately 30 cm of plastic tubing using the GoPro adhesive pads (Figure 1). The crest swims were completed along the reef crest at a rate of 1 min per 20 metre (as for the transect swims) for three minutes to cover approximately 60 metres (Figure 1). The swims started from a randomly chosen point of access on the crest that was in close proximity to the back-reef that was being sampled. Each swim was started 10 metres from the end of the preceding swim.

Figure 1. An example of the use of cameras mounted on the tubes used for swimming along transects and crests.

In conjunction with the swims, the second method was trialled on the back reef along a 60 metre transect laid parallel to shore, 10 metres inshore from the reef crest (Figure 2). Here, a stand was used to which three GoPros were attached on a disk. Each of these cameras was set to an 180° field of vision, thereby, capturing a 360° field of view (Figure 3). These GoPros were attached to the stand using the GoPro adhesive mounts placed on a plastic ring attached to the stand at a height of approximately 60 centremetres to show the 360 ° view including the water column and reef. A rope with a buoy was attached in the middle of the cameras so that the snorkelers could identify its location when returning to it. The transect tape was set and the stand was first placed at 60 metres (Figure 2). The cameras were turned on and at the same time, the two snorkelers started their swims to cover 60 metres in three to four minutes, slowly and following the transect. The cameras on the stand were left for 8 minutes, allowing 3 minutes for the fish to return to normal behaviour and 5 minutes for use in analysis. After completion of each 8 minutes, the stand was moved 30 metres along the transect and reset (Figure 2). This allowed three replicates to be recorded on each station. The distances between stations, transects and swims were dependent on the shape of the reef and cover of the reef, particularly in some of the offshore sites.

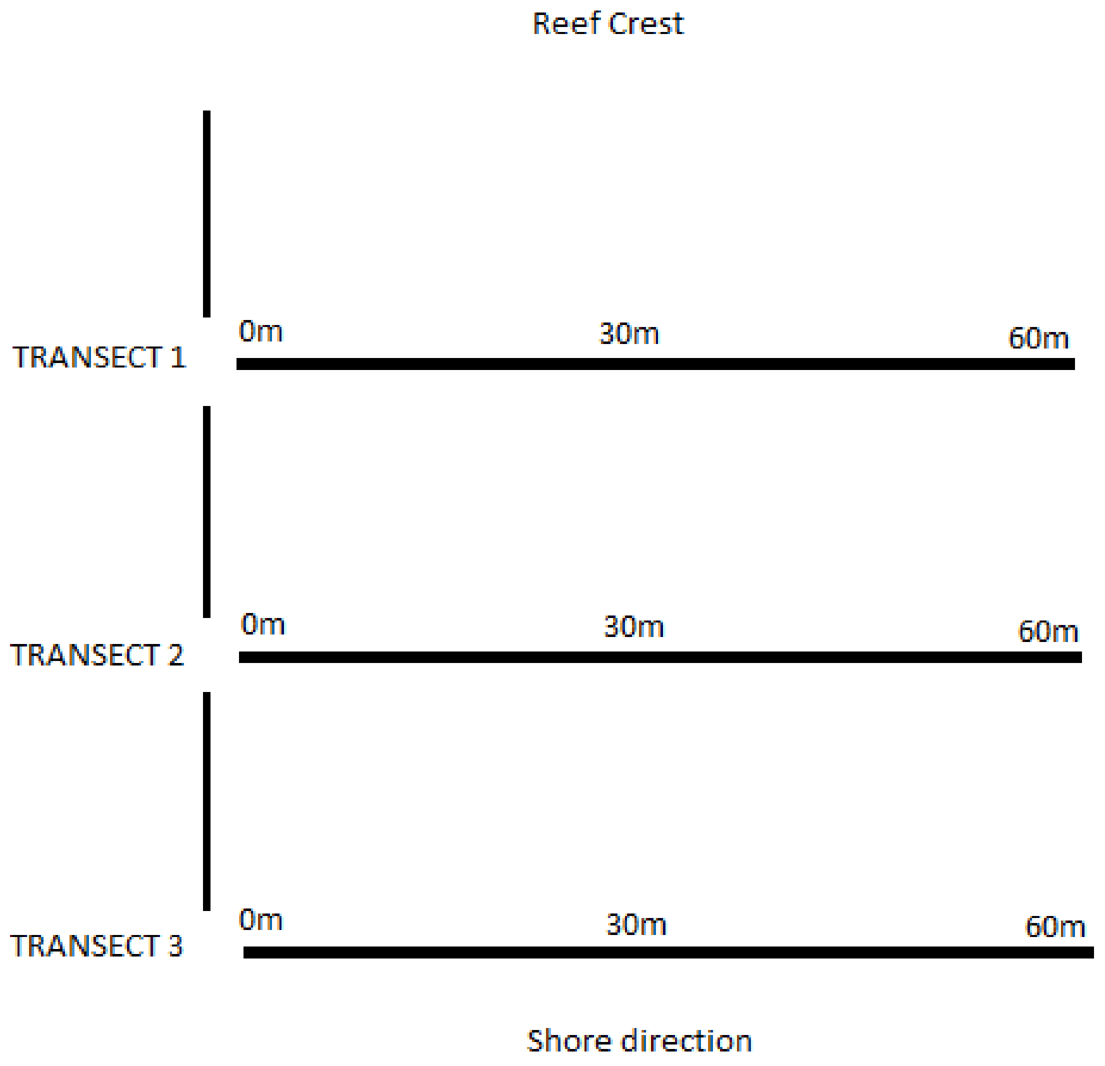

Figure 2. Example of the transect formation used for swim and stand sampling.

Figure 3. The GoPro mount on the novel stand method, for 360 degree video footage.

The video footage was viewed using AVS Video Editor. All damselfish and parrotfish within approximately three metres were then identified and counted and the behaviour recorded. The footage was stopped where necessary for a still capture. For example, footage of large harems or fast swimming fish were photographed for further analysis of the image to gain correct abundance counts and species identification. The three videos were viewed at the same time when possible to remove duplication in counting the same fish as it swum through the different frames. Otherwise, when videos were watched singularly, the time at which the fish were sited entering and leaving frames was noted to reduce the potential for duplicate counts. Fish were counted when they initially entered the frame of the video, or when they were already in frame. Damselfish that remained in view of the video in their territory were only counted the first time they appeared from within the reef. Species identification of parrotfish was completed using still images when they were less conspicuous. Juveniles that were seen in larger schools or harems were not identified to species level. The images of the fish were compared to multiple sources to ensure correct identification, including photographs of fish I had identified with DNA techniques (Figure 4, Randall *et al.* 2003, Michael 2009, Choat *et al.* 2012, van Lier 2013, FishBase 2015). Fish pectoral fin clippings were taken from a range of parrotfish in Tonga and transported to New Zealand. The DNA was extracted, and fragments amplified through the use of polymerase chain reactions (PCR) of the mitochondrial 16S and COI genes. These DNA fragments were then sequenced to support the visual identification of parrotfish in the field (Full method and results in Appendix 1, Table A1).

Behaviour was identified in the first 5 seconds of watching an individual; swimming was recorded as No Behaviour as it was the baseline behaviour. Behaviours for both taxa included chasing other fish, being chased, and eating. If the fish was swimming initially but a behaviour occurred within the first five seconds then that behaviour was recorded instead of No Behaviour. Harems of parrotfish in this study included those made of different species and varying arrangements of phases. This was due to the imprecise knowledge of male fish with regards to their distance to a group of females to be identified as a harem. Also as mentioned, the issues with identifying each phase correctly means a harem could not be identified with one hundred percent accuracy.

Figure 4. Example from left to right on top is identification of *Chlorurus spilurus* in initial and terminal phases, *Scarus globiceps* (bottom left) and *Scarus schlegeli* (bottom right).

2.3 DATA ANALYSIS

The data from both methods were tested for significant difference in the examined variables, for the pelagic and territorial fish taxa used in this research. This was completed using paired t-tests and test of independence of methods were done using chi square tests in the statistical programme R-studio 3.2.2 (2015, R Core Team, R Foundation for Statistical Computing, packages: car, vegan) and Excel (2013).

Abundances

All abundance data, combing the sampling sites, for both damsels and parrotfish, was initially tested for normality using the Shapiro–Wilk test for normal distribution. The larger the sample size in a Shapiro–Wilk test the more likely you will get a statistically significant result (Shapiro and Wilk 1965). Monte Carlo simulation has found that Shapiro–Wilk has the best power for a given significance, followed closely by Anderson–Darling (Razali *et al*. 2011). If the test results suggested that the data came from a normally distributed population then a paired t- test can be completed. I used the paired t-test, as the dependent variables for this study were from related groups – swim and stand were deployed in the same area at the same time. This test specifically determines if the mean difference between the groups is statistically significant. The dependent variable (count) is also continuous and the independent variable has two categorical pairs (swim/stand).

When the data did not meet the assumptions of normality, nor had greatly different standard deviations in different groups, then transformations were completed (McDonald 2014). This is not abnormal as many biological variables do not initially meet the assumptions of parametric statistical tests: they are not normally distributed, the standard deviations are not homogeneous, or both; particularly for pelagic fish species where aggregations or shoals may be expected. Using a parametric statistical test, such as an ANOVA or linear regression, on such data may give a misleading result. In some cases, transforming the data can make it fit the assumptions better and therefore parametric tests can be completed. I used the common square root and exponential transformations on the parrotfish and damsel fish abundance data. When it was applied I reran the test for normality, and the damsel data then met the assumption for normality, and therefore the paired t-test was completed. In contrast, the parrotfish abundance data was still not significant with a transformation. In this case, a non-parametric Wilcoxon signed-rank test was performed. This test compares two paired groups, also known as a rank sum test or signed rank test (McDonald 2014). Essentially it calculates the difference between these paired groups and assumes that there is a difference between them. It is non parametric, as the data does not meet the normality assumptions, that it has a normal distribution, so is an alternative to a paired t test.

The assumption for the homogeneity of variances was also used as the assumption for normality to help identify whether parametric or non-parametric tests could be completed (McDonald 2014). The homogeneity of variances were tested using a Bartlett test and also a Levene test. Bartletts test

(Bartlett 1937, Snedecor and Cochran 1983) is sensitive to departures from normality, the Levene test is an alternative to this but is less sensitive to departures from normality (Levene 1960).

Behaviour

To test the hypothesis that capturing behaviour is independent of the method of capture, I used a chi-square test. The behaviour data was separated into the frequency of Behaviour and No Behaviour captured on film from the total number of fish seen. A Chi- square goodness of fit was used because it measures how well the observed Behaviour data fitted the expected value distribution if the Behaviour frequencies were independent from the methods. It analyses the categorical data of Behaviour vs No Behaviour and the count for each method.

Harem structure

The harem structure, or frequency of different harem sizes, is discrete data for which I tested the independence of structure on the method used. Initially, I had eight categories for the harem sizes 3 – 10 individuals, but more than one size category had frequencies less than five, excluding the use of a Chi-square test, and with eight categories the alternative Fishers exact test (for smaller samples) was also not acceptable. Given these barriers to analysis, I pooled the sizes into four categories 3 to 4, 5 to 6, 7 to 8, 9 and above; the Chi-squared test was then appropriate.

Species richness

Species richness data was tested for a difference between methods, with the dependent variable being the count of species and the independent variable being the method used (swim and stand). The data was initially tested for the assumptions of normality and homogeneity of variances, using the same methods mentioned above. When these were not seen even with transformations applied, then the data was deemed non-parametric. I then used a test equivalent to a paired t-test, the Wilcoxon signed rank test, as the dependent variables are from related groups.

2.4 DATA RESULTS

There was no difference between the two methods for frequency of behaviour captured or abundance counts for damselfish. However, for parrotfish abundance counts, species richness, and harem size structure the difference between the methods was identified as not being due to chance.

Abundances

The parrotfish abundance was found to be significant under the Shapiro–Wilk test for normality, both with and without transformations applied (Appendix 1). However, normality was supported with a plot of the residuals and quantiles being very close to normal (Figure 1A). In contrast, only square root transformation addressed the assumption of homogeneity of variances. The Bartlett test was significant without transformation (p = 5.553-05) and also when transformed exponentially (p = 2.2e-0.16), but when using the square root transformation it was not significant (p = 0.3983). This means that there is homogeneity of variances and, therefore, this assumption was met. This was supported by the Levene test for homogeneity of variances being not significant (p = 0.067). Statistical tests are more robust to deviations from normality, so square root and non-parametric tests were thought to be more appropriate. The Wilcoxon signed-rank non parametric test was not significant (p = 0.102), which means the null hypotheses cannot be rejected, there was no significant difference in abundance count data between swim and stand methods. Outliers were calculated and removed; however, when the statistics were run there was no difference in the overall outcome of the statistical tests.

The medians of the two methods were similar at approximately 35 (Figure 5), although the stand method had a greater spread of data than the swim method. With the middle 50% of the data points having a greater spread in stand method than the swim method. There are a few outliers pulling on the maximum of the boxes, with a greater number of them and at a greater abundance for the stand method creating this spread.

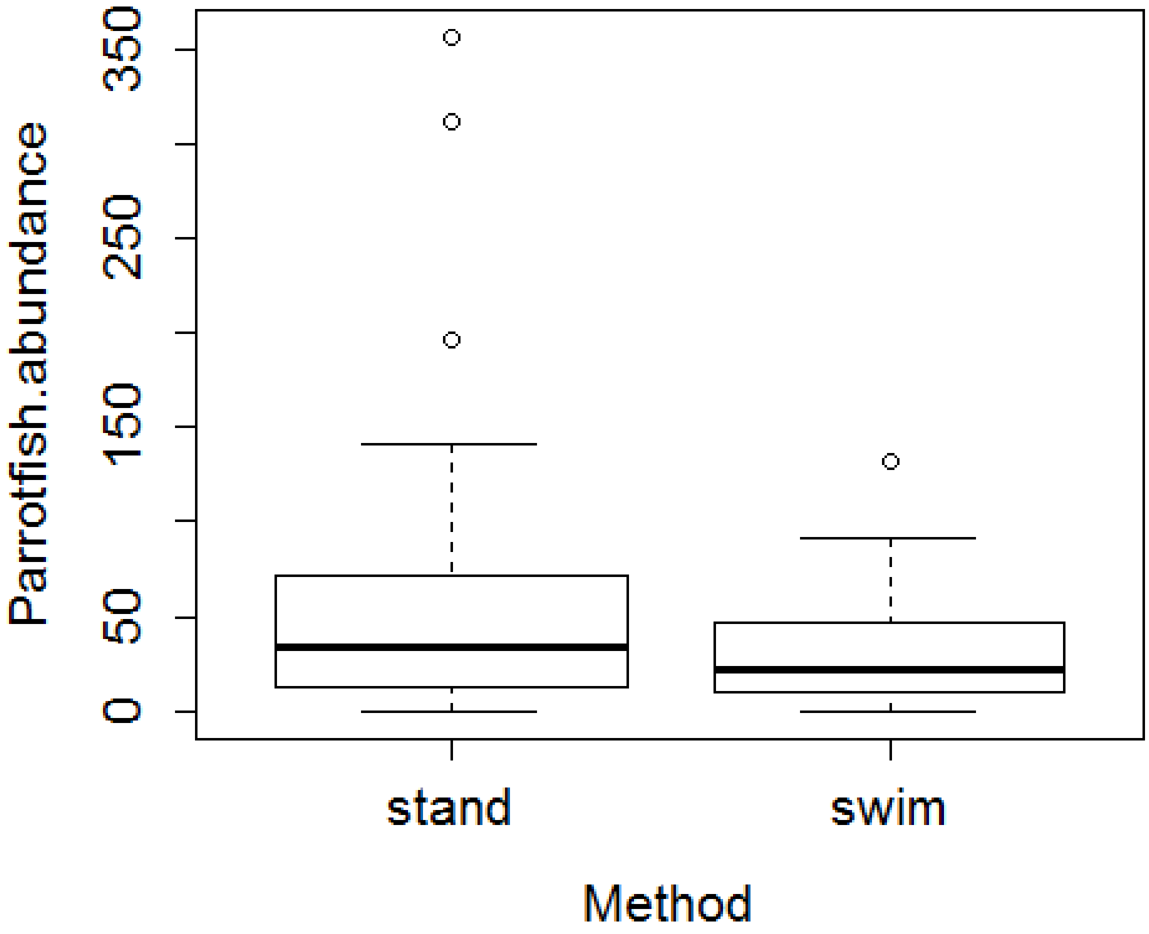

Figure 5. Box-whisker plot of parrotfish abundances captured on video showing medians and spread for the stand and swim methods deployed in Tongatapu in May 2015.

For the damselfish the median count for the swim methods was greater than the stand method, and also had a wider range between the upper and lower quartile (Figure 6).

A paired t-test on damselfish abundance data with square root transformation (Appendix 1) showed no significant difference between the methods of sampling (t-stat = -0.251, df=31, p= 0.803), however, when the outliers were removed the swim method recorded significantly more fish than the stand method (t=5.109, df=31, p=1.6E-05).

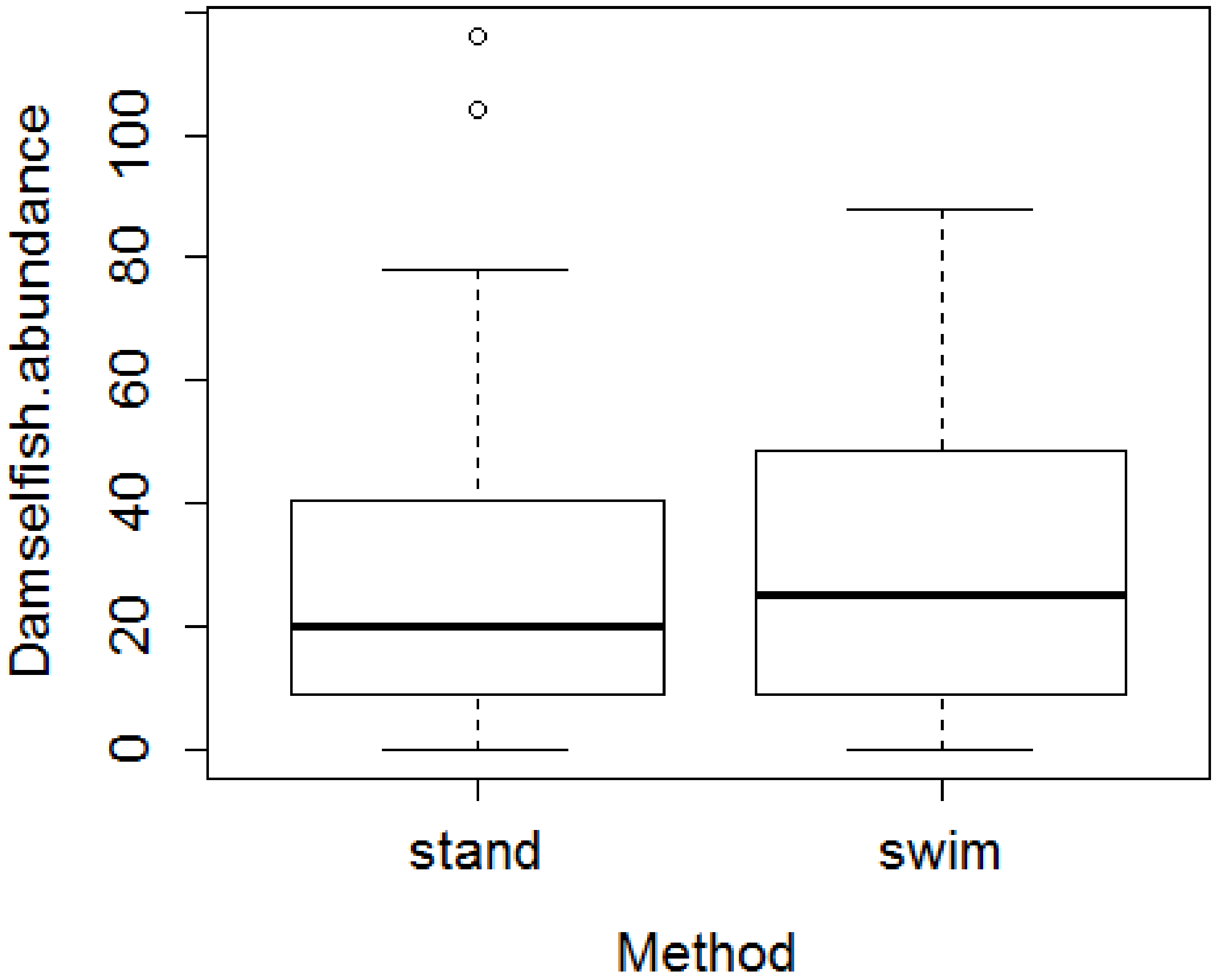

Figure 6. Box-whisker plot of damselfish abundances captured on video showing medians and spread for the stand and swim methods deployed in Tongatapu in May 2015.

Behaviour

The behaviour data was separated into the frequency of Behaviour captured on film versus No behaviour captured from the total number of fish seen. A Chi-square test for the parrotfish (pelagic) showed that the Behaviours captured were independent of the methods used (χ-stat= 2.432, df= 1, p= 0.118). The spread for both methods appears to be influenced by outliers beyond the upper quartiles (Figure 7). The middle 50% of the data spread is shown to have a wider spread in the stand method, with also a greater median than the swim method (Figure 7).

In contrast, a Chi-square test showed that the frequency in capture of damsel Behaviour was dependent on the method used (χ-stat= 92.623 df= 1, p= 6.223E-22). For the damsel fish, the data spread of the stand method was greater than the swim method data, while both methods were pulled by one outlier (Figure 8). The median was higher in the stand method than the swim method (Figure 8).

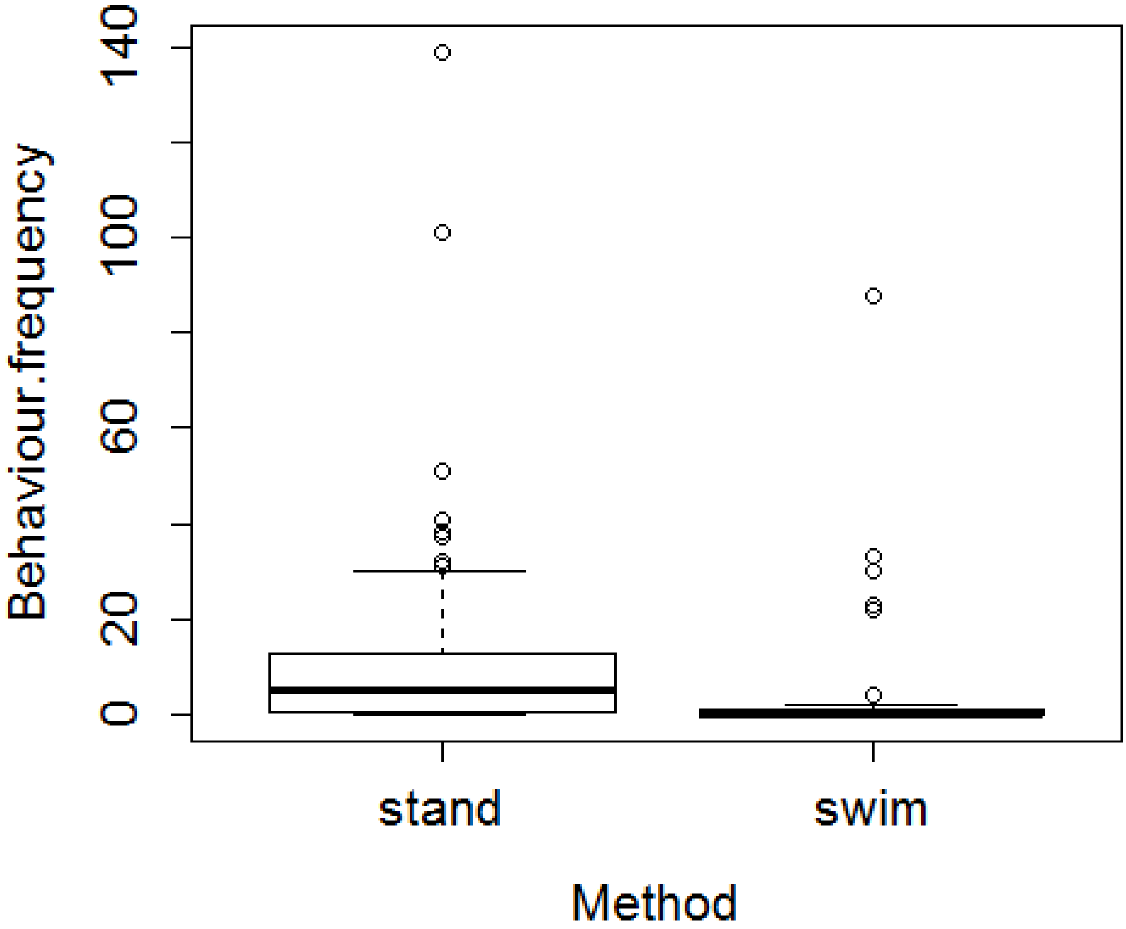

Figure 7. Box-whisker plot of parrotfish behaviour frequency captured on video showing medians and spread for the stand and swim methods deployed in Tongatapu in May 2015.

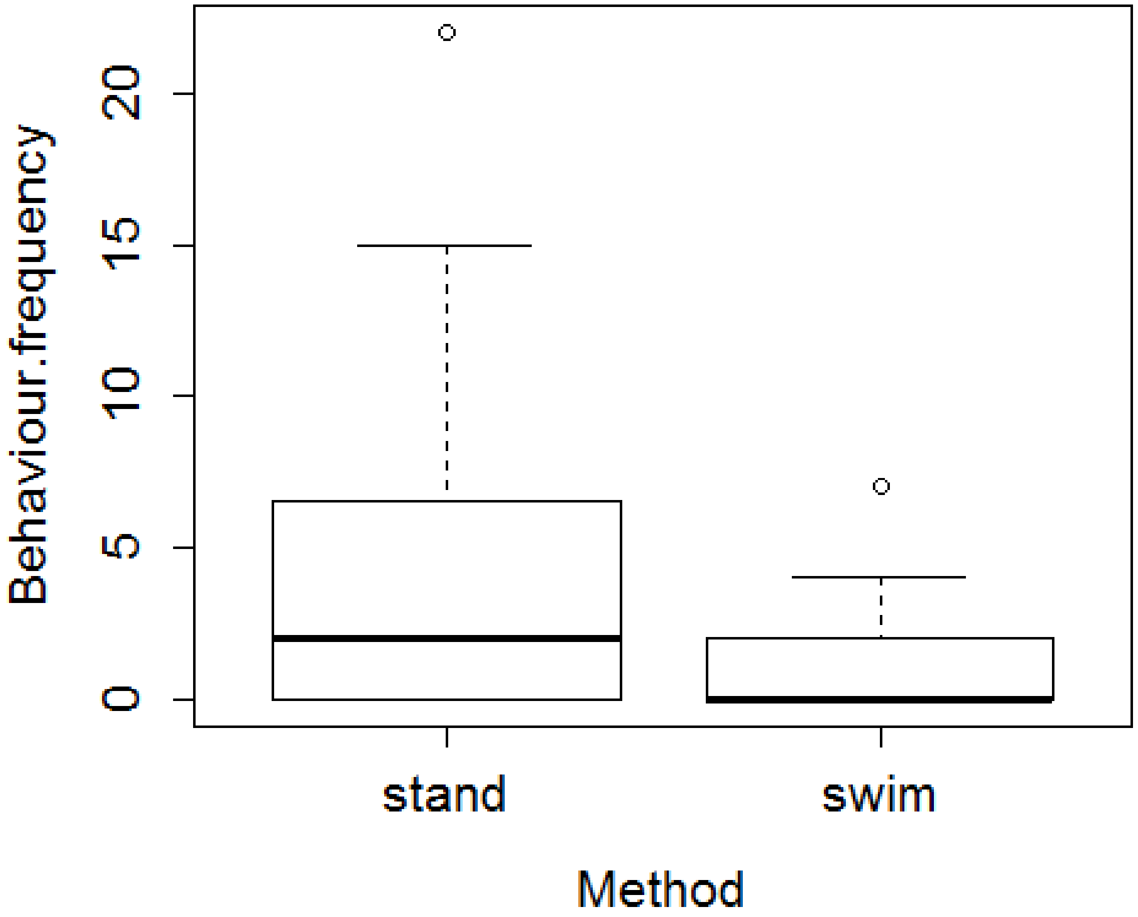

Figure 8. Box-whisker plot of damselfish behaviour frequency captured on video showing medians and spread for the stand and swim methods deployed in Tongatapu in May 2015.

Harem structure

The Chi-square test run on four size categories (Figure 9) shows that the likelihood of observing different harem structures was dependent on the method used (χ-stat=15.905, df=3, p= 0.001). This difference in harem structure and size classes between the stand and swim method was clear, with the stand method across all size categories having a higher frequency then the swim method (Figure 9). This was especially prevalent in the smaller size categories 3 to 4 and 5 to 6.

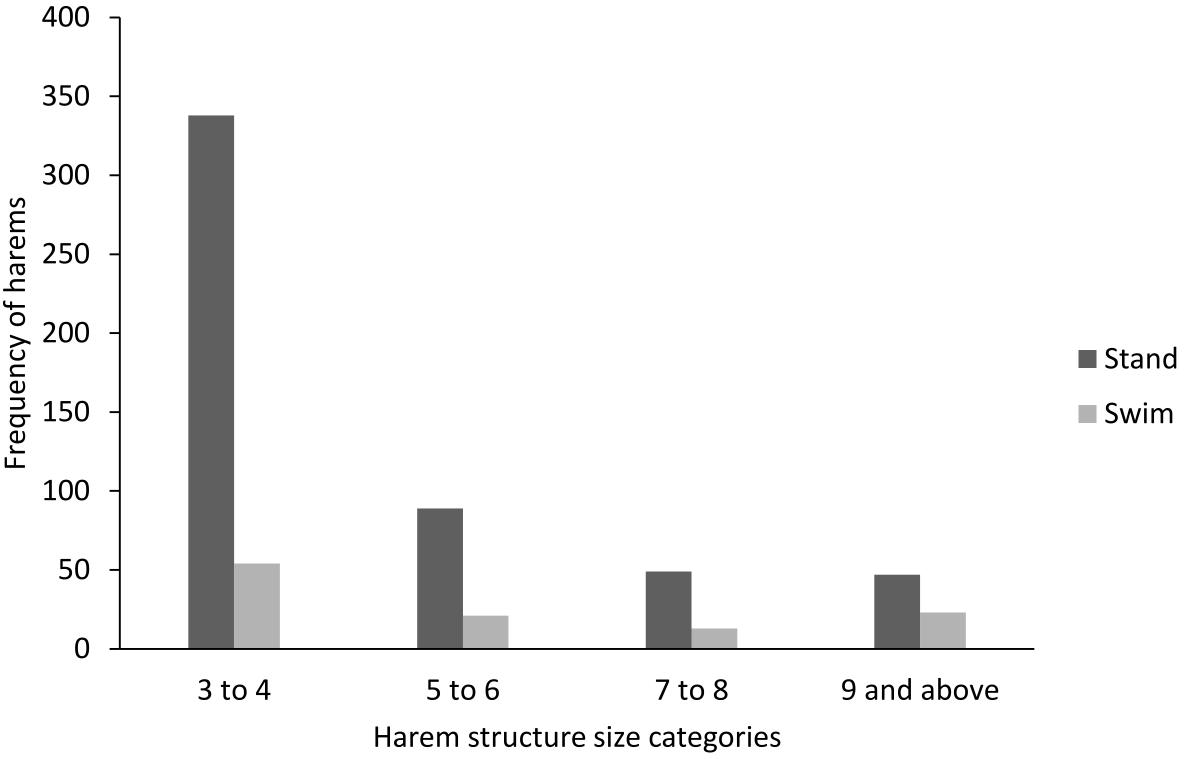

Figure 9. Comparison of stand and swim methods for the frequency of harem size categories.

Species richness

The Shapiro-Wilk tests for normality were all less than the significance value, even when transformations were applied, therefore, this data set did not meet the assumptions of normality. The Bartlett test showed that both with and without transformations the data set was less than the significance value and therefore did not meet the assumptions of homogeneity of variances. The Levene test was also completed with the same outcome until it was exponentially transformed (p = 0.108), meeting the assumption. However, not meeting both of these assumptions means that this data set is not parametric. The non-parametric test using the Wilcoxon signed rank test was significant, (p=0.0001), suggesting the larger number of fish species observed with the stand method than the swim method was not observed by chance (Figure 10).

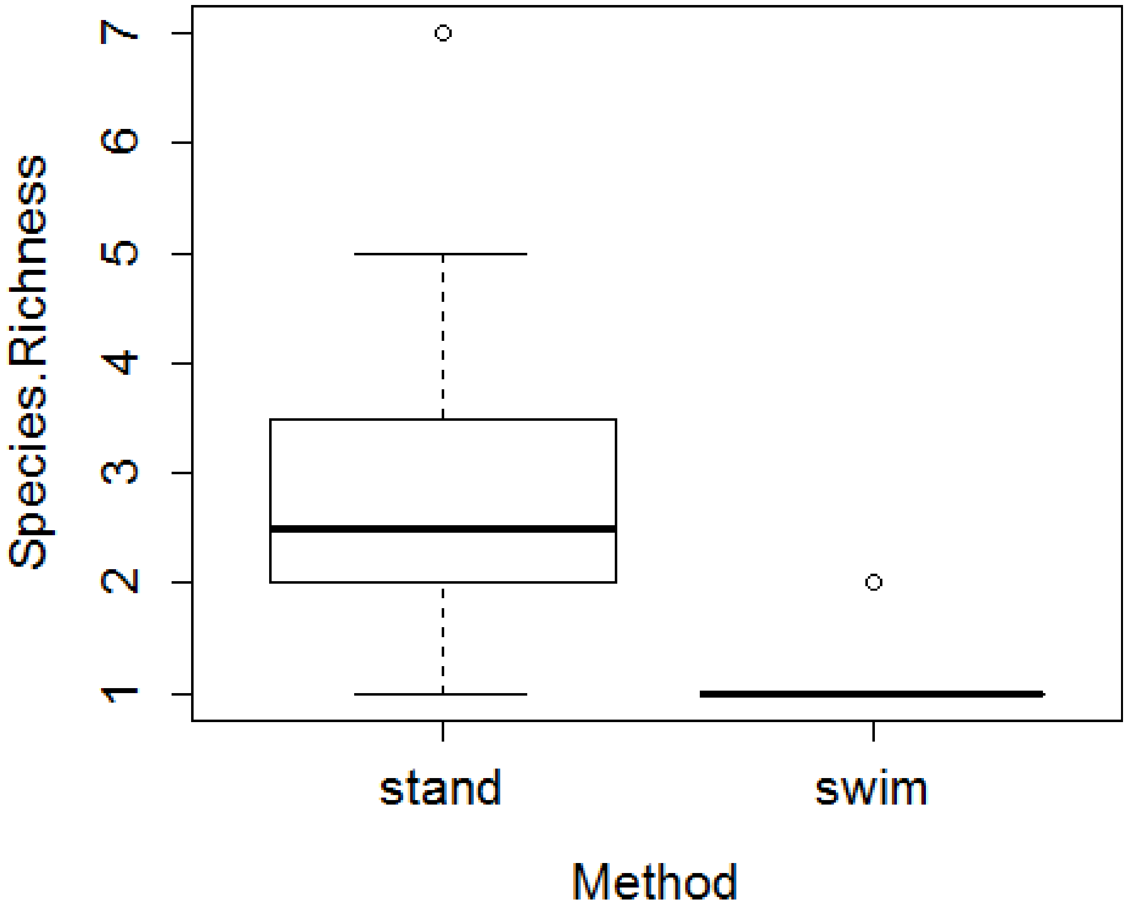

Figure 10. Box-whisker plot of parrotfish species richness captured on video showing medians and spread for the stand and swim methods deployed in Tongatapu in May 2015.

2.5 DISCUSSION

The results of this study suggest that the choice of underwater videoing methods should be question-based and group specific. Damselfish and parrotfish represent pelagic and territorial reef fish groups, and the methods performed differently for these groups. Similarly, the camera stand method was comparable to the swim method for measuring abundance and capturing behaviour of parrotfish, but the swim method appears to be more discernible when capturing harem sizes. The abundance of pelagic fish and territorial fish, with the examples of parrotfish and damselfish used here, showed no significant difference in their count between the swim and stand method, incorporating five different reef habitats studied. However, the distribution of pelagic and territorial fish differs across a reef and is likely to impact the abundance and variable data collection from across the reef. The parrotfish group comprises fish species that have a home range area, but also roam across areas of the reef to find resources such as food, shelter and mating partners (Streelman *et al*. 2002). Their distribution is therefore patchier across the reef than a territorial fish such as the damselfish. The damselfish, as a territorial fish, has a more homogenous distribution in comparison, as they protect a territory of specific distances from conspecifics (Booth 1991). Unless resources are unevenly distributed they will generally be homogenously spread across the reef (Booth 1991). The territorial fish are less likely to come from outside of the area of view into the vision of these 360^{o} view cameras as they hold their territory. In contrast, the swim method allows more area to be covered, but may cause greater disturbance resulting in gaps along the transect due to fleeing and hiding of territorial fish as researchers swim over and through their territories. The pelagic fish are potentially more likely to be observed swimming through the area with the stand method as the disturbance from deploying the stands is brief. Dickens *et al.* (2011) found that reef fish were displaying less defensive, natural behaviour once the stand had been placed and people had moved away.

The ability to capture behaviours for the pelagic fish (parrotfish) was found to be independent of the type of method used. However, behaviours were more frequent with the use of the stand method. The behaviours recorded for these fish were eating and being chased by other pelagic fish and by territorial fish and separating these behaviours may be worthwhile in the future. Behaviours are reactive to many UVC methods and have an effect on abundance also (Stoner *et al.* 2008); for example, with reaction distances of reef fish from divers approaching them (Lindfield *et al.* 2014). The territorial fish (damselfish) behaviour frequency was found to be dependent on the method used. The behaviours of this species were mainly chasing other fish from their territory or being chased. The stand method collected a greater frequency of behaviours, and likely similar to the parrotfish, these fish showed these more natural territorial behaviours with this method over the swim method. When territorial fish emerged from their reef shelter after the camera was set up, chasing behaviour soon occurred. Bearing in mind that these fish will continue to protect that same territory throughout the stand video at that point, therefore, the viewer must be careful or take caution not to count the same fish multiple times, which will influence the count of all the variables, not just behaviours seen. In comparison, when swimming, the damselfish retreat back into their space in their territory out of the view of the camera, and with no behaviour being captured. This would also affect the abundance count for the territorial species using this method.

Harem structure frequencies were also found to be dependent on the method used, with the stand method capturing higher frequency of all harem structure size categories. In particular, the smaller harem size categories such as 3-4, and 5-6 were captured more on the stand videos. These results strongly suggest that the definition of a harem needs to be confirmed or explained in each study. The parrotfish harems seen in these videos were smaller in size in general; potentially due to the classification of a harem used here, which ranged from three individuals to ten and above. The definition needs to include the number of species counted in a harem, the interaction of a male, and the distance of the male in relation to the other fish for example.

Species richness of the parrotfish was generally highest from the data collected from the stand method over the swim method. Across all sites the stand method collected more species then the swim method, with varying degrees of difference. A wide range of species were seen, with *Chlorurus spilurus* being the most common species seen at every site with both methods, and also being a common species seen at the local fish markets (Lovell and Palaki 2000). Capturing species richness between the two methods is likely effected by the species individual response to people and disturbances; for example, those fished at a greater pressure may have a lower threshold to disturbances (Lindfield *et al.* 2014). In the Kingdom of Tonga, a number of the species identified are regularly caught for the local fishery by spear fisherman or divers (Lovell and Palaki 2000). The swim method is similar to a spear fisherman and may present a greater threat to the fish, thereby causing those particularly sensitive species to not be seen in the videos. In comparison, the stand method allows the return of some of those sensitive species, and the richness we observed may be reflective of species specific fishing pressures. Juveniles made up the majority of the fish in the harems, but were not identified to the species level because of the difficulty in distinguishing them at this level (Bellwood and Choat 1989). Species richness may also be affected, however, an assumption has been made that the species we see as adults are likely to be the species seen as juveniles also. With future research these could be reassessed and the juveniles identified at the species level to further identify the full species richness collected from these methods across these sites.

Transect vs Point count

Stationary point count technique allows a more careful census of an area, and minimizes the behavioural disturbance from the divers themselves, although the area surveyed is limited to a single point (Colvocoresses and Acosta 2007). In comparison, the transect surveys can cover a larger area per sample, but the behavioural disturbance comes from the moving diver or snorkeler. Less time spent over a particular area gives greater opportunities for fish to hide from the divers view and field of vision. In addition, fish that are attracted to divers or are not disturbed by divers have more chance to be seen in view, or to enter the survey area, in a point count census (Bohnsack and Bannerot 1986, Colvocoresses and Acosta 2007). Colvocoresses and Acosta (2007) used SCUBA to conduct transect survey and stationary point counts at 26 sites each month from May to September of 1999 and April to December of 2000 in the Florida Keys. They concluded that these two different methods required a similar amount of diver effort to complete, but that the transect survey collected approximately 1.5 times more individuals (Colvocoresses and Acosta 2007). However, since transect surveys encompassed 4.3 times as much area as point counts, the overall observed densities averaged approximately three times higher for the point count samples (Colvocoresses and Acosta 2007). Comparisons of the strip transects and stationary point counts found that they are equally effective UVC methods for certain species (Samoilys and Carlos 2000). Of course, the species being researched, their mobility and behaviour (Samoilys and Carlos 2000) must be considered when interpreting the results. The higher and, presumably more accurate, density estimates provided by the point count method, along with other factors such as presumed increases in accuracy of counts and size estimates associated with increased observation time of individual fishes (Colvocoresses and Acosta 2007, van Lier 2013), led me to select this method.

Method biases and comparisons

The advantages and disadvantages of the UVC method have been summarized in several papers (Harmelin-Vivien *et al.* 1985, Harmelin-Vivien and Francour 1992, Cappo and Brown 1996, Samoilys 1997, Willis *et al*. 2000, Watson *et al*. 2005). For example, some cryptic or shy species are not accurately observed because they avoid the presence of the divers doing the observing (Watson *et al.* 1995, Kulbicki 1998, Stewart and Beukers 2000, Willis and Babcock 2000, Willis 2001, Watson *et al.* 2005, Dickens *et al.* 2011, Pelletier *et al.* 2011, Lindfield *et al.* 2014). Although De Girolamo and Mazzoldi (2001) found no relationship between the presence of divers and fish abundances, in some cases, it has been shown that fish change their behaviour and approach (Chapman *et al*. 1974, Cole 1994) or avoid divers (Brock 1954, Pita *et al.* 2014). Plasticity in the behavioural response of large reef fishes to SCUBA divers means that commonly used UVC techniques do not always provide reliable estimates of relative density (Willis and Babcock 2000). Most of the reef species targeted by fishing are mobile and have extensive home ranges, and sometimes shy behaviour in the presence of divers. Avoidance from divers tends to be more pronounced in areas with greater fishing pressure (Lindfield *et al.* 2014).

Comparisons among abundance estimates obtained by different techniques need to be made with particular care, because each technique has different biases that can affect the results (Guidetti *et al*. 2005). In this regard, active and diurnal species are, in general, adequately censused by divers. Diver UVC obtained more complete and reliable fish inventories (Pita *et al.* 2014). At low speeds, mobile species tend to be overestimated, whereas at faster speeds small cryptic fish tend to be underestimated (Valle and Bayle-Sempere 2009, Pita *et al.* 2014). However, it has been shown that there is a strong tendency by divers to underestimate nocturnal, cryptic, small, abundant and mobile species (Christensen and Winterbottom 1981, Brock 1982, DeMartini and Roberts 1982, Jennings and Polunin 1995, Ackerman and Bellwood 2000, Willis 2001, Blyth-Skyrme *et al*. 2006). Comparing ROV, RUV and UVC, it was found that divers were more precise, were faster and more reliable in estimating the abundances of the more mobile, less abundant, less frequent, cryptic and smaller fish (Pita *et al.* 2014). In comparison RUV had the poorest abundance estimates, and video cost was greater than divers. Although this may vary with weather, conditions, species and the type of study being completed (Pita *et al.* 2014).

While UVCs are generally quick, inexpensive, non-destructive, and adaptable, a growing body of literature documents the limitations of their use (Samoilys and Carlos 2000, Gilbert *et al.* 2005, Colvocoresses and Acosta 2007, Murphy and Jenkins 2010, Green *et al.* 2013). Diver observation differences can also have an effect on UVC diver technique (Thompson and Mapstone 1997, Edgar and Barrett 1999, Blyth-Skyrme *et al*. 2006, Côté and Perrow 2006, Pita *et al.* 2014). For example, when divers are swimming they may have an inclination to follow certain species or only identify the bright species. Biases in underwater surveys have been shown to come from a variety of factors, including survey time and area, water visibility, fish movement, fish behaviour, number of species censused, the level of identification and observer variability, to identify a few (Colvocoresses and Acosta 2007). These can vary between methods and also species (Lincoln Smith 1989, Watson *et al.* 1995, Sale 1997, Samoilys and Carlos 2000, Willis *et al.* 2000, Williams *et al*. 2006).

Biases with BRUV is that individuals who are hungry may defend the food from others, also potential predators could prevent the target species from being caught on video (Borard *et al*. 2014). It is important to also understand the behaviour of the target species, such as when they are most active, for example, if they are nocturnal (Unsworth *et al*. 2014), and to recognise that cryptic species may not be identified with this method unless the camera are orientated correctly (Unsworth *et al*. 2014). Understanding the biases of different abundance estimation methods is essential to management and conservation (Erős *et al.* 2009). Biases due to altered fish behaviour are assumed not to be an issue for fish surveys if the biases are consistent across the spatial scales investigated (Lindfield *et al.* 2014). To minimise biases in the UVC method procedures for deploying visual counts were standardised wherever possible in this study, as suggested by Samoilys and Carlos (2000).

Manual UVC can be an effective and inexpensive method for collecting independent replicates, however, another key limitation of manual UVC has been logistical safety for human observers (e.g., diving times/depths, water temperatures and currents; Sale and Douglas 1981, Pelletier *et al.* 2011, Pink and Fulton 2015). Numerous studies have emphasized the shortcomings of common underwater visual survey methods for the detection of cryptic fishes and advocate instead the use of modified methods that account for species characteristics and behaviour (Kulbicki 1998, Ackerman and Bellwood 2000, Samoilys and Carlos 2000, Willis 2001, Gilbert *et al.* 2005, Green *et al*. 2013). The UVC method generally underestimates the abundance, in particular for small, cryptic and hole-dwelling species (Brock 1982, Ackerman and Bellwood 2000, Willis 2001, Gilbert *et al*. 2005).This may be due to the biases mentioned from the snorkelers and divers: Minimising the effect of divers and the exposure to fish is essential (Dickens *et al.* 2011). Fixed distance transects using tapes or lines deployed by a second diver would appear to maximise fish counts and minimise diver effects (Dickens *et al.* 2011). The effect of the diver on the fish is time dependent on how long the diver spends disrupting them, and many studies use a 5 minute recovery period before starting a count (Dickens *et al*. 2011). This is one way to potentially decrease the diver effect, another may be to use video techniques.

Images and video aid study, in that they can be reused and also analysed for other research aims (Shobha 2014, Hui *et al.* 2015). One negative of video is that manually analysing underwater videos requires a lot of time and human concentration (Spampinato *et al*. 2008, Boom *et al.* 2014). For a one minute video it can take a person up to 15 minutes for classification and annotation (Spampinato *et al*. 2008). In this sense manual observation has a clear advantage over video UVC, as it provides an effective means of collecting a large data set of independent observations across multiple species and activities in a relatively short time period with less time out of the field needed (Pink and Fulton 2015).

Continued development of non-intrusive technology for marine monitoring studies is needed (Zeller and Russ 2000, Murphy and Jenkins 2010, Giddens *et al*. 2014, Pita *et al.* 2014). Models for measuring population dynamics and variables of reef fish are continually being devised, adjusted, modified and applied. Wielgus *et al.* (2007) created a method for estimating population growth rates with data that is collected for reef fish, with the use of grouper data, using demographic estimates. A simulation model was created by Ward-Paige *et al.* (2010) to evaluate counts obtained by divers deploying non-instantaneous belt-transect and stationary point count techniques, this included fish speed, survey procedure that affect fish count. They found caution is needed when describing abundance, biomass, and community structure based on non-instantaneous UVC, especially for highly mobile species (Ward-Paige *et al.* 2010). This model was written in R, showing mobile fish were over counted in non-instantaneous UVC as the effect of entering the survey area after the survey starts, relative bias increases with fish speed (Ward-Paige *et al*. 2010). The problem with overestimates is that it can lead to larger quotas causing overexploitation as well as preventing adequate protection status (Léopold *et al.* 2009, Ward-Paige *et al.* 2010).

There is no way to account for all biases, there is no best method, and standardizing estimates should occur so that as UVC increases in use it can be comparable in a database (Kulbicki *et al.* 2010). The main finding from Murphy and Jenkins (2010) was that a combination of observational techniques, rather than a single method, was the most effective approach to marine spatial monitoring this is supported by Karnauskas and Babcock (2012). Combining observational methods has proved to be an effective means of reducing inherent biases in each technique and increasing the range and detail of data obtained from spatial surveys of management areas (Zeller and Russ 1998, Willis *et al.* 2000, Watson *et al.* 2005, Godoy *et al.* 2006). By using a combination of observational techniques to target specific species or habitats, spatial monitoring surveys can provide information on different aspects of the ecosystem (Bennett et al. 2009, Murphy and Jenkins 2010). When assessing the suitability of either method, availability of time, funding, and relevant expertise should be primary considerations (Holmes *et al*. 2013).

2.6 CONCLUSION

In the case of this study, for measuring or estimating pelagic and territorial fish variables, the swim and stand methods did not overall have significant differences in the data recorded. It then comes down to the benefits from each method. The swim method has some biases in that the movement as the video captured can be more erratic from the person swimming, currents and weather. The novel stand method helped to remove the biases of having the person in the water and allowed for the fish to return to what we could assume as more natural behaviour. As with many studies it appears the use of more than one method would be more beneficial, so using the UVC method along with support from a visual census completed at the time would help to balance out those biases. This review and research has been useful in gathering more information on the effectiveness of UVC methods and in particular the novel method of using a 360° videoing stand. Both methods provide benefits and biases towards the data collected for pelagic and territorial fish. With further application and alterations these methods may be used for further research and monitoring of the reef and its fish inhabitants.

3 CHAPTER 3 – PARROTFISH POPULATION DYNAMICS

3.1 INTRODUCTION

Protogyny in parrotfish

In hermaphroditic fish, exogenous factors such as the gender composition of the harem have been identified to trigger an individual to change sex (Warner *et al.* 1975, Robertson and Warner 1978, Warner and Hoffman 1980, Warner 1982, Warner 1988). However, this seems naïve given the complexity of environmental factors that may contribute to sex change (Taylor 2014). The different stages in maturation and sex transitions in parrotfish species appear as differences in sizes and colourations (De Girolamo *et al*. 1999, Kume *et al.* 2010); therefore, transitions between these stages are difficult to ascertain, and the triggers and processes driving these changes are even more challenging to determine. In addition, protogynous sex reversal (male back to female) has been seen in the rusty parrotfish *Scarus ferrugineus* (Abdel-Aziz *et al.* 2012). However, sex change is seemingly irreversible for many protogynous parrotfish species due to changes in gonads anatomy and function, and fish colouration and behaviour (Kusen and Nakazono 1991, Chang *et al.* 1997, Godwin *et al.* 2000).

In general, terminal phase parrotfish, which are exclusively male, primarily display a blue or green base with varying bright colourations overlaid (Michael 2009, Ali *et al*. 2011). The initial phase includes females and primary males, with duller monochromatic colouration. For example, the initial phase of *Chlorurus sordidus* is a brown/red colour with a clear white spot on its tail (Crook 1997, Crook 1999). The juvenile phase of *Chlorurus sordidus* have been identified as displaying physiological colour patterns (PCPs), these are non-permanent patterns that occur from fast shifting pigments in the chromatophores (Odiorne 1957, Crook 1997). The PCPs have different displays including uniformly dark, striped and bullseye (Crook 1997). It must be noted that *Chlorurus sordidus* has been identified as another species in the Pacific, now known as *Chlorurus spilurus* (Choat *et al.* 2012). Munoz and Warner (2003) completed experiments in patch reefs of St. Croix, U.S. Virgin Islands, whereby males were removed from the reefs and the behaviour of the remaining fish was observed. They noted a change of colour in 7 out of 22 females or initial phase fish from harems, followed by the development of male behaviour that was fully expressed approximately 20 days after removal (Munoz and Warner 2003). Behavioural sex change therefore does not precede morphological sex change (Munoz and Warner 2003). However, given the potential for colouration to misrepresent the true gender and developmental stage, species gonadal histology and gonadosomatic index is often completed in conjunction with reef observations (De Girolamo *et al.* 1999, Kume *et al.* 2010). A visual census, completed by De Girolamo *et al.* (1999), used the knowledge of species identification and colour differentiation among females and males at different stages to calculate the sex ratio of adult *Sparisoma cretense*. This study supported the sexual pattern of protogynous sex change, also similar social structure and behaviour of other scarids, such as *Scarus iserti* (Robertson and Warner 1978). Gonadal histology was completed to support the colour identification (De Girolamo *et al.* 1999). This replacement of males may not only occur due to sex change, another bachelor male can also replace the harem vacancy (Munoz and Warner 2003).

Social structure of parrotfish

Protogyny commonly results in a haremic social structure when one large male gathers a group of females. However, the spatial extent of harems and the male territory are unknown. Therefore, the definition of a harem is not concrete and many questions remain unanswered. For instance, what is the standard distance required between the male and females to prevent females transitioning to dominant males? *Sparisoma viride* (van Rooij *et al.* 1996) and *Sparisoma aurofrenatum* (Munoz and Motta 2000) both have limited ranges proposed to arise from the partitioning of the reefs between harems (Welsh and Bellwood 2012). In this case, a dominant male will usually exclude other individuals from within their territory and maintain almost exclusive utilisation of resources for the harem (Bruggemann *et al.* 1994, Welsh and Bellwood 2012).

When the abundances of competitive species are high enough, territoriality, or haremic behaviour, is suggested to no longer be favourable and schooling social structure may become more favourable, where the male still leads, but is more nomadic (Bonaldo *et al.* 2006, Welsh and Bellwood 2012). This has been seen in *Sparisoma* species in Fernando de Noronha Archipelago, tropical West Atlantic, where initial phases aggregate more frequently than terminal phases, therefore, phase is an important factor altering social structure (Bonaldo *et al*. 2006). Individuals likely maximize fitness by devoting more time to feeding and reproducing in another (non-territorial) fashion (Itzkowitz 1977, Clifton 1989, Kuwamura *et al.* 2009, Welsh and Bellwood 2012). It may be this trade-off, which favours the schooling behaviour, with limited school fidelity, in some species of parrotfish such as *Scarus rivulatus* (Welsh and Bellwood 2012).

Schooling or foraging aggregations, are a social structure similar to harems, except these can be made of juveniles, females, males and of mixed species. Another question then arises in regards to harem definition; whether a school contains a harem, or does the harem become part of the school? Males were seen in schools of only males, and schools containing no males (Personal observation).

School size in *S. rivulatus* has been shown to be highly variable on the Great Barrier Reef, with a mean size of 5.7 individuals (Welsh and Bellwood 2012). Schooling affected their functional role, where feeding rates in school were two times higher for *S. rivulatus* and three times higher for other scarid species (Welsh and Bellwood 2012). All *S. rivulatus* were site attached, roving a limited area of the reef (Welsh and Bellwood 2012). Schools benefit members by reducing predation risk and allowing access to resources through increased foraging opportunities (Robertson *et al.* 1976). Schooling generally occurs in parrotfish juveniles, used for feeding and protection. This has been seen with two mixed species of juveniles from *Chlorurus* and *Scarus*. Shoaling or schooling behaviour is estimated to occur in 10,000 fish species during their lives (Shaw 1978, Ward *et al.* 2012). Some parrotfish form large foraging groups that overwhelm territorial herbivores such as damselfish and surgeonfish, and allow them to reach areas inaccessible when in low numbers; occasionally forming mixed species groups with tangs, surgeonfish and doctorfish (Michael 2009). Wrasse may also swim near groups of parrotfish, as their feeding behaviour can extract cryptic prey species. In the cyprinids with schooling behaviour, for example the chub (*Leuciscus cephalus*) and the European minnow (*Phoxinus phoxinus*) in this study, it was shown that focal fish chose schooling with conspecifics over heterospecifics (Ward *et al.* 2002). Both member size and species have been noted as important characteristics of the structure of the mixed-species foraging groups of *Scarus iserti* (Quinn *et al.* 2012). The number of fish within schools is highly variable as smaller individuals have less benefits when schooling with larger individuals, as they are more competitive and reduce the opportunities for smaller individuals to feed (Pavlov and Kasumyan 2000).

Many parrotfishes co-occur in mixed-species aggregations as juveniles, but diverge in resource use and social structure, such as territoriality over home ranges, as adults (Overholtzer and Motta 1999). Home ranges of adults often overlap, and the proportion of microhabitats present within a home range are generally similar for all species (Overholtzer and Motta 1999). Home range size is found to increase with body size for *S. aurofrenatum*. Diets of all species studied by Overholtzer and Motta (1999) in the Florida Keys were extremely similar, all species fed selectively from the available foods and fed primarily on the calcareous macro algae *Halirneda opuntia.* This diet was maintained even despite the fact it was energetically costly to consume and has low food value. The focal individuals of their study interacted aggressively with conspecifics, other juvenile parrotfishes, damselfishes, and occasionally wrasses, and of these individuals *S. aurofrenatum* and *S. viride* were most aggressive toward conspecifics (Overholtzer and Motta 1999). Both of these *Sparisoma* spp. were chased more often by territorial damselfishes than any other species. Damselfish are highly aggressive and defend territories against egg predators and potential food competitors (Thresher

1976, Tolimieri 1998). Resident damselfish have also been seen to direct significant agnostic behaviour towards stoplight parrotfish recruits (Tolimieri 1998). Benthic processes such as this can alter recruitment patterns from initial settlement patterns. These findings of aggression, both from and towards the parrotfish, support evidence that herbivorous fish do not feed randomly from all potential foods within home ranges (Overholtzer and Motta 1999). The aggressive interactions observed among juvenile parrotfishes are likely affecting their use of resources and may affect their subsequent territoriality as adults (Overholtzer and Motta 1999).

Impacts on parrotfish worldwide

Influencing these important variables, such as behaviour, feeding and social structure of the parrotfish are anthropogenic impacts such as over exploitation, pollution and environmental change. As a group, the parrotfish are historically less exploited then others (Russ and Alcala 1999, Taylor *et al.* 2014). There are many reasons why these fish would be susceptible to harvesting, these include: their conspicuous colouration, shallow distribution, small home ranges and that they are easy targets at night (Taylor *et al.* 2014). A study by Hawkins and Roberts (2003) has provided evidence that protogynous hermaphrodites are particularly sensitive to overfishing and maybe more sensitive to other anthropogenic and natural impacts due to their lower abundance. With continual human population growth and demand for protein, more provisions will be used to capture the larger fish, such as these herbivores on the reefs. Another influence of increased human population is greater pollution, influencing both the health of the fish and health of the coral reef ecosystems (Wilkinson and Brodie 2011). As well as the pressure for sustainable feeding of the population, people also pollute the ocean with chemicals or waste. If there is not sufficient facilities for the growing population raw sewage can be leaked into the ocean and chemicals from urban development (Wilkinson and Brodie 2011). This is especially important in areas where there is low water change and these toxins therefore remain in the water for longer periods. Also the excess nutrients from agricultural activities may increase the growth of algae (Wilkinson and Brodie 2011) and in turn effect the job the parrotfish have, with increasing competition of macro algae against the coral growth.

Tongan anthropogenic influences

Tonga is the size of 699 km squared land area, with a growth rate in 1999 estimated at 0.5% (Morrison 1995, Morrison 1999). The human population of Tonga has increased, as shown by the national census in 2011 when the population increased by 1.3% from 2006 (2011 Census Report:

Analysis), in turn increasing the effect on the environment (Morrison *et al*. 2013, Global Environment Facility 2013). Water quality in the Fanga'uta Lagoon is compounding the problem as excess nutrients and sediments result in eutrophication in coastal waterways, in seagrass beds and on coral reefs (Global Environment Facility 2013). There has also been contamination of the groundwater through the accidental or deliberate release of persistent organic pollutants, such as agricultural pesticides and PCBs, and since the 1980s there has been greater sewage and pollution entering the lagoon (Global Environment Facility 2013). There is high residence time in the lagoon, leading to high potential of eutrophication (Lovell and Palaki 2000). The effects of pollution vary in relationship to their vicinity to the populated areas. Pollution like this comes through groundwater, run off and direct dumping (Morrison and Munro 1999, Global Environment Facility 2013). The storms and rain also flood the lagoon with extra pollution the water picks up, this pollution can come from leaky septic tanks, oils, chemical fertilizers and garbage dumps, such as those around Popua, located just outside of Nuku'alofa (Global Environment Facility 2013). Water quality was found to be worse through the summer months with greater number of storms and rains (Prescott *et al.* 2007).

The connectedness of the land, its influence on the reef and the environments around the reef are important to understand. One affects the other, while both playing important roles for the people, for resource security and for cultural significance. These anthropogenic impacts, such as pollution and developmental changes affecting the environment, on the reef and neighbouring habitats have significant impacts on top of driving pressures directly on the reefs, such as overfishing.

For quality scientific base line data sets its best to have precise experimentation, objective interpretation and temporal monitoring (Maynard *et al*. 2008). Identifying a baseline is the critical first step for conservation or restoration of a coral reef (Bruno *et al*. 2014). In some cases baselines are taken before a marine park or reserve is in place to identify the changes that occur with controls in place (Lam *et al.* 2006). This baseline information helps to provide a case study for understanding the population dynamics and role of environmental and anthropogenic factors on parrotfish (Pakoa *et al.* 2010). As Tonga continues to grow there is need for monitoring of their resources, particularly those that supply a key ecosystem role, cultural role and economic role, such as the parrotfish. Growing populations cause demands for greater living standards increase pressure on the coastal and marine resources (Morrison 1999). The parrotfish are shown to provide an important ecological role in the coral reef ecosystem, while also being a key resource for the growing population of Tonga. Understanding their dynamics, current abundance, species richness and structure will help future management so that their role and use as a resource can be maintained for future generations.

A similar study completed by Clua and Legendre (2008), where they specifically looked at fishing pressure effect on the parrotfish community. They observed a decrease in average fish size and a change in dominance, those larger bodied species were replaced by smaller bodied species that were less harvested. Only two sites, Manuka (also known as Kolonga) and Ha'atafu were sampled by Clua and Legendre (2008), the study completed here can therefore expand on this research and the more localised impacts around sites within the Tongatapu semi-enclosed lagoon. It is important to see if these shifts in the parrotfish communities are also seen in areas with higher anthropogenic impacts from more populated areas, such as Sopu near Nuku'alofa the capital. Not all reefs are fished equally, so to assess the state of the fishery it is important to understand the haremic structure and protogyny dynamics that occur under different fishing pressures.

Aims

The present study sought to expand on previous work and formally establish a baseline data set for parrotfish within the semi-enclosed lagoon of Tongatapu. The two aims were: 1) to create a baseline data set for parrotfish, which will allow for quantitative measurements of ecosystem change in a heavily anthropogenically influenced area and direct further scientific investigation and management options; and 2) to investigate site-specific differences in species richness, abundance and social structure, and factors that may be influencing these patterns. Both of these will be examined spatially and temporally.

3.2 METHODS

Site description

The sample locations on the island of Tongatapu are the same as described in the previous Methodology chapter (Figure 11). The Sopu site is located near the capital and the most populated area of Nuku'alofa. Due to its location this site has a high amount of anthropogenic pressure through pollution, fishing and physical alterations (Morrison *et al*. 2003, Clua and Legendre 2008). On average, this site was 420 metres offshore from the mainland, 1 kilometre from the nearest offshore reefs, and 3.3 kilometres from the closest onshore reef site of Tukutonga. The Tukutonga site is located further from the capital with potentially fewer anthropogenic effects, however, pollution is potentially greater due to the proximity to a poorly maintained landfill (Chesher 1985, Morrison *et al.* 2003, Friedman *et al.* 2009). It is approximately 868 metres from shore, and the closest offshore reef system is approximately 1.16 kilometres away. Another sampling location on the mainland was Kolonga, which is located near several villages and experiences medium-high fishing pressure and moderate levels of pollution from these village populations (Morrison *et al.* 2003, Clua and Legendre 2008). This site has greater variability in its coral cover then the other two mainland sites, with some areas being dominated by macro algae and also a range of depths. The distance from shore is approximately 470 metres, with nearby patch reefs being on average 380 metres away.

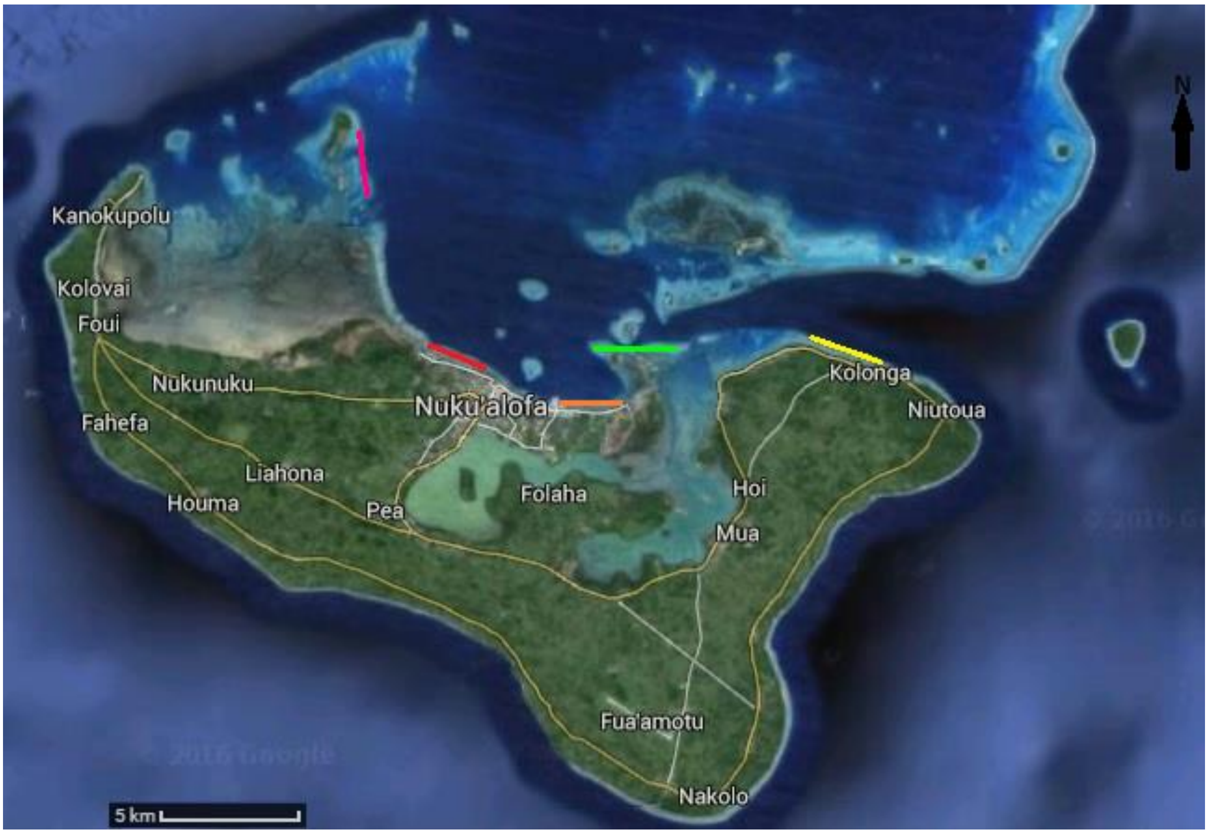

Figure 11. The Island of Tongatapu, in the Kingdom of Tonga (Google Earth 2016). Sites and stations within the Tongatapu semi-enclosed lagoon, each circle representing a station within a site. Pink – Atata, Red – Sopu, Orange – Tukutonga, Green – Offshore, Yellow – Kolonga.

The offshore island sampling sites were located around the islands of Pangaimotu and Manima on the east of the lagoon, where the closest points of these islands were approximately 800 metres from the mainland. These sites have lesser anthropogenic impact as it has a lower populous, however, Pangaimotu is an island that is popular with tourists. Therefore, there still remains some anthropogenic effects, through the physical alterations for this tourism, pollution and some fishing pressure. Another sampling site location was off the main island around the island of Atata, to the west of the lagoon. The closest point of Atata reef to the mainland was approximately 900 metres, and 600 metres to nearby islands. There is only a small population on the island; therefore, the anthropogenic pressures experienced are quite low. The reefs themselves were located approximately 100 metres from the island and are surrounded by deeper waters with more changeable bathymetry. Atata Island is surrounded by a Special Management Area (SMA), as set up under the Fisheries Management Act (2002; FMA, FFF & SPC 2010). The sampling at Atata 2014 occurred on either side of the island, where the west side is a Fish Habitat Reserve (FHR) (Figure 12).

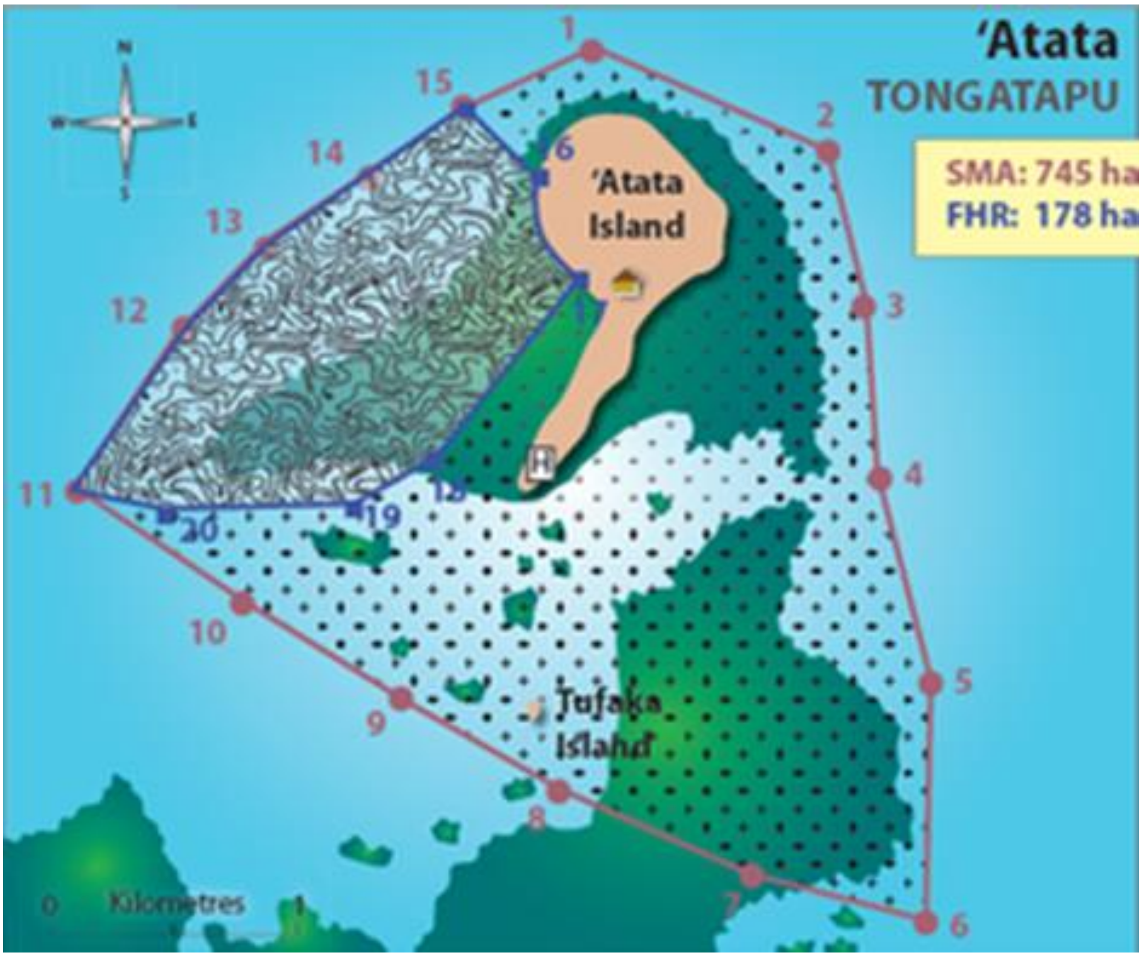

Figure 12. Outline of the SMA and FHR around Atata Island within the semi-enclosed lagoon of Tongatapu (FMA, FFF & SPC 2010).

Sampling

Sampling occurred in 2012 (van Lier 2013), November-December 2014 and April-June 2015 in daylight hours, sampling 2-3 hours around the high tide. The method used for all three sampling years followed the set-stand method described in Chapter two. Data was collected in 2012 by an Honours student van Lier (2013) and the other two sampling years were completed by myself. Three of the five sites sampled in 2012 were used for comparison with my data: Sopu, Atata and Manuka. The sites identified as Manuka and Kolonga are in the similar area and therefore are interchangeable, but from now on will be mentioned as Kolonga. Two stations were sampled per site where possible. The 2014 data was collected at four sites; Sopu, Atata, Kolonga and Tukutonga, all with two stations sampled per site where possible. The 2015 data set comprised five sites; Sopu, Atata, Kolonga, Tukutonga and Offshore, and each site comprised four stations where possible.

Each of these stations included three habitats: crest, back and edge. Sixty metre transects were placed horizontal to the shore in each strata, along which three points were videoed in 2014 and 2015, where only two were videoed from the 2012 data. All video footage was viewed using AVS Video Editor. The video footage was viewed using AVS Video Editor. All parrotfish within approximately 3 metres were then identified and counted and the behaviour recorded. The footage was stopped where necessary for a still capture. For example, footage of large harems or fast swimming fish were photographed for further analysis of the image to gain correct abundance counts and species identification. The three videos were viewed at the same time when possible to remove duplication in counting the same fish as it swum through the different frames. Otherwise, when videos were watched singularly, the time at which the fish were sited entering and leaving frames was noted to reduce the potential for duplicate counts. Fish were counted when they initially entered the frame of the video, or when they were already in frame. Species identification of parrotfish was completed using still images when they were less conspicuous. Juveniles that were seen in larger schools or harems were not identified to species level. The images of the fish were compared to multiple sources to ensure correct identification, including photographs of fish I had identified with DNA techniques (Randall *et al*. 2003, Michael 2009, Choat *et al.* 2012, van Lier 2013, FishBase 2015).

Behaviour was identified in the first 5 seconds of watching an individual; swimming was recorded as No Behaviour as it was the baseline behaviour. Behaviours included chasing other fish, being chased, and eating. If the fish was swimming initially but a behaviour occurred within the five seconds then that behaviour was recorded instead of No Behaviour. Harems of parrotfish in this study included those made of different species and varying arrangements of phases. This was due to the imprecise knowledge of male fish with regards to their distance to a group of females to be identified as a harem. Also as mentioned, the issues with identifying each phase correctly means a harem could not be identified with one hundred percent accuracy that it contained a single male and multiple females.

3.3 DATA ANALYSIS

The parrotfish data from the stand methods of the three sampling years 2012, 2014 and 2015 were analysed for comparisons among sites and habitats in Tongatapu. This was completed using statistical tests with the use of R studio 3.2.2 (2015, R Core Team, R Foundation for Statistical Computing, packages: car, vegan) and graphics using Excel (2013).

Abundances

All abundance data of parrotfish across the three different years were initially tested for normality through the use of the Shapiro-Wilk test. The assumption of homogeneity of variances was also tested using a Bartlett test or Levene test. When data met the assumptions of normality and homogeneity of variances, with or without transformations, a parametric test, such as an ANOVA, was completed. I used an ANOVA as the dependent variables (count) in the study were from related groups – habitats (crest, back, edge) within sites (Atata, Kolonga, Offshore, Sopu, Tukutonga. The ANOVA was also completed between sites, as these were located in a similar area of varying size. These were tested for each year (2012, 2014 and 2015), and also across all years. A paired t-test was completed to see if the two stations within the Atata 2012 site were significantly different due to its influence on the significance between sites in 2012.

Behaviour

The behaviour data was separated into the frequency of Behaviour captured on film vs No Behaviour captured from the total number of fish seen (Chapter 2). This frequency behaviour data was tested for independence of behaviour from habitats and sites comparing the years (2012, 2014 and 2015) with a Chi- square test. A Chi- square goodness of fit was used because it measures how well the observed Behaviour data fitted the expected value distribution if the Behaviour frequencies were independent from the habitats and sites. It analyses the categorical data of Behaviour vs No Behaviour and the count for each habitat and site.

Harem structure

The harem structure (frequency of different harem sizes) is discrete data for which I am testing the dependence of structure on habitat, site and year sampled. These structures were put into categories, 3 to 4, 5 to 6, 7 to 8, 9 and above (as per Chapter 2).

Further analysis was completed for 2015 because this was the largest data set, I used non metric multidimensional scaling (NMDS) to interpret the similarity in harem sizes and lone male abundance, comparing both sites and habitats for the 2015 data. The goal of NMDS is to collapse information from multiple dimensions (e.g., from multiple species, sites, etc.) into just a few dimensions for better visualisation and interpretation. Ordination methods geometrically arrange sites so that distances between them in the graph represent their ecological distances. Use of ranks makes NMDS robust even if relationships between distances and dissimilarities are not linear. The Bray-Curtis index was used as the default measure of difference, this uses sums and differences and ranks them by harem mean.

Species richness

Species richness data was tested for differences site and year. Through the Shapiro-Wilks test and Bartlett test the assumptions of normality and homogeneity of variances were met. Therefore, a parametric test was completed, this was an ANOVA. A multifactor design was used so to test the effect of site and year on the species richness. Site comprised five levels, although not all sites were sampled each year, and year comprised three levels. When all combinations of the levels are included (as they were here), the design is a factorial design.

3.4 DATA RESULTS

In total, 13 parrotfish species were identified across the three years of sampling on five reefs within the semi-enclosed lagoon of Tongatapu. Most species were present in all three years, with *C. spilurus* being the most abundant across all sites and years. Others, such as *Chlorurus strongylocephalus* was only observed in 2015 on Tukutonga reef. Harem structure was mostly consistent across years, with the smaller sizes (3-4) being most frequent in all years. Males were noticeably present in a harem only four times overall, twice in 2014 and 2015.

Abundances

Parrotfish abundance for each sampling year, and with all years combined, met the assumptions of normality and homogeneity of variances. For 2012, site (F=4.326, df=2, 6, p=0.069), and habitat (F=0.294, df=2, 6, p=0.755), there were no statistically significant differences between the site means. These data support the means shown in Figure 13a, with the edge habitat showing average parrotfish abundances below the other habitats but no significant difference between the three habitats. Habitats were shown to have no statistically significant differences between the group means in 2014 (F=0.652, df=2, 9, p=0.544) (Figure 13b). The final sampling year, 2015, was different again from the previous two sampling years. With there being a statistically significant across habitats (F=8.519, df=2, 12, p=0.005). Figure 13c shows this difference in parrotfish abundance means across habitats, with edge and back habitats having their average abundances approximately 300 below the crest average abundance.

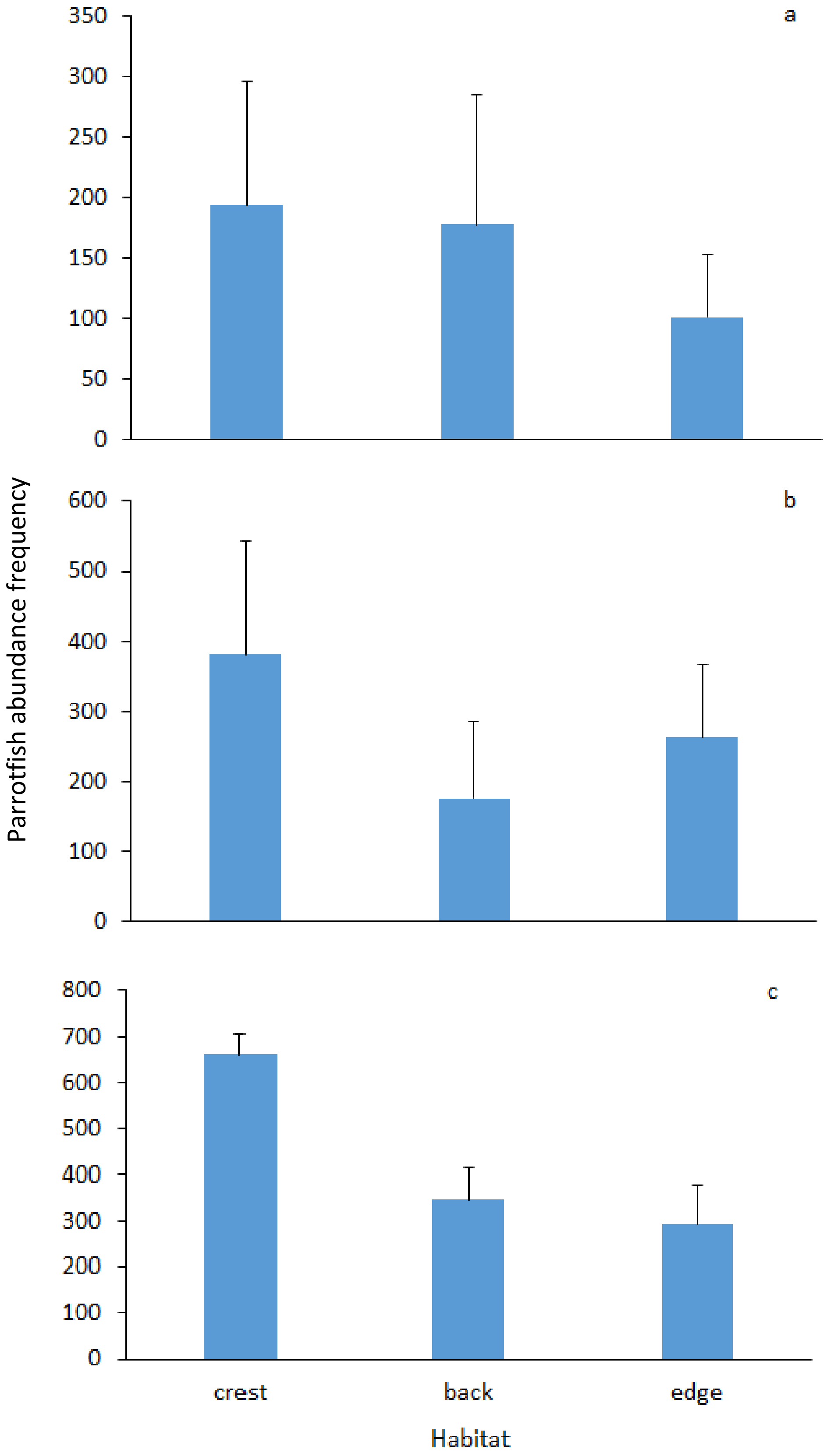

Figure 12a-c. Bar chart showing mean (± S. E) parrotfish abundances across habitats and years (a – 2012, b – 2014, c – 2015) sampled in shallow reef areas within Tongatapu semi enclosed lagoon.

The increased mean abundance (± S. E) across Atata may be explained due to the differences in sampling stations and the management applied to the fisheries occurring there (Figure 14a). Therefore, the stations within Atata were tested for differences, there was no significance between the two stations within Atata site (t=0.943, df=1, p=0.519). The data does support the large variance seen between sites (Figure 14a).

In comparison, in 2014 there was a statistical significance in the means for parrotfish abundance across sites (F=9.532, df=3, 8, p=0.005), supported by the variation in means between sites (Figure 14b). This difference between means of the sites is as large as approximately 500 between these two sites, with Sopu and Tukutonga lying in-between them (Figure 14b).

The final sampling year, 2015, was different again from the previous two sampling years. With there being no statistically significant difference in group means across sites (F=0.352, df=4, 10, p=0.837), with similar variation around the mean also (Figure 14c).

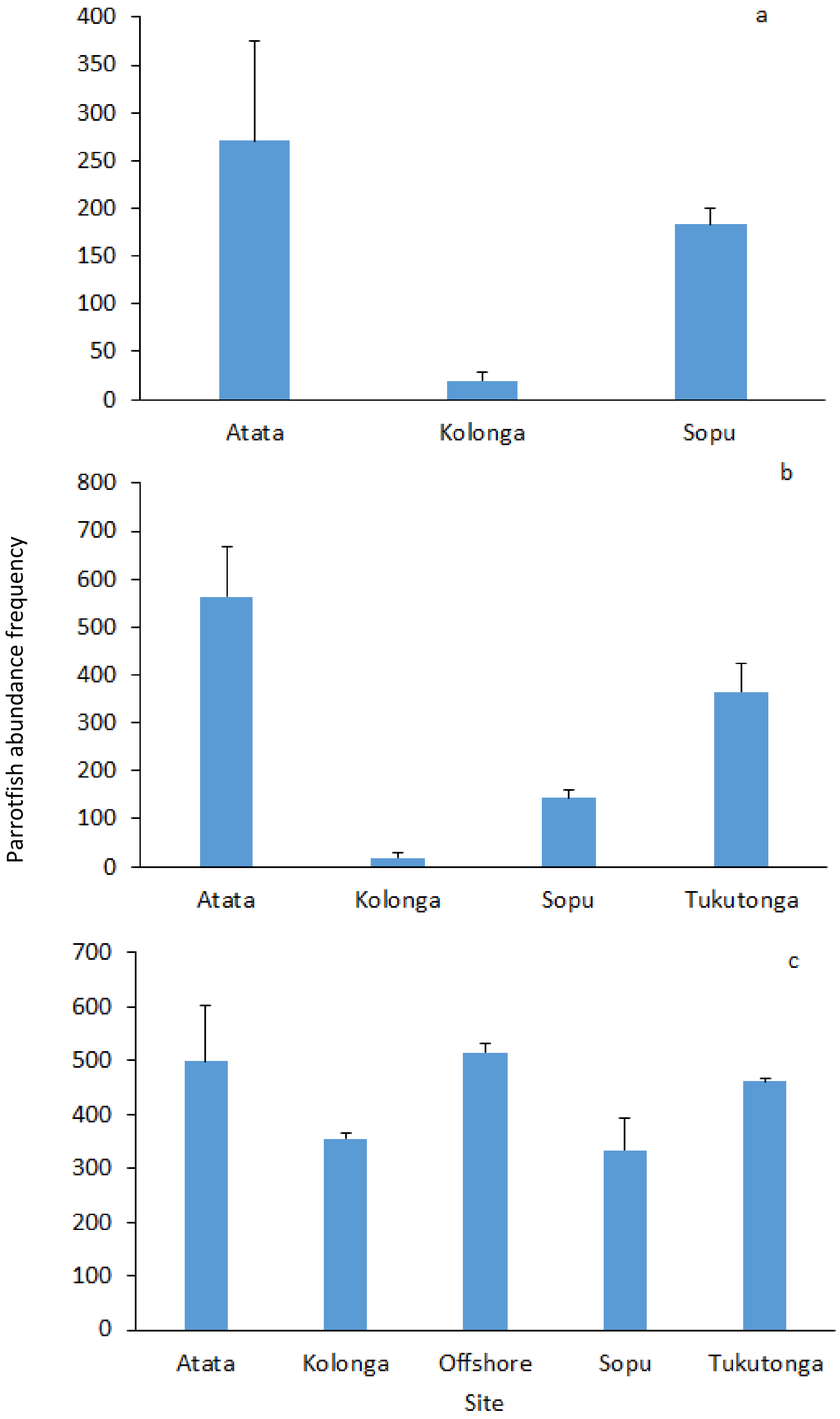

Figure 14a-c. Bar chart showing mean (± S. E) parrotfish abundance across sites and years (a - 2012, b – 2014, c – 2015) sampled in shallow reef areas within Tongatapu semi enclosed lagoon.

A comparison across all years showed that habitat (F=6.515, df=2, 9, p=0.018), site (F=6.567, df=4, 9, p=0.009), and year (F=8.974, df=1, 9, p=0.015) had no statistically significant differences between the group means. However, all interactions showed that there was a difference in parrotfish abundance that could be explained by the differences among habitat and site means, or both (F=1.066, df=8, 9, p=0.458), the differences among habitat and year means, or both (F=1.405, df=2, 9, p=0.294) and by the differences among site and year means, or both (F=0.311, df=3, 9, p=0.817).

The three way interaction of year, site and habitat showed that the difference in parrotfish abundance could be explained by the differences among all independent variables (F=0.467, df=6, 9, p=0.817).

Behaviour

A Chi-square test for habitats from 2012 showed that the Behaviours captured were all dependent on sites; Atata, Kolonga and Sopu (Table 1). This is supported by the overall chi square test on habitat (Table 1). For the 2012 data set overall the Behaviour was shown to be dependent on habitats and independent of sites (Table 1). Box-whisker plot of this 2012 data show the Behaviour frequency to be most varied at the crest habitat but the spread of their frequencies overlap (Figure 15). Figure 16 shows the variation in Behaviour frequency across sites, where Kolonga and Atata are significantly different, with no overlap.

Table 1. Chi square statistics from the parrotfish behaviour data sampled in 2012 in the shallow reef areas of the Tongatapu semi-enclosed lagoon, on the dependency of behaviour on habitat and sites overall. Also habitats within each site.

	χ-stat	**P value (α>0.05)**	**df**
Habitat	8.465	0.015	2
Site	0.614	0.736	2
Atata	28.896	<0.00001	4
Kolonga	11.960	0.003	4
Sopu	33.901	<0.00001	4

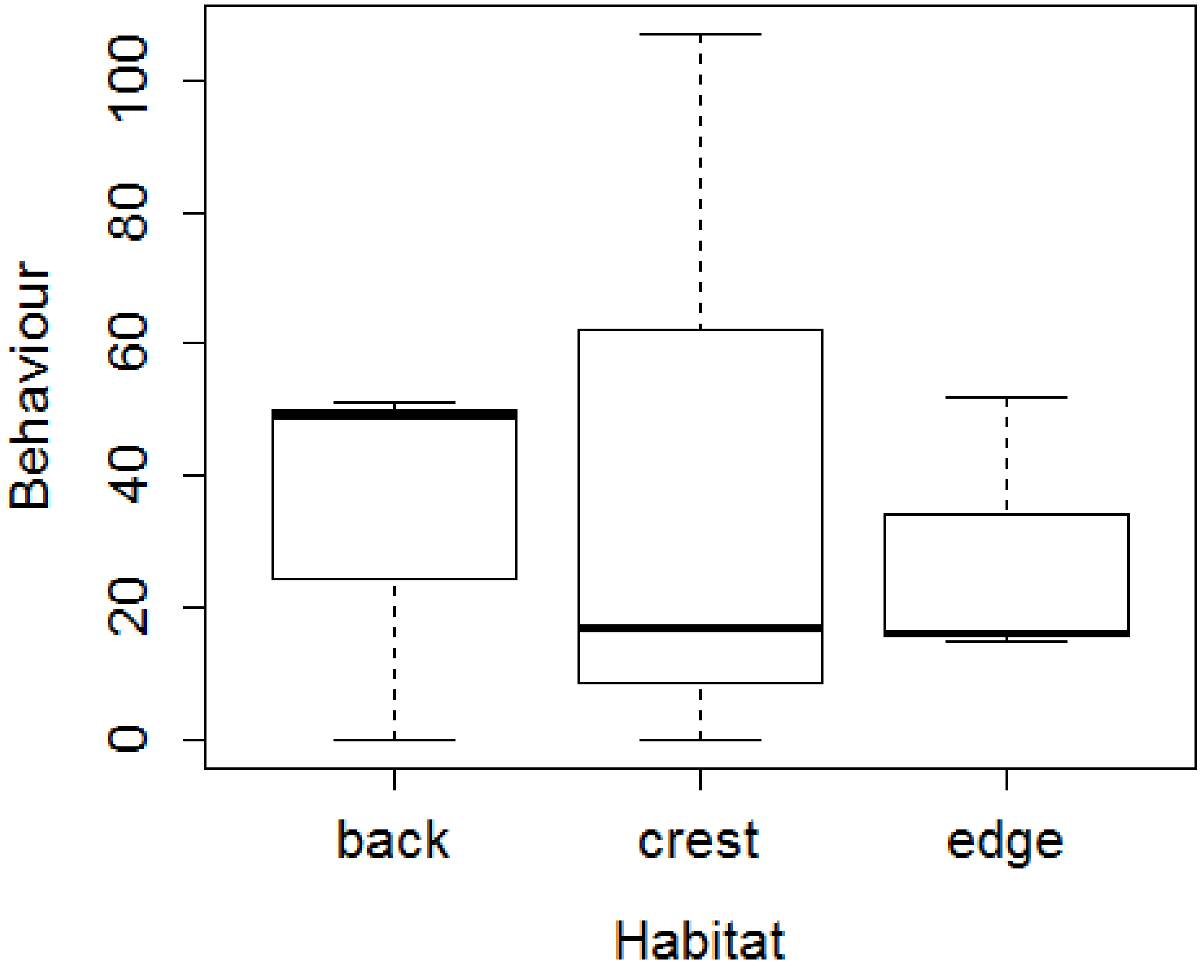

Figure 15. Box-whisker plot of the frequency of parrotfish behaviour across three habitat transects (Back, Crest, Edge), sampled from the shallow reef in the semi-enclosed lagoon of Tongatapu in 2012.

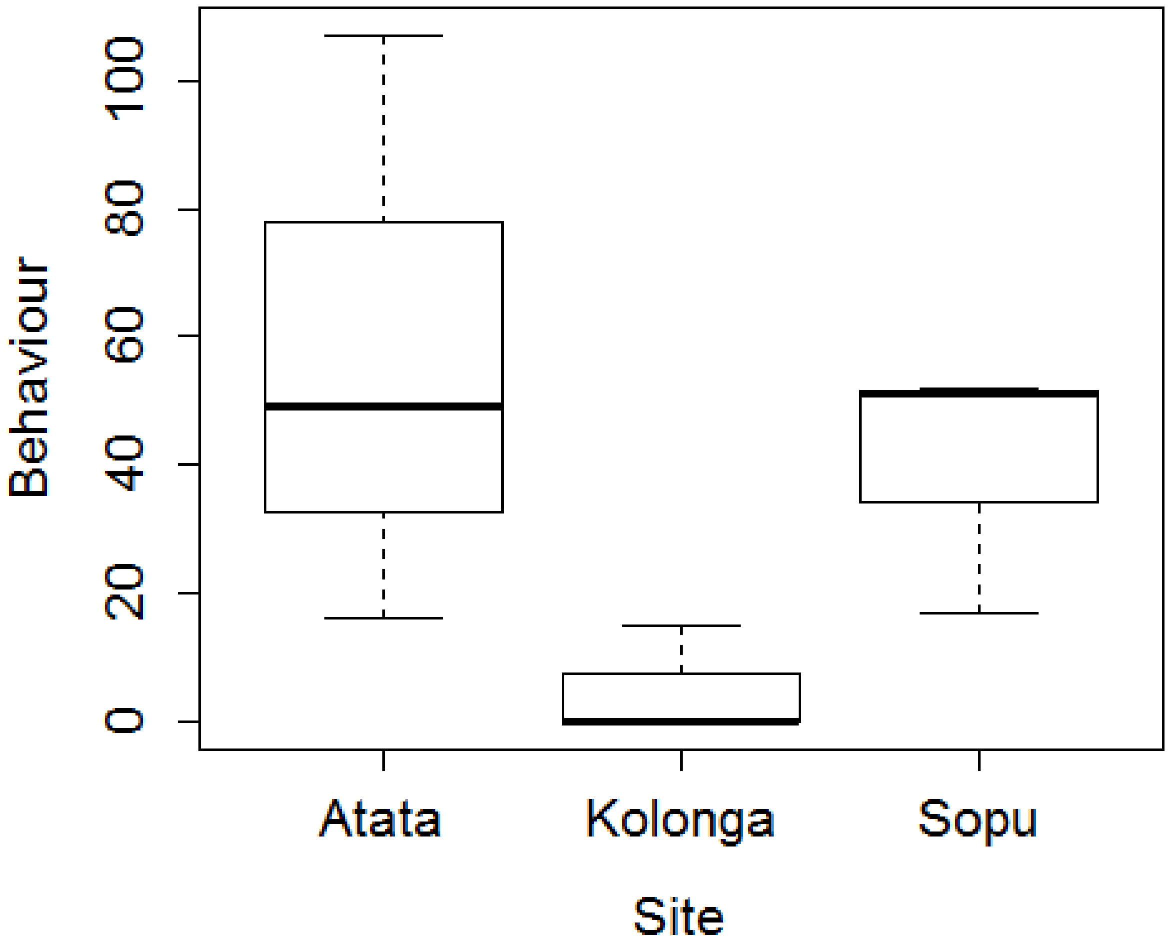

Figure 16. Box-whisker plot of the frequency of parrotfish behaviour across three sites (Atata, Kolonga, Sopu), sampled from the shallow reef in the semi-enclosed lagoon of Tongatapu in 2012.

A Chi-square test from 2014 showed that the Behaviours captured were independent of habitats for each of the sites Kolonga and Sopu, and dependent on habitats for the sites Atata and Tukutonga (Table 2). For the 2014 data set overall the Behaviour was shown to be dependent on both sites and habitats (Table 2). These test support the box-whisker plots (Figure 17, Figure 18), seeing large variation in both habitats and sites within 2014, with outliers removed.

Table 2. Chi square statistics from the parrotfish behaviour data sampled in 2014 in the shallow reef areas of the Tongatapu semi-enclosed lagoon, on the dependency of behaviour on habitat and sites overall. Also habitats within each site.

	χ-stat	**P value (α>0.05)**	**df**
Habitat	90.538	<0.00001	2
Site	175.289	<0.00001	3
Atata	186.790	<0.00001	6
Kolonga	0.756	0.685	6
Sopu	0.391	0.822	6
Tukutonga	18.574	0.0001	6

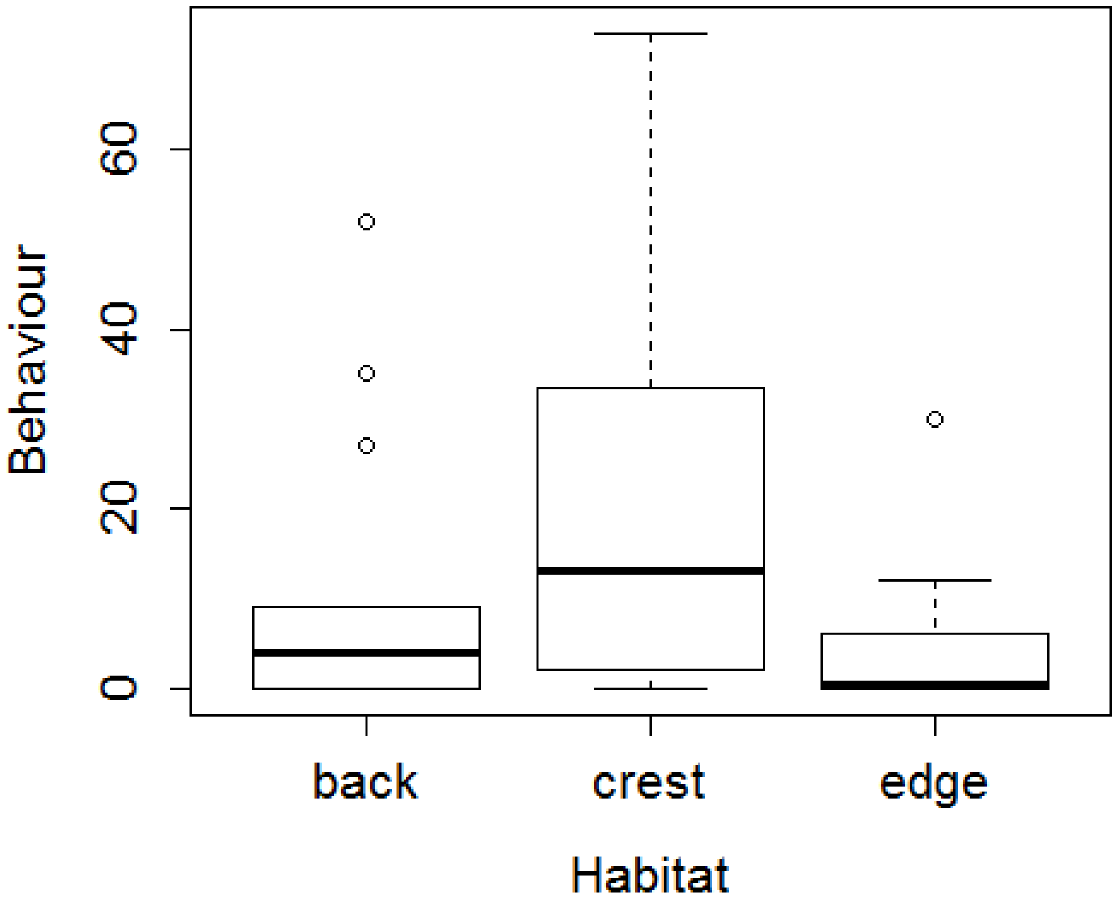

Figure 17. Box-whisker plot of the frequency of parrotfish behaviour across three habitat transects (Back, Crest, Edge), sampled from the shallow reef in the semi-enclosed lagoon of Tongatapu in 2014.

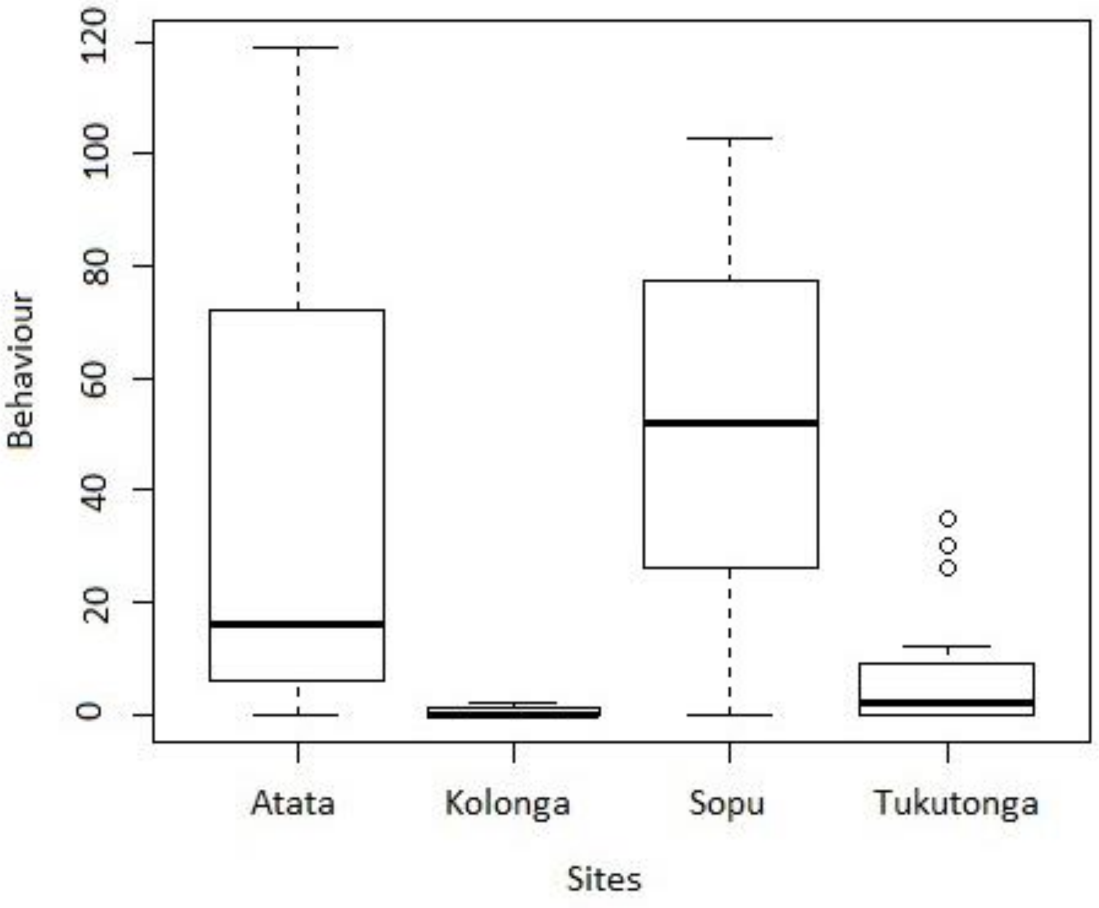

Figure 18. Box-whisker plot of the frequency of parrotfish behaviour across four sites (Atata, Kolonga, Sopu, Tukutonga), sampled from the shallow reef in the semi-enclosed lagoon of Tongatapu in 2014.

Chi-square tests for 2015 data and showed that the Behaviours captured were independent for the sites Kolonga and Sopu, which is similar to the 2014 sites. The sites where Behaviour was dependent on habitats were Atata, Offshore and Tukutonga.

As with 2014, the 2015 data set overall was shown that Behaviour was dependent on both sites and habitats. (Table 3). Outliers were removed from the 2015 data set, however there are still high data points pulling out the dependency on site and habitats (Figure 19, Figure 20).

Table 3. Chi square statistics from the parrotfish behaviour data sampled in 2015 in the shallow reef areas of the Tongatapu semi-enclosed lagoon, on the dependency of behaviour on habitat and sites overall. Also habitats within each site.

	χ-stat	P value (α>0.05)	df
Habitat	40.517	<0.00001	2
Site	222.301	<0.00001	4
Atata	37.059	<0.00001	8
Kolonga	4.810	0.090	8
Offshore	20.809	0.00003	8
Sopu	5.804	0.055	8
Tukutonga	20.209	0.00004	8

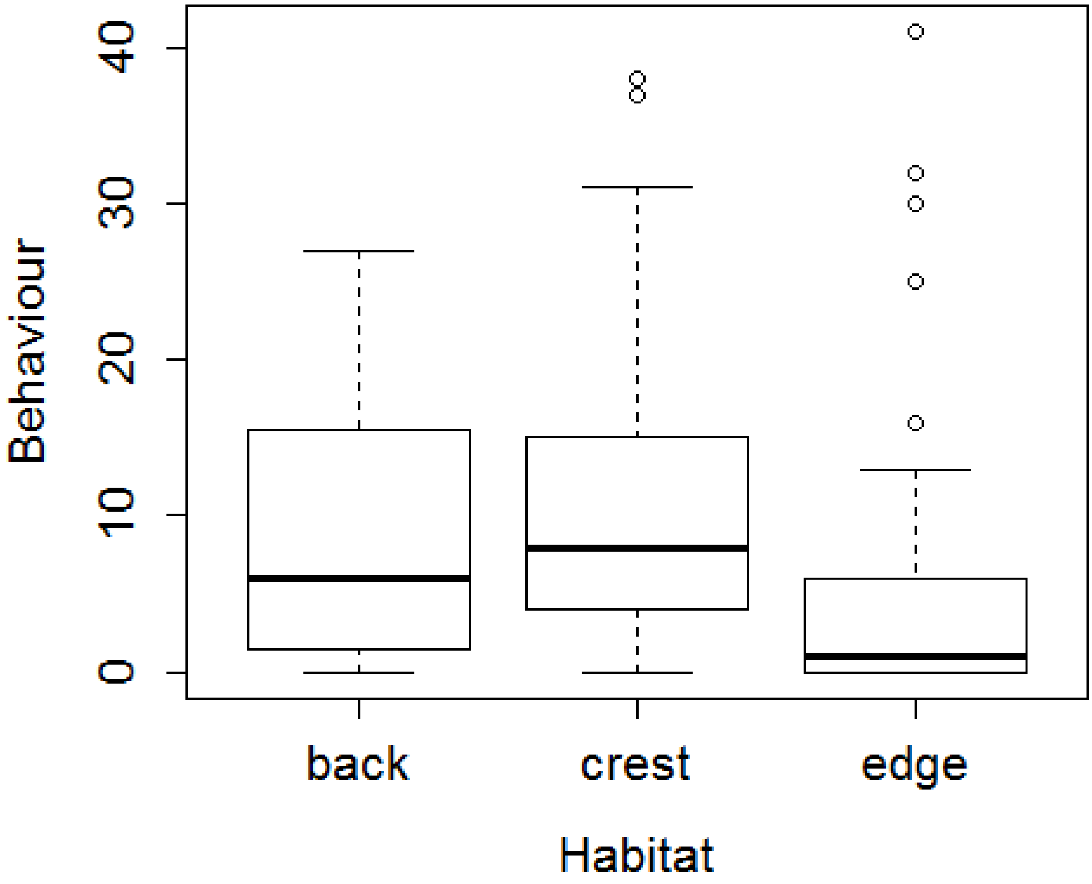

Figure 19. Box-whisker plot of the frequency of parrotfish behaviour across three habitat transects (Back, Crest, Edge), sampled from the shallow reef in the semi-enclosed lagoon of Tongatapu in 2015.

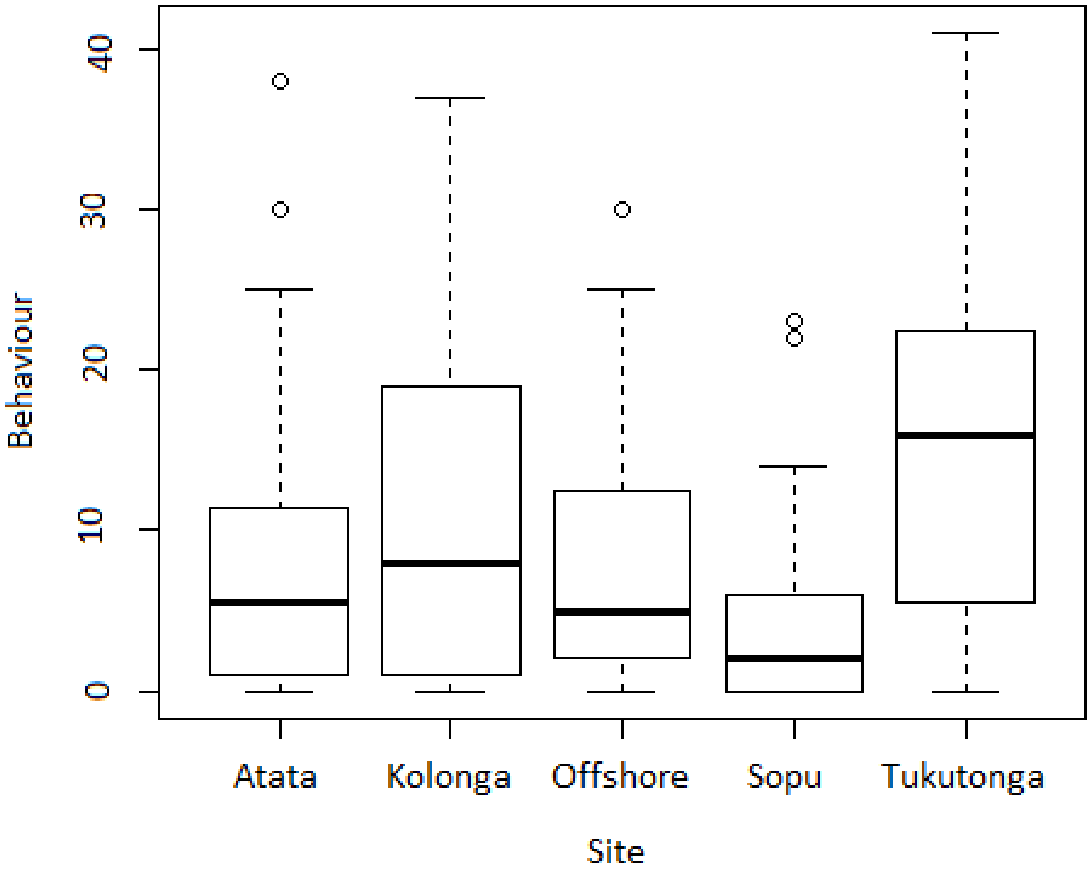

Figure 20. Box-whisker plot of the frequency of parrotfish behaviour across five sites (Atata, Kolonga, Offshore, Sopu, Tukutonga), sampled from the shallow reef in the semi-enclosed lagoon of Tongatapu in 2015.

Harem structure

Chi-square tests were run for each year on the different size structure categories to show the likelihood of observing the different harem size structures was dependent on the site and habitat. For the data set from 2012, harem size structure was shown to be independent of the site but was dependent on habitat type (Table 4). This was the opposite for the 2015 data set with harem structure being dependent on site and not on habitats (Table 4). For the final data set from 2014, harems structure was not dependent on site or habitats (Table 4). Chi-square tests were then run to investigate each habitat separately across the years (Table 4). From 2012 harem size structure was found to only be dependent on the edge habitat, not on crest habitat or the back habitat (Table 4).

Table 4. Chi square statistics from the parrotfish harem structure data sampled in 2012, 2014 and 2015 in the shallow reef areas of the Tongatapu semi-enclosed lagoon. Completed on the dependency of harem structure on habitat and sites overall. Also habitats within each site.

Year	Data tested	χ-stat	P value (α>0.05)	df
2012	Site	9.76	0.135	6
2012	Habitat	25.937	0.0002	6
2012	Crest	9.474	0.149	6
2012	Back	3.186	0.364	6
2012	Edge	23.559	0.001	6
2014	Site	4.624	0.866	9
2014	Habitat	10.101	0.120	6
2014	Crest	8.06	0.528	9
2014	Back	7.670	0.568	9
2014	Edge	7.144	0.622	9
2015	Site	30.060	0.003	12
2015	Habitat	6.172	0.404	8
2015	Crest	18.65	0.097	12
2015	Back	42.603	0.000003	12
2015	Edge	3.52	0.757	12

In the 2014 data set none of the habitats significantly influenced the harem size structure. In comparison in 2015, the back habitat was the only one harem size structure was shown to be dependent on (Table 4). These test support the overall dependency results from the previous chi-square tests, finding dependency on habitat for 2012 which must be driven by the edge transect and no dependency for the 2014 data (Table 4).

Across years, sites and habitats the smaller harem sizes were at a greater frequency (Figure 21a-l). In 2015, across all sites there was a greater number of all harem size classes observed, however, there the sample sizes were also larger in this year (Figure 21a-e). The Kolonga sample site was lowest in frequency for all harem size classes in 2012 and 2014. However, in 2015 the high frequency of the smallest harem size classes increased it so that it was more similar to the other sites that year (Figure 21c, h, l). The highest frequency of harems appeared in general over all sampling years and sites, appeared to be in the crest or edge habitats (Figure 21a-l). There is great variability in the frequency of harem size structures between year, site and habitat.

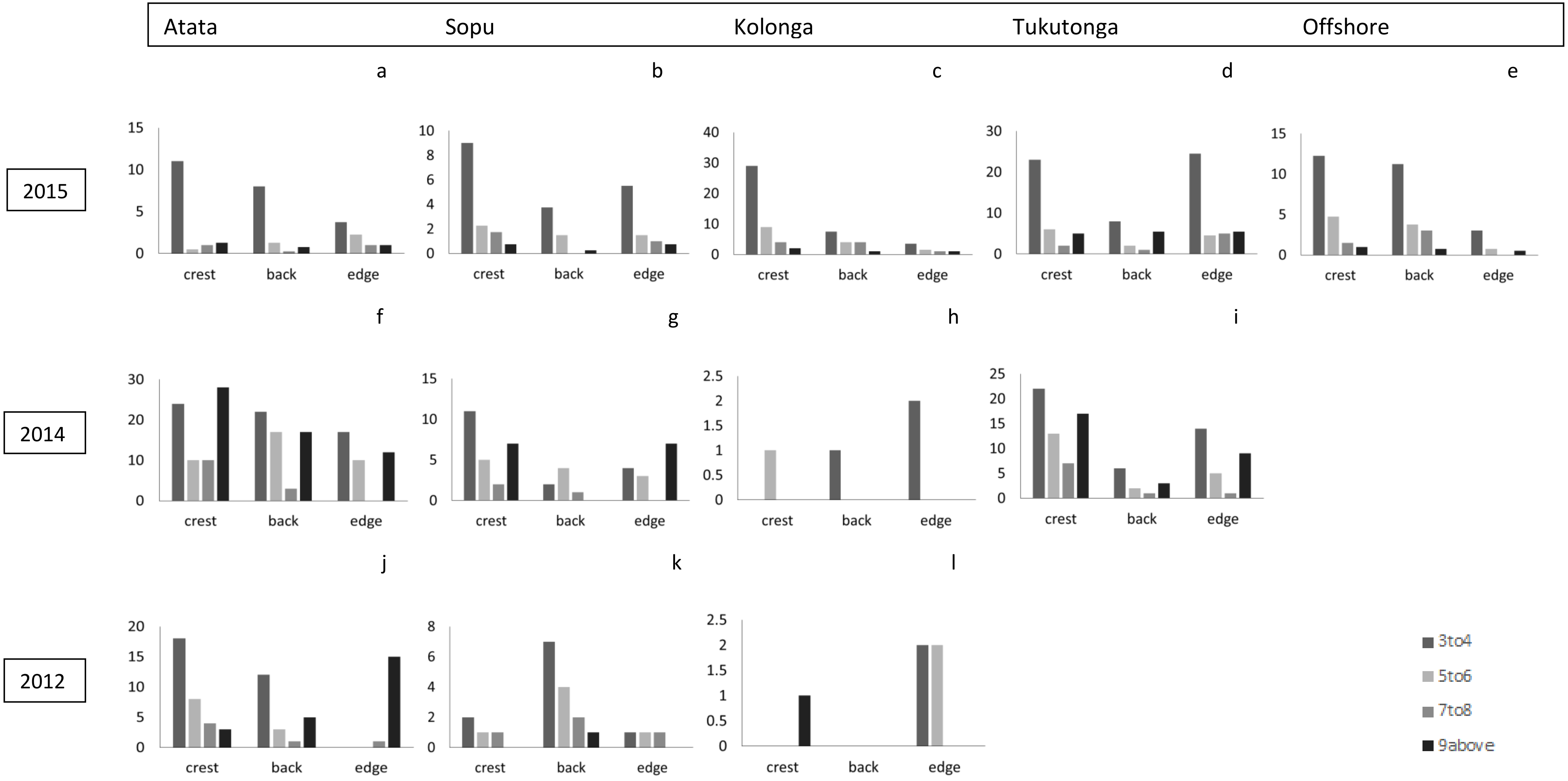

Figure 21a-l. Size frequency of harems in four class sizes across the three sample years (2015 a-e, 2014 f-i, 2012 j-l), at a range from three to five sites, and three transect habitats within each site. The frequency of the harems on the y axis and transect habitats along the x axis.

Harems with males were not very frequent, with none seen in 2012 and only two seen in 2014 and 2015 respectively (Table 5). The size structure and site at which they were seen varied for both years showing no discernible pattern.

Table 5. Frequency of Harems which contained males across sampling years 2012, 2014 and 2015. Also the sites and their size structure where they were sited in the shallow reef areas of the Tongatapu semi-enclosed lagoon.

		Year	
	2012	2014	2015
Frequency	0	2	2
Size	0	48, 18	5, 13
Site	NA	Atata, Tukutonga	Offshore, Kolonga

The NMDS were completed on the 2015 data to investigate the similarities of sites based on the frequency of harem sizes sampled, also including the lone male count (Figure 22). These plots below were completed using the harem abundance per Transect (3 camera stands) to make it easier to present here, although the patterns remain with the full data spread.

At first glance there appears to be no pattern in the data (Figure 22). However, we can see some grouping, more like clouds than tight clusters that are worthy of further investigation (Figure 22). For example, Tukutonga is represented by the purple diamonds and is loosely clustered on both axis NMDS1 and axis NMDS2 (Figure 22). In contrast, Sopu, offshore and Kolonga are spread out across axis NMDS1, but tight on axis NMDS2 (Figure 22).Finally, Atata spread across both axes, with all stations more similar to stations from other sites (Figure 22). What appears to be the main driver of these patterns is the presence of the large harem in one station in Atata, reducing its similarity to the other sites and the abundance of lone males in another station. The effect of these drivers is not ameliorated by the presence of smaller harems that were observed in other sites.

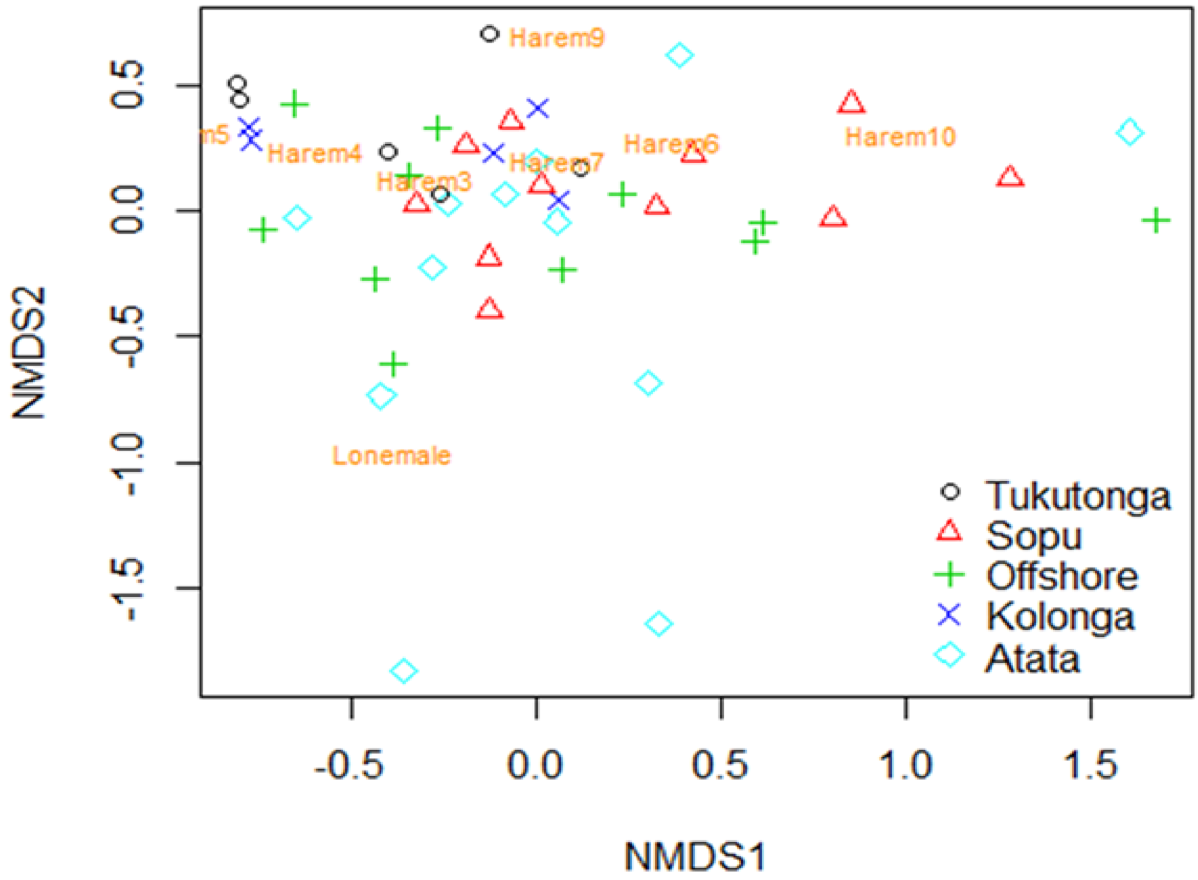

Figure 22. Non-metric multidimensional scaling plot of parrotfish harem size structure across the five sites (Atata, Kolonga, Offshore, Sopu, Tukutonga). Harem size identified by the number following Harem. With the lonemale being the lone males.

The spread of the data is the same as the initial NMDS with no clear patterns; however, with some cloud groupings occurring (Figure 23). For example the edge data represented by the green plus shape, is loosely clustered on both axis NMDS1 and axis NMDS2 (Figure 23). In contrast, crest is spread loosely across both axes, as is back but slightly tighter spread on axis NMDS1 (Figure 23). As in Figure 22, the main drivers of these patterns appear to be the presence of a large harems in crest and back habitats, and also lone male presence, reducing the similarity these stations have to others. The clouding pattern for the edge habitat appears to be due to stations being more similar to each other in terms of smaller harem sizes (Figure 23).

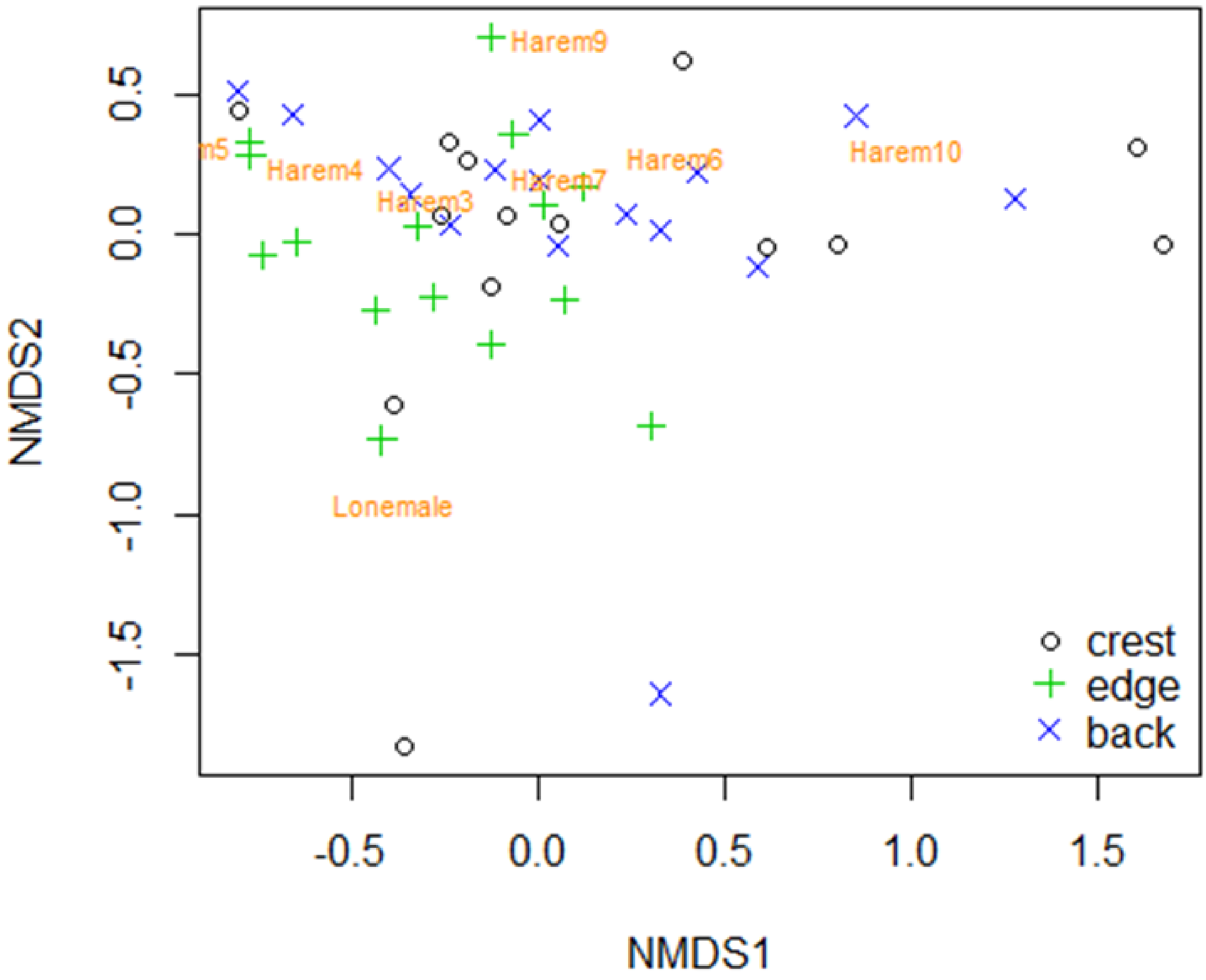

Figure 23. Non-metric multidimensional scaling plot of parrotfish harem size structure and lone males across the three transect habitats (crest, back, edge) from 2015. Harem size identified by the number following Harem. With the lonemale being the lone males.

From both NMDS plots the polarity on axis NMDS1, appears to be the large harems that are unevenly distributed across sites, and the polarity, or driver of dissimilarity on axis NMDS2, appears to be the presence of males not easily associated to a harem (Figure 22, Figure 23). Actually, only at two stations did I observe a male with a harem.

Species richness

The parrotfish species richness data met the assumptions of normality through the Shapiro-Wilk test (p=0.078), and the assumption for homogeneity of variances for Year with the Bartlett test (p=0.439). The Bartlett test could not be completed with Site as there needed to be at least two observations in each group. ANOVA tests showed that Site was nonsignificant (F=2.496, df = 4, 1, p=0.239) also Year did not have a statistically significant effect on total species richness (F=0.947, df= 1, 4, p=0.402), and there was no significant interaction of site and year (F=0.164, df= 4, 3, p=0.914).

There is a difference in species richness across the sites, where the two sites further from the mainland: Atata and Offshore, have the highest species richness across all years and sites (Figure 24). Offshore was only sampled in 2015 so cannot be compared across years, however, Atata has been consistently with high species richness across all three sampling years (Figure 24). Those sites closer to shore such as Kolonga, Sopu and Tukutonga generally had a lower species richness across the years they were sampled (Figure 24). This supports the first ANOVA that site has a significant effect.

There is no trend over the sampling years in species richness. Overall the 2015 sampling had higher species richness identified across over half of the sites but there is no visible trend within and between sites over the years as they are all variable (Figure 24).

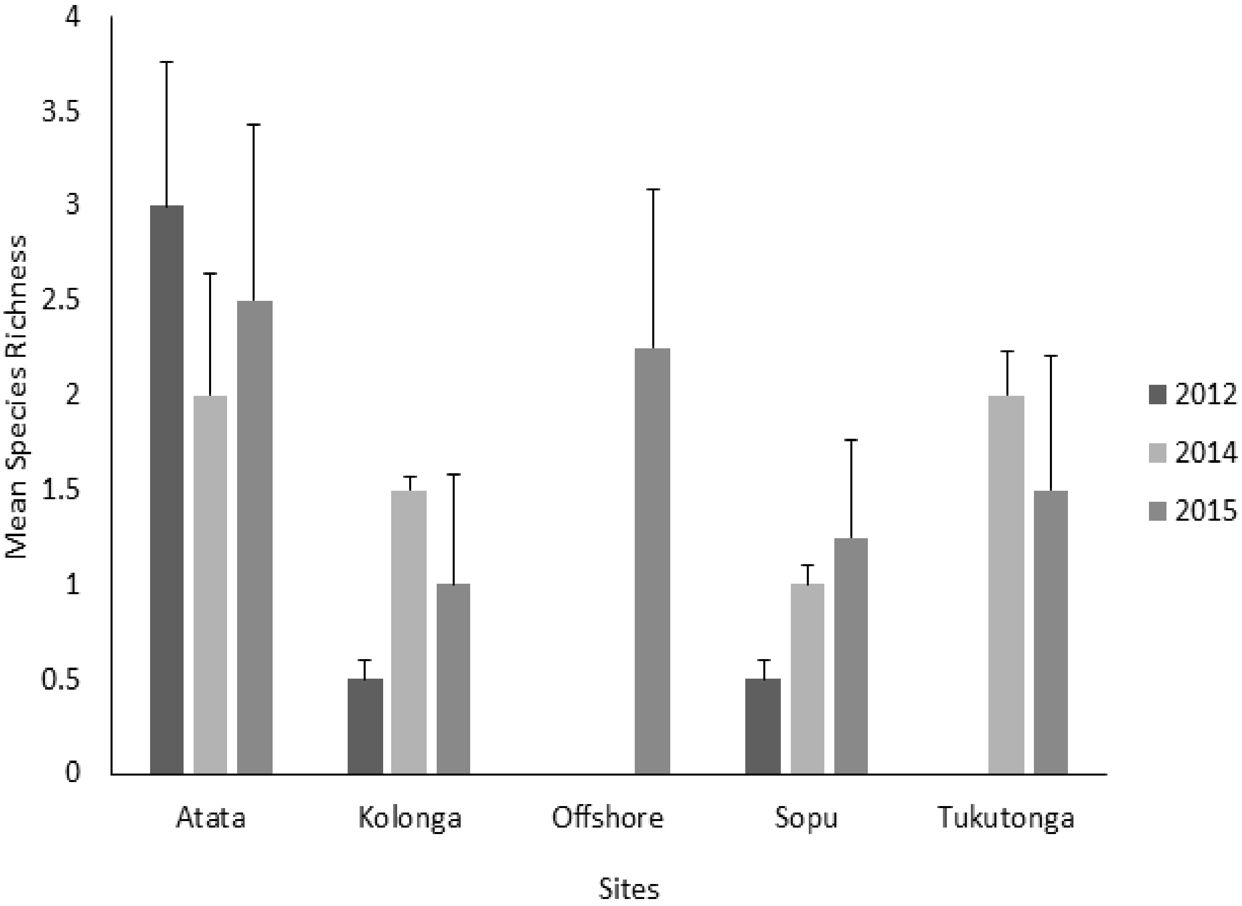

Figure 24. Bar chart showing mean (± S. E) of mean parrotfish species richness across all sites (Atata, Kolonga, Offshore, Sopu, Tukutonga) sampled in shallow reef areas within Tongatapu semi enclosed lagoon in 2012, 2014 and 2015.

3.5 DISCUSSION

This study sought to establish a baseline data set that includes temporal and spatial variation, with the use of three sampling years, for parrotfish within the semi-enclosed lagoon of Tongatapu, Kingdom of Tonga. Supporting the continually growing body of literature studying the baseline data of herbivorous reef fish and their roles (Clua and Legendre 2008, Taylor 2014, Taylor *et al.* 2014, Choat *et al.* 2012). Results across years varied in the effect sites and habitats had on the population dynamics of parrotfish. Overall, there were 14 species identified on these reef-flats, which was low compared to the 99 species of parrotfish that are known worldwide (Parenti and Randall 2011) and the 28 species identified in the shore and epipelagic waters within the Kingdom of Tonga in 2003 by Randall *et al.* (2003) (Full list of species is in Appendix 2, Table A2, Table A3, Table A4). There are many reasons that this could be low, including the use of camera technique, as they are known to under-estimate species diversity (Tessier *et al.* 2005). It may also vary due to the anthropogenic influences on their behaviour and abundance, therefore what species were caught on camera in this research (Chapter 2).

The Atata reef was found consistently to have the highest average parrotfish species richness across all years; this is likely due to its location and the health of the reef. As even moderate fishing pressure can effect coral reef fish communities (Craig *et al.* 2008). Removing these extra pressures makes for a healthier reef, in terms of its complexity in overall reef fish species diversity (Graham and Nash 2013), allowing for important ecosystems functions to be completed and to maintain reef health. The Fish Habitat Reserve (FHR) may have an effect on the species average and abundance making it higher than usual, on top of these anthropogenic stressors being reduced. These communities close to marine reserves have noted positive effects with greater governance towards their resources (Gillett 2009). This provides incentive to conserve these resources, however outsiders may be negatively impacted as they feel they are restricted or denied fishing rights (Gillett 2009). It has been shown that with these reserves there can be enhancement of adjacent fisheries, which ultimately will benefit all fishing communities, not only those in proximity to the resource (Roberts *et al.* 2001). The abundance in particular in 2014, when the FHR was sampled, was greater than the other sampling sites, supporting the conservation efforts put in by the community. The reef surrounding Atata appears, as a whole, to have high average species richness and abundance, which may support the use of these SMA and FHR principles (Gillett 2009). It is key to note that location may be an important factor for these results.

The Offshore sampling reefs were similar to Atata, in that they had a high average parrotfish species richness and they were located offshore. Offshore sites were only sampled in 2015, the temporal comparison was not possible. However, these sites show a high average species richness within the years they were sampled. As with Atata, the location of these sample sites influences the anthropogenic pressure the reef fish face. One of the islands, Pangaimotu, does have a tourist site, with a restaurant and swimming area. However, the reefs are located further into the semi-enclosed lagoon, away from the main tourism side of the island. The lower exposure to anthropogenic factors from fishing and pollution, with increased distance from the mainland and other environmental stressors, are what is shared between the Offshore and Atata sites that would influence their high species richness and abundance (Craig *et al.* 2008, van Lier 2013).

The Tukutonga reef was the middle ground for average species richness, and only ranged a small difference between the two sampling years. The abundance of parrotfish were also reasonably high. This site was not expected to have this result as it is located near an old rubbish dump, where there may be release of the rubbish waste chemicals leaching into the water (Global Environmental Facility 2013). This is also a low economic area with high cases of homemade latrines and piggeries (Morrison 1999, Lovell and Palaki 2000, Global Environmental Facility 2013). These latrines and piggeries are built on the waters edge of the Lagoon, flushing out at high tide causing water quality degradation (Morrison 1999, Reopanichkul *et al.* 2009). This water degradation was shown to have a significant influence on the decline of fish abundance on the reef (Bozec *et al.* 2005, Fabricius *et al*. 2005, Reopanichkul *et al.* 2009). Phosphorous released through sewage and pig waste is found to be toxic to fish at high concentrations, whereas at lower concentrations it causes eutrophication (Okun 1997). The study completed by Reopanichkul *et al.* (2009) found with the increase of macro algae due to nutrient enrichment, this was accompanied by lower abundances of herbivorous fish. There was considerable concern about the clearance of mangroves and land activities along the shoreline in the Pe'a sector where Tukutonga is located, therefore influencing the sedimentation in this sector (Morrison 1999). These mangroves also help to collect the nutrients before reaching the reefs and prevent the water degradation.

The two sites Sopu, and Kolonga share similar levels of high fishing pressure and anthropogenic effects (Clua and Legendre 2008). The Kolonga reef was variable across the years in both average species richness and abundance. In 2012 and 2014, the abundance of parrotfish was the lowest by far and this would have an influence on the species richness. As with Kolonga, the Sopu reef was low in average species richness. Concern arises for sites, such as Kolonga and Sopu, where the average species richness is lower than one at some stages. Different parrotfish species play different parts to their functional role and may influence the ecosystem structure if there is no species richness (Bellwood *et al.* 2004, Kazancioglu *et al.* 2009). The consumption of the different parts of algae, will influence the coral reefs resistance to change state. Both Sopu and Kolonga sites were similar in their close proximity to land and mangroves, with medium to high fishing pressure. This fishing pressure will reduce the number of parrotfish and species seen on these reefs, which will alter the community assemblage (Clua and Legendre 2008). Their location closer to land may be facilitating this high fishing pressure, influencing the reef complexity and the species assemblages that are there.

Across habitats at the sampling sites in 2012 and 2014, there was no significant difference in abundance. The crest habitat did have a higher median across all years, however, was only significantly different in 2015 due to the high number identified on the crest habitats. Across all years the crest habitat had greater abundances from the other habitats. In general, the largest sized harems were identified across all years in the crest and edge habitats. With the smallest sized harems nearly consistently being the most abundant across all habitats. Across all years Atata, comparing between sites, had the highest abundance of larger harem sizes. The larger harem sizes were more often than not made from juveniles of conspecifics and other species. This may show the juveniles roaming further from the safety of mangroves as a nursery environment due to the lower fishing pressures, or that they may not use mangroves. On reef slopes and crests there is patchy large turf algae and macro algae that are defended (Michael 2009). This helps to explain different species diversity between the habitats, dependent on the number of fish species that feed at the different algae located spatially within the reefs.

Informal discussions with local people, in particular the fishers, suggests the coral reefs were in a healthier state only 10-20 years ago. Within this time the rapid development of areas around Tongatapu, especially the capital Nuku’alofa and along the waterfront, have affected the nearshore environment. Information was emerging during this time that the lagoon was occasionally turning green, with increased turbidity and declines in fish catches (Global Environment Facility 2013). Many of the species of seagrass were being covered in algae and there was greater removal of mangroves (Global Environment Facility 2013).The reef has been blasted and mangroves removed to create channels for boats and to build new wharves (Zann 1994). The mangroves filter the terrestrial ground water, settle sediment and trap trace metals (Global Environment Facility 2013, Morrison and Brown 2003). Additional anthropogenic threats could exacerbate the degradation of Tonga’s mangrove ecosystems include overfishing, overharvesting of mangroves for firewood, construction material and tapa dye and the expansion of coastal developments into the mangrove areas (Lepa 2013, MESCAL 2013). An example of this coastal removal is in Hahake, the eastern side of Tongatapu, along Kolonga. Effects of this removal include wave and storm surge events that damage the roads. Sand mining along the coast has also reduced the beach height, with significant reduction in environmental quality of the reef. This is linked to high nutrients from the lagoon, creating the potential for eutrophication, there is a lack of ability of the system to generate sediment to replace the loss (Tonga: Climate resilience Sector Project 2013). This highlights the importance of the connection between land and sea to be protected, in this case the effects on mangroves and land impacting the coral reefs (Steneck *et al.* 2009). Mangroves have also been identified for the connection they play with reefs when acting as nurseries for reef fish (Mavrogenis and Kelman 2013). The striped parrotfish, *Scarus iserti*, has a high dependence on mangroves and seagrass, where its adult populations appear to be effected by the absence of these nursery habitats (Nagelkerken *et al*. 2002). No surprise then that the proximity and management of the mangrove habitat in relation to the coral reefs has been shown to have significant effects on reef fish, even for some species it may have greater implications then fisheries management (Nagelkerken *et al.* 2012). This may be of particular importance for those sites, such as Kolonga and Sopu, with close proximity to mangroves.

Social structure and species richness

The schooling behaviour has an effect on the feeding and chasing behaviour of parrotfish on the reef. As noted with further analyses on the 2012 data (van Lier 2013), a dominant fish by biomass in Kolonga was the damselfish (Pomacentridae). This species is well known for its highly territorial behaviour of filamentous algal patches, which actually provides little benefit to the wider reef, its health and maintenance (Randall 2005). To overcome these type of herbivores it is more likely that the parrotfish in larger schools or harems can feed and overpower the damselfish. This aggressive behaviour towards parrotfish, causing them to swim away, was counted for its frequency. Being chased by damselfish was particularly common, rarely chasing occurred due to aggression from conspecifics. Another behaviour was feeding; scraping, excavating and consuming algae. These behaviours were seen to be dependent on transect habitats for all sites, where the crest had the highest frequency across all sites, and the edge habitat consistently had the lowest. This is supported in general by the abundances found on these habitats, with greater number of abundances generally there was greater number of behaviours. In those sites with higher damselfish abundance such as Kolonga, being chased was the primary behaviours. If these feeding behaviours are being overcome by territorial chasing rather than feeding, then the parrotfish will likely be moving further away to areas with less aggression and greater resource availability. This becomes a cycle, as they do not complete their functional role and there is a greater abundance of algae and less parrotfish available to maintain the algae levels. There is then a distinct possibility that reefs, such as Sopu and Kolonga, could be in danger of changing state, to a macro algal dominated state (Hoey and Bellwood 2011). This has been well identified as being a negative state for the reef and is difficult to reverse (Hoey and Bellwood 2011). Parrotfish have been known to be deterred from areas with a certain high level of macro algae (Hoey and Bellwood 2011), therefore they cannot facilitateits return to a coral dominated state.

There is a lack of clarification for parrotfish in the distance between these fish which makes them behave as a harem (Robertson 1972, Mumby and Wabnitz 2002). In this study, all groups of parrotfish were identified as harems, whether or not they contained males due to this lack of clarification. Where we could not identify males in proximity to these aggregations of females with/without juveniles, out of the focal range of this study due to the territorial behaviour of some species, therefore could not exclude them as harems (Mumby and Wabnitz 2002). However, only four true harems, males seen with females, were observed over the three sampling years. These occurred both in the distant sites and sites in close proximity to the mainland. Initial phase fish will only change to the terminal phase when they are required to lead the harem. Due to this, only a low number of terminal males are expected to occur in any population. An additional confusing factor is colouration occurring first, this means that we may be identifying these individuals as males, yet they may be in the process of changing phase without actually behaving as a male (Munoz and Warner 2003).

The sex change process is influenced by fishing pressure, it plays an important role in shaping coral reef fish communities and populations by removing the large biomass individuals (McClanahan *et al.* 1997, Nyström *et al*. 2000, Mumby *et al.* 2006). These large biomass individuals are generally the males and older fish, with their removal there is greater pressure on the population to have females to change sex (Clua and Legendre 2008). This will continue to occur at smaller sizes and may affect the overall sizes of the population. These larger biomass individuals are also more likely to move into deeper reef areas where there are greater resources. This is another reason we may not have seen as many males in this study. Understanding the age and size dynamics of parrotfish on a coral reef is important due to the different functional roles in a single species (Hoey and Bellwood 2007, Bonaldo and Bellwood 2008, Fox *et al.* 2009). Throughout the lives of a parrotfish, as their body size changes so does their bite size which in turn influences the different species of macro algae they target when feeding (Bonaldo and Bellwood 2008). This alteration in their feeding behaviour has an effect on the sediment recycling, transportation and reef stability (Bellwood *et al.* 2004, Bonaldo and Bellwood 2008).

More often than not, males were seen alone and not with a harem of females, no displays of territoriality were seen in the videos and there were on occasion, in particular at Atata, schools of males together feeding. The high counts of lone males were seen in stations of Atata in 2015 and were often not associated with sites that had large harem sizes. This may be important to note as they were not around where the larger harems or schools of juveniles were but instead at other stations. These stations may have provided greater food availability, complex environment for hiding for these generally larger biomass fish. Males that were seen with a harem were both in sites close to the mainland and offshore. No true harems were identified in 2012, with greater number of sites and stations completed in 2014 and 2015 two each year were seen, and their size was not consistent. Due to this low number of sightings no significant results can be drawn except that they were not very frequent. This may be of particular importance for monitoring the reproduction behaviours of the parrotfish species in Tongatapu. It presents a potential risk to their populations, their role on the reef and in the fisheries.

The species that had the greatest abundance over all the years was *C. spilurus*, and was the most dominant species from species diversity work completed by van Lier (2013). This species has recently undergone a taxonomic reassignment from, *C. sordidus* (Choat *et al*. 2012). *Chlorurus* species are suggested to have a Pacific origin and a westerly expansion (Choat *et al.* 2012). They have been identified as sister species of *Scarus*, five of six sister species pairs had allopatric distributions and only one pair (*Chlorurus bowersi* and *Chlorurus bleekeri*) had significant overlap (Choat *et al.* 2012). *Chlorurus sordidus* and *C. spilurus* are only shown to have a 9.1% overlap in range (Choat *et al*. 2012). *C. spilurus* appears to exhibit the expected behaviour of many reef fish species, with older individuals moving offshore to more resource rich habitat and the distribution of juvenile fish also lends further support to mangroves acting as nursery habitats (Kimirei *et al.* 2013).

The results from this research compare favourably with other studies that have examined fish communities around Tongatapu (Clua and Legendre 2008, Friedman *et al*. 2009). In particular, Clua and Legendre (2008) and their results on the effect fishing pressure had on the parrotfish community. This is important as they found fishing pressure had a significant effect on shaping assemblage structure, with similar results to this we are most likely seeing this same effect. Fishing pressure has been shown to drive the biomass of parrotfish and its variation across sites, as this pressure has been identified as the most influential factor on length at sex change (Taylor 2014, Taylor *et al*. 2014). This is supported by length at sex change from the 2012 data set (van Lier 2013). The transformation stage from juvenile phase to a sexually active initial phase was identified to be between 8-10 centremetres, while individuals typically transformed from an initial phase to their terminal phase between 25-28 centremetres. This transformation to initial phase is early when compared to literature values that estimate it at approximately 15 centremetres (Randall *et al*. 1990). The increased pressure of fishing may be favouring smaller body sized individuals, which is known to occur in parrotfish (Hawkins and Roberts 2003). On top of the influence from fishing pressure, this could be influenced by higher sea temperatures due to climate change. What occurs in some fish is the increase in developmental rate at the cost of growth (Dhillon and Fox 2004, Jonsson and Jonsson 2009, Neuheimer and Grønkjaer 2012). However further study would need to occur in this area to confirm this. This could be tested using the videos, in particular of the Atata and Offshore sites, where with less fishing pressure we may see larger sized fish.

Across all sites the most dominant species were *C. spilurus* and *S. rivulatus*. From previous analysis completed on the 2012 data Atata contained larger individuals, on average, and the Sopu and Manuka reefs contained, on average, smaller individuals (van Lier 2013). What is also mentioned as an effector on the parrotfish communities is proximity to mangroves, and macro algae cover. Distance from mangroves, in relation to biomass, tends towards the hypotheses of mangroves acting as a nursery, as the sites closer to mangroves exhibited a smaller biomass that is a requisite of juvenile fish (Nagelkerken *et al.* 2000, Mumby *et al.* 2004, Kimirei *et al.* 2013). Even with this support, further investigation is necessary for testing this connectivity of the different habitats to definitely conclude that this phenomenon is occurring between the mangroves and reefs. With increased macro algae cover these is likely a negative effect on the biomass of fish. Once the cover of macro algae reaches a threshold then many fish, including the parrotfish, tend to avoid that area (Hoey and Bellwood 2011). Overall this decreases the density of fish, effecting the species richness, abundance and biomass (Hoey and Bellwood 2011).

Tongan impacts

The Tongan people's health relies heavily on a healthy reef ecosystem (Sun *et al*. 2011). The understanding of parrotfish and their dynamics to this subsistence fishery is important for fisheries management to consider. Knowledge from Tongatapu, where it is the most developed and populated, will help with preparation with areas of further development in other Tongan islands, such as the tourism destination of Vava'u. As mentioned, due to this effect and shifting of potential adult and larger species offshore for resource availability, there is an effect on the environment (Kimirei *et al.* 2013). The result is that these environments are healthy with more complexity and greater fish diversity, likely supporting the observed Atata and Offshore results. With increased complexity there is more niche space and micro habitats, allowing for more diverse communities (Vergnon *et al.* 2009).

Conclusion

The parrotfish are an important component in the coral reef environment. The population dynamics of the parrotfish varied between and within transect habitats, sites across years. The important anthropogenic effects, which are influenced by location of the sites and their proximity to other habitats, are likely to drive these demographics. Fishing pressure has been highlighted by many previous studies, and is suggested, with support from these results also, to be the factor that needs to be under great consideration for its influence on haremic structure and species diversity. This study provides a good baseline data set to monitor any population dynamic changes for the parrotfish in the semi-enclosed lagoon of Tongatapu.

4 CHAPTER 4 – DISCUSSION

This research presented here was based on two aims: 1) to investigate underwater visual consensus methods and their application to parrotfish research; and 2) apply one of these methods to investigate population dynamics of parrotfish in coral reefs on the island of Tongatapu, in the Kingdom of Tonga. Understanding the parrotfish population dynamics including; abundance, behaviour, species richness and harem size structures and the effects upon them, both anthropogenic and environmental, allow me to begin addressing necessary conservation and management issues that have been highlighted throughout. Also further areas for investigation to enhance our knowledge and applicability of possible management structures have been proposed.

Methodology

Results from my study highlight the need to consider the question posed when choosing sampling methodology for reef fish research. This supports work by Dee *et al.* (2014) and Holmes *et al*. (2013) who also found the methods used need to consider many factors, including; the taxonomic level, the habitat and diversity of the study. The use of video in comparison to in-the-field recording has been widely discussed, with many biases and advantages to each method (Samoilys and Carlos 2000, MacNeil *et al.* 2008, Stoner *et al*. 2008, Bernard *et al.* 2013, Holmes *et al*. 2013, Mallet *et al*. 2014). In particular, there are time constraints with analysing the videos, adding to processing time and cameras adding to the cost (Spampinato *et al.* 2008, Boom *et al.* 2014). Video work may be more time intensive, but the images are accessible for multiple analyses (Boom *et al*. 2014). For example, the footage in this research was used for parrotfish and damselfish of the coral reefs only, future research could focus on other species of fish or investigate fine scale changes in different habitats and sites. Video resolution is also important for the analysis of more refined parameter estimates such as fish length, colouration and species identification.

As with many studies it appears the use of more than one method would be more beneficial, so using the UVC method along with support from a visual census completed at the time would help to balance out the biases (Bennett *et al.* 2009, Murphy and Jenkins 2010, Bergseth *et al.* 2015). In the case of this study, for measuring or estimating pelagic and territorial fish variables, the swim and stand methods did not have significant differences on the data overall. It then comes down to the benefits from each method. The swim based video method has some biases, in that the movement as the video is captured can be more erratic due to the person swimming, currents and weather. The fish also react to the swimmer, on top of abiotic factors. I found it more difficult in the turbid conditions to identify fish from the swim method than the stand method as the video view is continually moving through the medium, rather than there being a stationary view. Other studies support this, as turbidity has been reported to influence the visibility of transects and sampling observations when using UVC techniques (Halford and Thompson 1994, Labrosse *et al*. 2002, Hill and Wilkinson 2004). The novel stand method helped to remove the biases of the person in the water, which allowed the fish to return to what we assumed to be natural behaviour. In comparison, fish react towards the swim videoing method showing general fleeing or inquisitive behaviour, as the person was continually in the fish's vicinity. The stand method also provides a 360° view and a longer time captured along the same distance of the habitat. This is equivalent to a diver circling in deeper water, but again without the diver influencing the behaviour of the fish during the data collection period.

The research completed here has been useful in gathering more information on the effectiveness of UVC methods, in particular the novel method of using a 360° videoing stand. There is also potential of these methods to collect data on different variables of both pelagic and territorial fish on the in shore reefs of the Tongatapu lagoon, in the Kingdom of Tonga. Both methods provide benefits and biases towards the data collected for pelagic and territorial fish. With further application and alterations, these methods may be used for further research and monitoring of the reef and its fish inhabitants.

In regards to community or citizen science, videos may also be useful to teach volunteers or those without higher qualifications knowledge of the reef (Harvey *et al*. 2001). This will enable them to see what they will be viewing and identifying. In the Reef Check programme applied in places such as the Great Barrier Reef, there is a module in training volunteers both practical diving and in the classroom completing video transect methods (Hill 2008). In addition, video can reduce the volunteer bias that occurs as visual observers have been shown to over-or-underestimate the dimensions of a survey area, which results in a corresponding over-or-underestimation of fish density (Harvey *et al.* 2004, Gillett *et al.* 2012). With sufficient swimming ability and general knowledge of reefs, volunteers would have the ability to set up the stand videoing system, making data collection easier. Cost of the video is expensive and will need the technology to view the footage and analyse it, but once this is set up it is an easy system to follow, and should reduce bias from visual identification.

Parrotfish

The novel stand method was identified as showing a good representation of the parrotfish dynamics in Tongatapu coral reefs, with the least biases, and therefore, the stand method was used for the parrotfish research. This allowed for fish individuals to not be repeated twice, which has often been a difficulty with UVC without video through personal error (Mantyka and Bellwood 2007). Herbivorous fish feeding behaviour has been monitored by the use of stationary video on reefs such as the Great Barrier Reef (Mantyka and Bellwood 2007, Hoey and Bellwood 2009), and Ningaloo Reef (Michael *et al.* 2013), similar feeding behaviour studies have also been completed on artificial reefs (Takagi *et al.* 1997). The use of stationary video removes the presence of the observer, therefore having minimal or less of an influence on the behaviour of fish (Takagi *et al.* 1997, Bellwood *et al.* 2006, Mantyka and Bellwood 2007). To ensure the fish returned to a behaviour we assumed to be normal there was a period for them to adjust after the camera was set up. Time allowance for behaviour to return has been seen with the use of the STAVIRO method, which is similar to a stationary video, except it is one camera rotating the 360°, where it was left for 10 minutes without disturbance (Mallet *et al.* 2014). Stationary video has shown difficulty in capturing cryptic species (Colton and Swearer 2010), which is similar to hiding behaviour of parrotfish on the reef, this is why giving them time to return to normal behaviour with the stand method is beneficial. Data collected through stationary video methods, such as the stand method in this research, have been used for collecting data on biomass, weight, length and behaviour of reef fish (Hoey and Bellwood 2009). The video data collected in this research is available for further studies to investigate further into not only when behaviour occurs, as completed here, but what behaviour, such as algal species eaten, bite rate and size.

These video methods have also proven their worth in identifying species, however, there is difficulty using stationary videos for some reef fish (Michael *et al.* 2013). Parrotfish colouration has been discussed as conspicuous and variable, both between and within species (Warner and Robertson 1978). Smaller species and individuals were shown to be hard to identify in a study by Michael *et al.* (2013), where some IP scarids were indistinguishable they were grouped together as IP individuals, this is similar to what was completed in this research. They are believed to have this ability to change colour for antipredator response (Crook 1999), camouflaging (Cott 1940, Crook 1997), confusion (Bellwood 1986) or conspecific communication (Fernald 1977). The schooling parrotfish species, such as *C. spilurus, S. psittacus* and *S. globiceps,* are characterized with simple colour patterns, they can undergo physiological colour pattern (PCP) changes within a few seconds (Bellwood and Choat 1989, Crook 1997). In comparison, the species, such as *S. niger, S. frenatus, S. altipinnis*, have a solitary, non-schooling behaviour, as a result they display only one colour pattern as juveniles (Bellwood and Choat 1989, Crook 1997, Crook 1999). The bullethead parrotfish, *C. spilurus,* was identified as a main species from this research within Tongatapu. A study of *C. spilurus*, in Lizard Island, Australia, identified PCPs in the juveniles (Crook 1997). The influence on the PCP includes body size, schooling behaviour and foraging activity (Crook 1997). These influences, such as short term behaviour, has shown a relationship with freshwater fish colour change also (Barlow 1963). With the strongest influence identified as the individuals body size, followed by schooling behaviour.

These PCPs were seen in this research and provided further complexity with species identification. In this research the fish were counted for abundance, not species richness when in large schools of juveniles due to this complexity. There is potential for this colouration to be seen in other species of parrotfish that also have a schooling behaviour, which will have varying colouration due to many influences (Crook 1997). Further study by Crook (1999), found that there were up to seven distinct PCPs for *C. spilurus,* but that also some of these were shared with other juvenile scarids. This highlights the way these species form heterospecific schools, as it is likely to be more beneficial joining a school more like itself, especially under greater predation pressure (McRobert and Bradner 1998, Crook 1999). Short term colour change was seen in this research with not only juveniles but also the mature phases of parrotfish, in a range of species (Personal observation). It appears that similar to juveniles there are specific influences that cause this change. Purely through observation, the terminal phase and intermediate phase fish would lose brightness in colour, becoming dull and darker in colour. This change is likely influenced by stress, as it has been mentioned by Hemmings (1971) that the laboratory environment can influence them to behave differently. Further research should be completed in the field, to identify these short term colour changes in males, and initial phase parrotfish. An example of this is through the use of 3D imaging or HD video, to identify the slightest change in colours involved, helping to identify smaller species and individuals as well as variations within a single species colouration.

The potential for a parrotfish species to reverse phase shifts seems to be influenced by their food selection, which is highly dependent on the species composition (Feitosa and Ferreira 2014). The removal of macro algae by herbivorous fish is a key process on the coral reefs (Choat 1999, Choat *et al.* 2002, Streit *et al.* 2015); it moderates the competition for benthic space and facilitates the recovery of coral population following macro algal dominance (McCook 1999, Hughes *et al.* 2007, Streit *et al.* 2015). The social structure, as well as the parrotfish species present influences their ability to play this role. The question for the male parrotfish is: does the cost outweigh the gain for defending a territory, when they have less time to find food and in a smaller area. Very few males were seen in harems in this study, with a greater number being identified alone. This may support the idea that, at the time where these studies were completed, that territorial behaviour had the greatest benefits.

The functional role the parrotfish play is dependent on the variety of species, including sizes of species and phases, the feeding rate, bite size and algae type being consumed. Feeding preferences are greatly dependent on species composition and competition (Feitosa and Ferriera 2014). A greater abundance of schools may be due to the fact that feeding in large groups, when there are such large numbers, on the reef is more energetically beneficial. Schools can overcome territorial fish, such as the damselfish, and remove the algae within their territory (Vine 1974, Crook 1999). As interference competition is known to structure their habitat use when aggressive territorial damselfish defend their patch of algae causing reduced feeding rates and food availability (Low 1971, Itzkowitz 1990, Feitosa and Ferreira 2014). Encounters between schooling herbivores and these site-attached species are highly characteristic of shallow reef habitats (Crook 1999). The interactions with other species in the shallow in shore reefs have the capacity to influence both the location and duration of feeding behaviour in juvenile herbivores (Crook 1999). These interactions between damselfish and parrotfish were seen in this study. There was a high number of small harems across the reefs, however, these were often made of larger individuals. In comparison, the larger social structures generally contained many smaller sized individuals who, when in smaller sized harems, may not be able to overcome aggressive territorial reef fish. If the parrotfish are kept off sites due to this aggressive behaviour of trophic competitors, then they are not helping to maintain the macro algae levels on the reef. With removal of certain species there will hopefully be remaining functional diversity, where other herbivorous species, can continue completing their role. However, when species diversity decreases, such that a role is left empty, then alternate states can occur on the coral reefs.

These social structures help to explain parrotfish staying in a smaller location with favourable environmental factors, which means if sampling completed was not in favourable conditions we may have seen less. Whether they are large species and can be protected from most predators anyway then food is highest priority, whereas smaller species and phases are more likely to look for protection and therefore complexity in the reefs. Social structures in this study showed a trend where the greatest abundance of smaller harem sizes were on the crest of the reef, this may be due to the deeper edge off the crest providing habitat complexity for larger sized parrotfish individuals that were seen to make up these smaller harem size category. This may also explain why the smaller species and phases, which made up a majority of the larger harem sizes, across all years were in the Atata reef. From analysis of reef health by van Lier (2013), this was shown to be one of the healthier reefs in the study, providing greater habitat complexity. Habitat importance on smaller scales needs further study (Taylor *et al*. 2015), expanding on the influence humans are having and that the greater habitats for parrotfish may be further from anthropogenic hubs. Abundance is also important to maintain their functional role and maintain a level of macro algae that will not over power the coral reef system. Understanding the harem and school social structures of the wide range of parrotfish species will help with understanding the effects removal of connected areas will have, such as nursery mangroves and nearby offshore reefs. Further knowledge on the removal of larger biomass individuals, and its effect on sex change of the parrotfish is important for how they maintain a population on top of increased anthropogenic and environmental effects.

Coral reefs support fisheries that are essential for the well-being of millions of people (Pauly *et al*. 2002). The importance of the coral reef fisheries in general is not due to the overall quantities that are occurring, as it is only approximately 5% of the global fisheries catch (Pauly *et al.* 2003), but the importance lies more in the inshore food security, employment and the ecosystem services that it provides for the coastal inhabitants and themselves (Sabetian 2010). With the ever growing populations of people living on the coasts in coral reef regions, there is a continual understanding that they cannot rely solely on the coral reef ecosystem for resources (Mora 2008). This is particularly evident with depopulation of outer islands as people move into the urban areas to improve socio-economically (Craig *et al.* 2008). The impacts of fishing on coral reef communities can be seen in under representation of higher trophic level species such as sharks and barracuda (Craig *et al.* 2008). As these have been removed there is a movement of fishing pressure to other fish such as the parrotfish, changing the structure of the communities and their species richness. This is being seen worldwide as fishing down the food chain occurs (Mumby *et al*. 2006). As the species richness is affected, the important ecological roles species play is altered, affecting the structure of the community. With fishing targeting larger individuals there could be a change in parrotfish evolution. As the smaller individuals are favoured in survival and will reproduce, then the size of parrotfish may alter. This effect has been shown for scarids (among other families) on the Jamaican coast subjected to a gradient of fishing pressure (Klomp *et al.* 2003). Favouring of smaller individuals and species may lead to loss in genetic diversity, which will also affect the behaviour of the fish and the role they play. Smaller fish will feed on different algae and may not be as effective in the removal of algae to prevent macro algae state shifts. This has been evident at Kolonga sites in previous studies with less males and smaller species being more dominant (Clua and Legendre 2008, Friedman *et al.* 2009). We cannot say that fishing pressure alone is driving differences between sites within Tongatapu, however, there is substantial research showing it has an effect and there have been detrimental effects elsewhere (Koslow *et al.* 1988, Clua and Legendre 2008, Stockwell *et al.* 2009, Taylor 2014, Taylor *et al.* 2015, Caribbean; Hawkins and Roberts 2003, and Jamaica; Klomp *et al.* 2003). Further research into the other anthropogenic stressors such as pollution levels in the water, in the tissues of the parrotfish that may influence them is also important.

Coral and fish species richness were shown to be weak indicators of sewage and fishing pressure in a study by Grigg (1995) in Hawaii, however, fish abundance was a strong indicator. Eco toxicological assessment of these field observations depends on the availability of reliable biomarkers to enable a discrimination of natural variations of reproductive functions from anthropogenic impacts (Allner *et al.* 2010). Allner *et al*. (2010) identified that interpretation of field monitoring results concerning reproductive health requires large numbers of samples and detailed knowledge of the natural plasticity of sex differentiation in the study species. Therefore, a strong understanding of the mechanisms underlying the plasticity of sex differentiation in parrotfish is imperative to enable them to be used as a bio indicator. Macro algae and molluscs have been suggested to be most reliable bio indicators (Phillips 1977); however, with further research such as this, it will enable teleosts, such as parrotfish, to be used more extensively as bio indicators. This may include looking at sex changing hormones, such as 17α ethynyloestradiol (EE2), which would be detrimental to the mating system of hermaphrodites. The EE2 hormone is released into the environment through sewage treatments, releasing hormones such as oestrogens and xenoestrogens. It is a widespread concern as the efficiency of removal before it is discharged is highly variable depending on factors such as residence times, plant design and temperatures (Langston *et al*. 2005). An important class of compounds being released into the environment are xenoestrogens, these have the ability to mimic natural estrogens (Arukwe and Goksøyr 1998). The effects of endocrine disrupting chemicals (EDC) are increasingly being brought to public attention, as they are affecting aquatic life and their proliferation within the system can affect populations dynamics of organisms exposed (Vos *et al.* 2000, Wang and Zhou 2013). Skewed sex ratios leaning towards males have been seen both in clams and mussels where they have been exposed to these contaminants (Gagné *et al*. 2003, Hellou *et al.* 2003). Examples of EDCs affects are seen in fish, although concentrations of estrogenic chemicals in fish vary greatly and are dependent on bioavailability, bioconcentration and biotransformation (Arukwe and Goksøyr 1998).

Hormone disturbance has been reported in marine and fresh water fish from chemically induced reproductive impairment. This phenomenon may occur during sex differentiation but its effects may not be manifested until after sexual maturation (Oberdörster and Cheek 2000). Feminization as a result of chemicals released from industrialized estuaries in England has been observed in the flounder (*Platichthys flesus*; Lye *et al.* 1997). This endocrine disruption is well established and widespread, the synthetic and natural compounds which are entering the environment are responsible for many of these effects on marine and fresh water organisms (Langston *et al*. 2005).

There is particular concern for this in SIDs, such as the Kingdom of Tonga, as there is adavancement in building infrastructure, or as the infrastructure struggles to keep up with the population growth. With growing western influence, and the demand for growth and production there is going to be greater pressure on the environment and its inhabitants. The anthropogenic impacts of urbanization, pollution and sewage will need continual management to lessen their influence on the coral reef environments. The crises in the inshore fisheries is also pushing for change in management within the Kingdom of Tonga (Likiliki and Haraldsson 2006).

Management

Over the last two decades there has been a worldwide decrease in the health and resilience of reefs (Wilkinson 1999, Wilkinson *et al.* 2004), with many undergoing catastrophic phase-shifts from coral to macro algal dominance or to sea urchin barrens (Done 1992, Hughes 1994, McClanahan and Muthiga 1998, Adam *et al.* 2015). These shifts, and their frequency, have been influenced due to an increase in human impacts on coral reef ecosystems (Hoegh-Guldberg 1999, Jackson *et al.* 2001, Pandolfi *et al*. 2003, Hughes *et al.* 2005), and have frequently proven to be both persistent and difficult to reverse (Bellwood *et al*. 2005). In response to the increase in phase shifts seen on coral reefs, recent literature has emphasized the need for the management to preserve reef resilience (Nyström and Folke 2001, Hughes *et al.* 2003, Bellwood *et al.* 2004, Folke *et al.* 2004, Cvitanovic *et al.* 2007). This has led to a reassessment of the knowledge of reef ecosystems that play a role in supporting resilience. In many areas of the world with heavy artisanal fishing, the decrease in reef resilience has been linked to the decrease in abundance and species of herbivorous fishes (Lessios 1988, Bellwood *et al*. 2003, Mumby *et al.* 2006, Cvitanovic *et al.* 2007, Mumby *et al*. 2007). Comparisons of fish fauna of inhabited versus uninhabited islands in the Indo-Pacific demonstrate that even low levels of exploitation can rapidly deplete stocks of large fishes, including herbivorous parrotfish (Friedlander and DeMartini 2002, Bellwood *et al*. 2012). Herbivory is one of the key processes responsible for the persistence of coral reefs and is therefore central to the prevention of phase shifts (McCook 1999, Bellwood *et al*. 2006). Information on the distribution, abundance and ecosystem impact of individual species of herbivorous fishes will therefore be vital in the development of management plans for the coral reefs worldwide. This information will also help with understanding the roles that particular types of herbivores play, in both limiting harmful algae and facilitating corals.

Protection from fishing pressure significantly increases parrotfish size and density, which increases their grazing capacity, resulting in a fourfold reduction in cover of macro algae (Mumby *et al.* 2006).

Parrotfish are often overexploited with the use of spearfishing, especially when applying reserves will almost always increase browsing by these fish (Mumby *et al.* 2006). Mumby *et al*. (2007) identified in the Caribbean that this protection induces a trophic cascade, increasing the number of herbivores, leading to an increased grazing rate, reduced macro algae cover, which enhances coral recruitment and the health of the reef. Complementarity among herbivores is important for suppressing algae and facilitating corals, but functional redundancy may also support the resilience of reefs by making them less susceptible to the loss of a single species (Walker 1992). For example, when a species declines, due to overfishing or disease, species with similar functions may be able to fill that role (Adam *et al.* 2015). With increased frequency, intensity and scale of disturbances from environmental influences, such as storms, floods, hurricanes, there will be a change to the importance of parrotfish and herbivores (Mora 2008, Adam *et al.* 2015). Herbivore importance may increase under these disturbances, with new space available after a disturbance there is space for establishment and growth of algae, the herbivore feeding on this algae will allow for coral to grow (Adam *et al.* 2011).

Recognizing a dominance shift may be a critical step towards the identification of indicator species, which could be useful in the indicator-based management systems promoted by several authors (Boyd and Charles 2006, Clua *et al.* 2006), after being validated as reliable indicators of the stage of maturity of a reef fishery or the impact of a fishery on the ecosystem. It may also be an asset for setting up more specific and accurate size-spectra indicators that are shown to be relevant for coral reef ecosystems at the fish assemblage level (Graham *et al.* 2005). Although differences among the reefs herbivores, both intra and interspecific, have long been known, their importance for the reef has only recently become important with the rate of reef degradation and threats (Bellwood *et al*. 2004, Mumby *et al.* 2006, Bonaldo and Bellwood 2008).

Management for the protection and survival of fish, such as the parrotfish, does not need to be isolated to the reefs, it should also be extended to connected ecosystems such as the mangroves. There is the hypotheses for mangroves acting as nurseries for these fish. It is not only important for reef fish but also for deep sea fish, as fisheries in Tonga and worldwide, are largely based in the deep sea offshore fisheries, an example in Tonga is the tuna fishery (Lovell and Palaki 2000). Mangroves are often a neglected ecosystem in many tropical countries and the Kingdom of Tonga is no exception (Zann 1994, Granek and Ruttenberg 2008). A study researching this connectivity between the two systems of several important fisheries species, including these parrotfish, would provide evidence that could support the need for protection of this ecosystem. This is especially important in terms of maintaining the nursery areas in good condition to increase the fishery populations that use this habitat (Mumby *et al.* 2004). It has been seen in species of parrotfish, such as *S. guacamaia* in

Florida, which it does act as a nursery as it requires both mangrove and coral reef habitats to complete its life cycle (Machemer *et al*. 2012). This will have flow on effects on to the coral reefs providing greater fish resources for the community, both in terms of food and services. The habitat features that need to be included in management are sleeping sites, migration pathways and reef complexity, for the movement of fish and Marine Protected Areas (MPAs) that incorporate these (Munoz *et al.* 2014). This has been seen in the coral reefs of Roviana and Vonavona lagoons (Solomon Islands), where seascape connectivity enhanced the ability of the reserves in place to promote the abundance of fish (Olds *et al.* 2012, Olds *et al.* 2014).

No take reserves have been applied on Caribbean reefs for parrotfish (Mumby 2006). A speculative inference from the model completed by Mumby (2006) is that with failure to manage scarid populations outside reserves, then there will be a negative impact on the functioning of the reserves. To achieve scarid population management there needs to be involvement of gear restrictions (Mumby 2006). These no take reserves indirectly suppress algal growth and maintain coral dominance through their protection of herbivorous fish and the ecosystem they function in (Stockwell *et al.* 2009). No take areas can be beneficial as long as mortality outside the area is greater than the predation that occurs inside (Howard *et al*. 2009). This has been seen in the Exuma Cays Land and Sea Park (ECLSP) in the Bahamas, which is a no-take area where fishing has been banned since 1986 (Mumby *et al.* 2011). Mumby *et al*. (2011) found that no-take areas should enhance the biomass of parrotfish. Spill over of adult fish biomass is an expected benefit from no-take marine reserves to adjacent fisheries (Januchowski-Hartley *et al.* 2013). Globally, the coastal habitat there needs to be work with the management of no take areas, they are important for the global response of the coral reef crisis but cannot be used alone (Howard *et al.* 2009). One way to measure the indication of fishing effects on parrotfish has been suggested by Valles *et al*. (2015), from their study completed in the Caribbean. This is through the use of assemblage-level average individual parrotfish weight in cohesion with community based management, but could help as an indicator of over exploitation.

An important ecosystem approach to management that is increasingly being used is MPAs (Mumby 2006, Mumby *et al*. 2007). An MPA is where fishing is eliminated, and the area acts as a refuge for fish (Russ and Alcala 1994, McClanahan *et al.* 1999, McClanahan and Arthur 2001). Implementing MPAs will be effective if sustainable fishing management can also be applied outside of the protected areas (Adam *et al*. 2015). The species within these areas should also be managed individually due to their different life history traits and also their impacts on the coral reef ecosystem. For example, larger species are more vulnerable to over exploitation and are not functionally equivalent to smaller species, therefore, they should have a total or a higher level of protection from fishing (Adam *et al*. 2015). In accordance with this protection from fishing in areas highly affected there could also be implementations in lightly fished areas. Here, it might be appropriate to attract herbivores to these areas, which can be completed through removal of macro algae, creating suitable habitats to support coral growth and structure (Adam *et al.* 2015). When evaluating and deciding on a management option, abundance and characteristics, such as individual size and ontogenetic phase need to be considered (Bonaldo and Bellwood 2008). These MPAs having limited harvesting, in particular on the largest species, such as *Bolbometopon muricatum*, is necessary as they have the greatest risk of overexploitation (Bellwood and Choat 2011). Size of the parrotfish and density have been shown to be positively related to algal reduction rates, where they were up to 30% higher for the larger parrotfish species than the small ones per unit biomass (Jayewardene 2009). The success or failure of any no-take areas will depend on areas outside the park that are part of the same highly connected reef system (Howard *et al.* 2009).

Local management should focus on direct sources of mortality on the reef such as sedimentation, pollution and eutrophication, as well as the ecological processes being restored (Adam *et al.* 2015). This includes herbivory that is important for coral persistence and recovery from perturbations (Adam *et al.* 2015). Accumulation of parrotfish in marine reserves can increase the resilience of coral reef populations (Mumby and Steneck 2008). A complete ban on fishing parrotfish would not be politically viable or desirable economically, however, the trade-offs between sustainable fisheries and maintaining grazing on the reef can be achieved through spatial management of fishing (Adam *et al.* 2015). We can see the impact that the management of Atata reef is having, it is beneficial to maintain this management and identify changes difference over time and how the management is working. Marine Protected Area implementation, for example, at the Kolonga site could be beneficial due to its location to urban development and nursery, but this is a readily exploited area and is severely impacted by local stressors. Due to Kolonga's current state it may be too far into changing to an alternate state, where we cannot implement anything that will help to bring it back to its previous state. There would be a high level of investment needed and reinforcing this would be difficult due to its communal fishing history. All of these would influence the benefits that would actually be achieved by MPAs for this site (Guidetti *et al.* 2008).

It does not take a lot to damage a reef, overharvest from the top down, and add nutrients, then stressors of climate change and anthropogenic effects. It will be important to combine a series of management tools to the coral reefs such as community, government and scientific support. An example of this is water chemistry testing, this will be helpful in supporting management efforts by testing the water for any pollutants or levels of nutrients and their possible influence on reef health and macro algae growth. To have a hybrid of management between community-based and government management there needs to be a support for empowering customary management regimes (Aswani and Sabetian 2010). Reversing the fate of coral reefs will require the development of integrated management strategies that simultaneously address the multiple stressors on top of the impacts of fisheries on herbivore assemblages (Adam *et al*. 2015). There will have to be trade-off between the short term fisheries goals for maximising profit and yields, with the long term sustainability of coral reef ecosystems and their services (Brown and Mumby 2014). Overall with management the greatest chance of success is to act early before there is loss of these functionally important species and inevitably the coral reef and its health.

4.1 CONCLUSION

The findings of this research have contributed to the knowledge-base of parrotfish population dynamics in Tongatapu, the Kingdom of Tonga. The UVC method use has been validated through comparison of two methods for their ability to collect fish counts and behaviour on the in shore coral reefs. A novel stand method was then applied for the collection of parrotfish population data, this research helps to validate the use of stationary video UVC methods for reef fish research. By analysing the video data for diversity, abundance, harem structure and behaviour, I have been able to build a baseline dataset on parrotfish within the semi-enclosed lagoon of Tongatapu.

It is clear that coral reef fisheries are in the greatest need for protection when in close proximity to urban development. These effects include the anthropogenic and environmental effects, which we are likely to see more of as the Pacific island nations develop. The Tongan fisheries have undergone different management control over their marine resources with the change of constitution (Global Environment Facility 2013). Changing from community control to being available to everyone with no control, this led to sites being over exploited (Global Environment Facility 2013). A combination of traditional management complimented with scientific understanding and research may be best to manage parrotfish stocks (Thyresson *et al.* 2011). Temporal and spatial comparisons to the baseline data collected here, will be important to see changes in the inshore coral reefs of the semi-enclosed lagoon of Tongatapu. Ultimately, this will give a greater understanding of the environmental changes and their influences on the parrotfish species, and vice versa. For the coral reef ecosystem to be maintained there is urgency for greater management over the local fisheries and the connected ecosystems. Identifying which species to manage, whether they implement a no take zone, MPA or a harvesting period. Complimented with further investigation and potential protection for mangroves. Management on land is necessary, for example, involving any waste water and leaching that may occur, in this way the continued urban growth will not have a negative influence on the inshore coral reef ecosystems.

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6 APPENDICES

6.1 APPENDIX 1

The extraction kit Extract-N-Amp was used for DNA extraction. For each fish, 5mm^2 of muscle tissue was added to 25µl of Extraction buffer and 7.6µl of Tissue prep. The samples were then run in a thermocycler at 37^{o}C for 10 minutes and 97^{o}C for 3 minutes, and were finished with 25µl of Neutralising buffer.

For each extraction, a polymerase chain reaction (PCR) was used to amplify a 536bp fragment of the mitochondrial 16S and 570bp fragment of the Cytochrome Oxidase 1 (COI) gene, using the universal fish primer pairs 16Sas/16Sbr and FF1d/FF2d respectively. The PCR was done in a 10µl reaction; comprising, 7.5 µL of KAPA2G Robust HotStart ReadyMix (KAPA Biosystems, Boston, USA), 5.5 µL of Milli-Q water and 1.0 µL of DNA extract (c. 5 – 15ng/µl DNA). Negative controls, with Milli-Q water as the template were run with each batch of DNA extracts. Polymerase chain reaction profiles consisted of an initial denaturing period (94 °C, 5 min) followed by 40 cycles of denaturing (94°C, 45 s), annealing (48 °C, 45 s) and extension steps (72 °C, 60 s) finished off by a final extension (72 °C, 5 min).

Purified PCR products were sequenced in both directions by the University of Canterbury Sequencing service using an ABI300, and aligned in BioEdit Sequence Alignment Editor (version 7.2.5) (Hall 1999).

A BLAST search was completed for each sequence and the two GenBank sequencing with the greatest identity were chosen for further investigation. Collecting the two species, their accession numbers, identified proportios and percentages (%) (Table A1). These were confirmed with images taken at the time of the fin clipping.

Table A1. Results of the mitochondrial DNA 16S sequence for the parrotfish species. The fish ID code, species identification, GenBank accession number, identities proportion and percentage (%), are shown.

16s sequencing results				
Code	**Species**	**Accession no.**	**Identities proportion**	**Identities %**
B1	*Chlorurus spilurus*	JX026470.1	533/536	99
B1	*Chlorurus sordidus*	AP006567.1	533/536	99
B2	*Scarus niger*	JX026497.1	552/555	99
B2	*Scarus persicus*	JX026501.1	547/555	99

BF3	*Chlorurus spilurus*	JX026470.1	273/285	96
BF3	*Chlorurus sordidus*	AY081089.1	273/285	96
BF4	*Chlorurus spilurus*	JX026470.1	555/557	99
BF4	*Chlorurus sordidus*	AP006567.1	555/557	99
E54	*Chlorurus spilurus*	JX026470.1	555/557	99
E54	*Chlorurus sordidus*	AP006567.1	555/557	99
M2	*Chlorurus spilurus*	JX026470.1	555/557	99
M2	*Chlorurus sordidus*	AP006567.1	555/557	99
M6	*Chlorurus spilurus*	JX026470.1	555/557	99
M6	*Chlorurus sordidus*	AP006567.1|	555/557	99
BF5	*Chlorurus spilurus*	JX026470.1	482/487	99
BF5	*Chlorurus sordidus*	AY081089.1	482/487	99
BF7	*Chlorurus spilurus*	JX026470.1	555/556	99
BF7	*Chlorurus sordidus*	AP006567.1	555/556	99
E61	*Chlorurus spilurus*	JX026470.1	555/556	99
E61	*Chlorurus sordidus*	AP006567.1	555/556	99
E62	*Chlorurus spilurus*	JX026470.1	555/556	99
E62	*Chlorurus sordidus*	AP006567.1	555/556	99
E71	*Chlorurus spilurus*	JX026470.1	555/556	99
E71	*Chlorurus sordidus*	AP006567.1	555/556	99
E74	*Chlorurus spilurus*	JX026470.1	555/556	99
E74	*Chlorurus sordidus*	AP006567.1	554/556	99
BF8	*Chlorurus spilurus*	JX026470.1	554/557	99
BF8	*Chlorurus sordidus*	AP006567.1	554/557	99
BM	*Chlorurus microrhinos*	JX026466.1	554/557	99
BM	*Chlorurus strongylocephalus*	JX026472.1	554/557	99
E1	*Chlorurus spilurus*	JX026470.1	550/552	99
E1	*Chlorurus sordidus*	AP006567.1	550/552	99
E10	*Scarus globiceps*	JX026492.1	554/556	99
E10	*Scarus rivulatus*	JX026505.1	553/556	99
E21	*Scarus globiceps*	JX026492.1	554/556	99
E21	*Scarus rivulatus*	JX026505.1	553/556	99
E11	*Scarus spinus*	JX026513.1	555/556	99
E11	*Scarus persicus*	JX026501.1	546/556	98
E17	*Scarus spinus*	JX026513.1	555/556	99
E17	*Scarus persicus*	JX026501.1	546/556	98
E9	*Scarus spinus*	JX026513.1	555/556	99
E9	*Scarus persicus*	JX026501.1	546/556	98
E12	*Scarus spinus*	JX026513.1	553/556	99
E12	*Scarus persicus*	EU601255.1	528/529	99
E13	*Scarus globiceps*	JX026492.1	555/556	99
E13	*Scarus rivulatus*	JX026505.1	554/556	99
E14	*Scarus globiceps*	JX026492.1	555/556	99
E14	*Scarus rivulatus*	JX026505.1	554/556	99
E15	*Scarus globiceps*	JX026492.1	555/556	99
E15	*Scarus rivulatus*	JX026505.1	554/556	99
E16	*Scarus globiceps*	JX026492.1	555/556	99

E16	*Scarus rivulatus*	JX026505.1	554/556	99
E26	*Scarus globiceps*	JX026492.1	555/556	99
E26	*Scarus rivulatus*	JX026505.1	554/556	99
E30	*Scarus globiceps*	JX026492.1	555/556	99
E30	*Scarus rivulatus*	JX026505.1	554/556	99
E31	*Scarus globiceps*	JX026492.1	555/556	99
E31	*Scarus rivulatus*	JX026505.1	554/556	99
E34	*Scarus globiceps*	JX026492.1	555/556	99
E34	*Scarus rivulatus*	JX026505.1	554/556	99
E38	*Scarus globiceps*	JX026492.1	555/556	99
E38	*Scarus rivulatus*	JX026505.1	554/556	99
M11	*Scarus globiceps*	JX026492.1	555/556	99
M11	*Scarus rivulatus*	JX026505.1	554/556	99
M5	*Scarus globiceps*	JX026492.1	555/556	99
M5	*Scarus rivulatus*	JX026505.1	554/556	99
Scrib2	*Scarus globiceps*	JX026492.1	555/556	99
Scrib2	*Scarus rivulatus*	JX026505.1	554/556	99
E19	*Scarus globiceps*	JX026492.1	554/556	99
E19	*Scarus rivulatus*	JX026505.1	553/556	99
E18	*Cetoscarus bicolor*	JX026454.1	551/554	99
E18	*Cetoscarus bicolor*	AY081088.1	526/528	99
E1M1	*Chlorurus microrhinos*	JX026466.1	555/557	99
E1M1	*Chlorurus strongylocephalus*	JX026472.1	554/557	99
F1	*Chlorurus microrhinos*	JX026466.1	555/557	99
F1	*Chlorurus strongylocephalus*	JX026472.1	554/557	99
E2	*Scarus schlegeli*	JX026512.1	558/559	99
E2	*Scarus schlegeli*	FJ595020.1	558/559	99
E36	*Scarus schlegeli*	JX026512.1	556/557	99
E36	*Scarus schlegeli*	FJ595020.1	556/557	99
E5	*Scarus schlegeli*	JX026512.1	556/557	99
E5	*Scarus schlegeli*	FJ595020.1	556/557	99
E20	*Scarus schlegeli*	JX026512.1	555/558	99
E20	*Scarus schlegeli*	FJ595020.1	555/558	99
E22	*Chlorurus microrhinos*	JX026466.1	551/557	99
E22	*Chlorurus strongylocephalus*	JX026472.1	550/557	99
E23	*Chlorurus microrhinos*	JX026466.1	556/557	99
E23	*Chlorurus strongylocephalus*	JX026472.1	555/557	99
Ee21	*Chlorurus microrhinos*	JX026466.1	556/557	99
Ee21	*Chlorurus strongylocephalus*	JX026472.1	555/557	99
F2	*Chlorurus microrhinos*	JX026466.1	556/557	99
F2	*Chlorurus strongylocephalus*	JX026472.1	555/557	99
F3	*Chlorurus microrhinos*	JX026466.1	556/557	99
F3	*Chlorurus strongylocephalus*	JX026472.1	555/557	99
M18	*Chlorurus microrhinos*	JX026466.1	556/557	99
M18	*Chlorurus strongylocephalus*	JX026472.1	555/557	99
E24	*Chlorurus microrhinos*	JX026466.1	555/557	99
E24	*Chlorurus strongylocephalus*	JX026472.1	554/557	99
E25	*Scarus globiceps*	JX026492.1	554/556	99
E25	*Scarus rivulatus*	JX026505.1	553/556	99

E28	*Scarus compressus*	JX026478.1	553/556	99
E28	*Scarus aff. Ghobban*	JX026489.1	552/556	99
E29	*Cetoscarus bicolor*	JX026454.1	552/554	99
E29	*Cetoscarus bicolor*	AY081088.1	527/528	99
E3	*Scarus schlegeli*	JX026512.1	555/557	99
E3	*Scarus schlegeli*	FJ595020.1	555/557	99
E6	*Scarus schlegeli*	JX026512.1	555/557	99
E6	*Scarus schlegeli*	FJ595020.1	555/557	99
M9	*Scarus schlegeli*	JX026512.1	555/557	99
M9	*Scarus schlegeli*	FJ595020.1	555/557	99
E32	*Scarus compressus*	JX026478.1	555/556	99
E32	*Scarus aff. Ghobban*	JX026489.1	554/556	99
E35	*Scarus schlegeli*	JX026512.1	555/557	99
E35	*Scarus schlegeli*	FJ595020.1	555/557	99
E37	*Scarus compressus*	JX026478.1	553/557	99
E37	*Scarus aff. Ghobban*	JX026489.1	552/557	99
E39	*Scarus aff. Ghobban*	JX026489.1	555/556	99
E39	*Scarus compressus*	JX026478.1	554/556	99
E4	*Scarus schlegeli*	JX026512.1	555/557	99
E4	*Scarus schlegeli*	FJ595020.1	555/557	99
E40	*Hipposcarus longiceps*	JX026456.1	553/558	99
E40	*Hipposcarus harid*	JX026455.1	546/558	98
E41	*Scarus psittacus*	JX026503.1	556/557	99
E41	*Scarus collana*	JX026477.1	546/558	98
M4	*Scarus psittacus*	JX026503.1	556/557	99
M4	*Scarus collana*	JX026477.1	546/558	98
Scrib1	*Scarus psittacus*	JX026503.1	556/557	99
Scrib1	*Scarus collana*	JX026477.1	546/558	98
E51	*Chlorurus spilurus*	JX026470.1	553/557	99
E51	*Chlorurus sordidus*	AP006567.1	553/557	99
E53	*Chlorurus spilurus*	JX026470.1	554/557	99
E53	*Chlorurus sordidus*	AP006567.1	554/557	99
E63	*Chlorurus spilurus*	JX026470.1	554/556	99
E63	*Chlorurus sordidus*	AP006567.1	554/556	99
E7	*Scarus globiceps*	JX026492.1	554/556	99
E7	*Scarus rivulatus*	JX026505.1	553/556	99
E72	*Chlorurus spilurus*	JX026470.1	554/556	99
E72	*Chlorurus sordidus*	AP006567.1	554/556	99
E73	*Chlorurus spilurus*	JX026470.1	554/556	99
E73	*Chlorurus sordidus*	AP006567.1	554/556	99
E8	*Chlorurus spilurus*	JX026470.1	555/557	99
E8	*Chlorurus sordidus*	AP006567.1	555/557	99
Ee31	*Scarus globiceps*	JX026492.1	551/556	99
Ee31	*Scarus rivulatus*	JX026505.1	554/556	99
Ee32	*Chlorurus spilurus*	JX026470.1	554/556	99
Ee32	*Chlorurus sordidus*	AP006567.1	554/556	99
Ee33	*Chlorurus spilurus*	JX026470.1	554/558	99
Ee33	*Chlorurus sordidus*	AP006567.1	554/558	99
Ee34	*Chlorurus spilurus*	JX026470.1	552/557	99
Ee34	*Chlorurus sordidus*	AP006567.1	552/557	99

Ee35	*Chlorurus microrhinos*	JX026466.1	552/558	99
Ee35	*Chlorurus strongylocephalus*	JX026472.1	551/558	99
F4	*Chlorurus spilurus*	JX026470.1	553/558	99
F4	*Chlorurus sordidus*	AP006567.1	553/558	99
M1	*Chlorurus spilurus*	JX026470.1	554/556	99
M1	*Chlorurus sordidus*	AP006567.1	554/556	99
M10	*Chlorurus spilurus*	JX026470.1	553/556	99
M10	*Chlorurus sordidus*	AP006567.1	553/556	99
M12	*Scarus frenatus*	JX026486.1	550/552	99
M12	*Scarus aff. Rubroviolaceus*	JX026509.1	550/556	99
M13	*Scarus frenatus*	JX026486.1	549/552	99
M13	*Scarus aff. Rubroviolaceus*	JX026509.1	549/556	99
M14	*Scarus globiceps*	JX026492.1	555/557	99
M14	*Scarus rivulatus*	JX026505.1	554/557	99
M15	*Calotomus carolinus*	AY081092.1	521/524	99
M15	*Calotomus spinidens*	EU601228.1	501/527	95
M17	*Scarus festivus*	JX026483.1	552/556	99
M17	*Scarus chameleon*	JX026475.1	539/542	99
M19	*Chlorurus microrhinos*	JX026466.1	555/557	99
M19	*Chlorurus strongylocephalus*	JX026472.1	554/557	99
M20	*Calotomus carolinus*	AY081092.1	521/523	99
M20	*Calotomus spinidens*	EU601228.1	501/526	95
M3	*Chlorurus spilurus*	JX026470.1	554/556	99
M3	*Chlorurus sordidus*	AP006567.1	554/556	99
M7	*Scarus psittacus*	JX026503.1	554/557	99
M7	*Scarus collana*	JX026477.1	544/558	97
M8	*Chlorurus spilurus*	JX026470.1	554/556	99
M8	*Chlorurus sordidus*	AP006567.1	554/556	99
BF6	*Chlorurus spilurus*	JX026470.1	225/230	98
BF6	*Chlorurus sordidus*	AP006567.1	225/230	98
BF1	*Chlorurus spilurus*	JX026470.1	503/506	99
BF1	*Chlorurus sordidus*	AP006567.1	503/506	99

The Shapiro-Wilk tests for normality completed on the parrotfish abundance data were all significant, both with (square root p= 0.002, exponential p=2.2E-16) and without transformations (p=7.757E-14). Even with these results the plots of residuals and quantiles were close to normal (Figure A1).

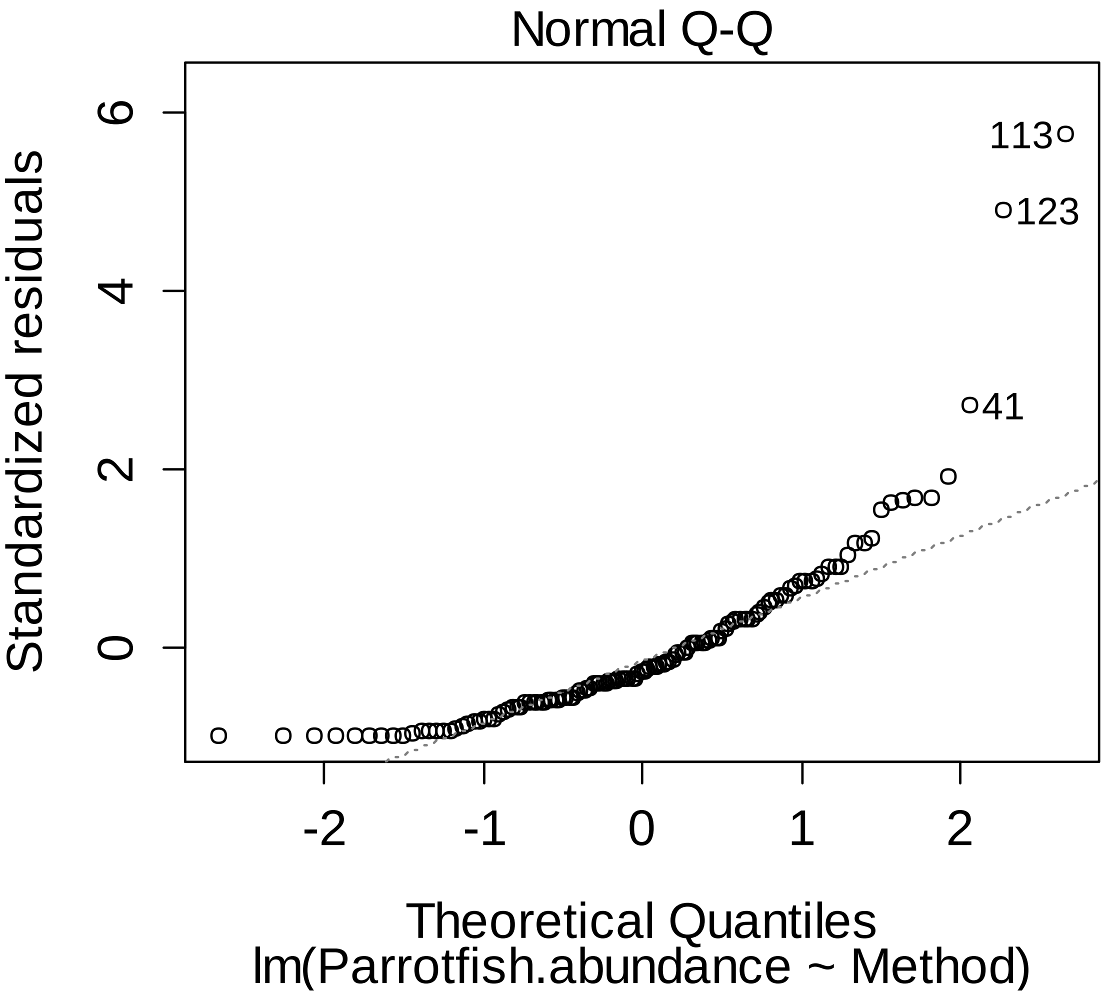

Figure A1. Plot of standardized residuals against Theoretical Quantiles for parrotfish abundance and method.

A Shapiro-Wilk test for normality was completed on the damselfish abundance data, it was significant without a transformation (p=5.059E-08), meeting the assumption of normality when a square root transformation was applied (p=0.1528). It meet the assumptions of homogeneity of variances through both the Bartlett test (p=0.933) and the Levene test (p=0.661).

6.2 APPENDIX 2

The overall number of parrotfish species identified over the sample sites of the semi-enclosed lagoon of Tongatapu in this research was 14. The number frequency seen for each sampling year and site varied (Table A2, Table A3, Table A4).

Table A2. Parrotfish species identified and their frequency across the three sites sampled in 2012.

Species	**Frequency per site**		
	Atata	Kolonga	Sopu
Chlorurus spilurus	5	1	1
Scarus rivulatus	0	0	2
Scarus oviceps	6	0	0
Scarus dimidatus	8	1	0
Scarus psittacus	2	0	0
Scarus niger	1	0	0
Scarus schegeli	0	0	1

Table A3. Parrotfish species identified and their frequency across the four sites sampled in 2014.

Species	**Frequency per site**			
	Atata	Kolonga	Sopu	Tukutonga
Chlorurus spilurus	1	1	1	1
Scarus frenatus	1	0	1	0
Scarus dimidatus	9	0	0	3
Scarus schegeli	0	0	0	1
Scarus chameleon	2	0	0	0

Table A4. Parrotfish species identified and their frequency across the five sites sampled in 2015.

Species	**Frequency per site**				
	Atata	Kolonga	Sopu	Tukutonga	Offshore
Chlorurus spilurus	22	4	8	4	21
Scarus frenatus	5	0	1	0	0
Scarus rivulatus	5	0	0	0	2
Scarus globiceps	2	2	0	1	2
Scarus spinus	1	0	0	0	0
Scarus oviceps	1	1	0	0	1
Scarus dimidatus	5	0	0	0	1
Scarus psittacus	1	0	2	0	0
Scarus niger	0	0	0	0	1
Chlorurus microrhinos	1	0	0	0	2
Scarus ghobban	0	0	1	0	0
Chlorurus strongylocephalus	0	0	0	1	0
Scarus schegeli	3	0	1	0	9