ASSESSING THE FISHERIES BENEFITS OF FOUR COMMUNITY–BASED NO-TAKE MARINE PROTECTED AREAS IN FIJI

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ABSTRACT

Locally managed marine areas (LMMA) are increasingly being implemented in many Pacific Island communities to address concerns regarding management of coastal fisheries. Widespread advocacy for these management regimes lies in their ability to build on existing community strengths in traditional knowledge, customary tenure, and governance; as well as promote a local awareness of resource stewardship. In Fiji, the prohibition of marine resource harvest, or *tabu*, is one traditional practice that has been commonly adapted to serve as a form of no-take Marine Protected Area (MPA). While completely no-take MPAs have gained recognition around the world as fisheries management tools, their sole efficacy for enhancing stocks of important fisheries species is still considered highly dependent on varying environmental and socioeconomic circumstances.

This study examined the effects of small no-take MPAs, or *tabu*, on the abundance, biomass, size, and sexually mature proportion of target reef fish species in four villages, across two LMMAs along the Coral Coast of Viti Levu, Fiji. Of the four village *tabu* areas, three have maintained protected status for 5-7 years, while the other was recently reopened to fishing within five months of the commencement of this study. Fish angling surveys revealed that all four factors examined were, in many cases, significantly ($P > 0.05$) greater for target species caught within the active MPA sites, with the converse effect occurring within the site recently opened to fishing. The combination of these findings supports the utility of no-take *tabu* areas as LMMA tools for enhancing stocks of certain target species, and also suggests that even temporary closures may be insufficient for conserving stocks. With increasing anthropogenic pressures and environmental uncertainties; as well as the limited knowledge regarding reserve population dynamics, permanent closures may be the most appropriate method of ensuring the sustainability of these small-scale fisheries.

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1.1 Introduction

An array of challenges threaten the well-being of the world's oceans, including overexploitation of living marine resources, habitat degradation and destruction, pollution, and climate change impacts (Harley *et al.* 2006; UNEP, 2006; Halpern *et al.* 2008; Jackson, 2008; Coker *et al.* 2009; Lester *et al.* 2009). A major consensus amid these threats is the need to reduce the intense pressure and destructive power that modern fishing practices inflict on the world's oceans and subsequent fisheries (e.g. FAO, 2002; Browman and Stergiou, 2004). Fisheries worldwide have been affected; with 52% being fully exploited, yielding catch at or close to their maximum sustainable limit, and a further 25% being either overexploited, depleted, or recovering from depletion (FAO, 2006). Poor fishing regulation, or the lack thereof, has the potential to lead to the depletion of economically and culturally important species, decreases in the abundance of rare species, habitat loss, altered community structure, and compromised eco-system functioning and delivery of ecosystem services (Lester *et al.* 2009). In turn, such factors have contributed to socioeconomic issues including fishery collapse, economic inefficiency, and loss of employment (Jennings *et al.* 2001).

The increasingly rapid and drastic degradation of the world's marine resources, through practices like overfishing, has initiated increasing calls for more effective approaches to protect, maintain, and restore marine ecosystems (Allison *et al.* 1998; Murray *et al.* 1999; NRC 1999, 2000; Lubchenco *et al.* 2003). However, regulation of fishing in complex ecosystems, such as coral reefs, presents exceptional difficulties due to the diversity of species and the complexity of interactions (McClanahan, 1995); which may compromise ecosystem stability and increase vulnerability to overfishing (Koslow, *et al.* 1988). Coral reefs hold tremendous ecological and economic value because of their different functions such as: recreational (e.g. tourism) and cultural value, support for biodiversity, coastal protection (ie. prevent erosion), and as a critical food source. Research previously carried out on both pristine and heavily fished reefs suggest that all of these functions can be affected by heavy fishing. (McClanahan and Muthiga, 1988, 1989; Russ and Alcala, 1989; McClanahan and Shafir, 1990; Russ, 1991; McClanahan, 1995). Throughout the tropics, particularly in developing countries, coral reefs support artisanal fisheries with an estimated annual yield of 6 million tonnes (Munro, 1996). While the perception that artisanal fishing has little impact on reefs has been widespread in the past (Hawkins and Roberts, 2004), concern has been growing (Russ, 2002) in the wake of increased fishing intensity to support growing populations (Polunin and Roberts, 1996).

1.2 Effects of Fishing

Of all resources available on tropical coral reefs, fish are the most widely exploited (Jennings and Polunin, 1996*a*). Consequently, numerous reef ecosystems are subject to both direct and indirect effects of fishing (Russ, 1991). Research carried out on coral reef fisheries frequently suggests the most obvious detectable effects of fishing pressure include declines in abundance or biomass of selectively targeted species (Russ, 1985; Russ and Alcala, 1989; Jennings and Lock, 1996; Watson *et al.* 2009). Numerous studies have documented such effects (Bohnsack, 1982; Ayling and Ayling 1986*b*; Polovina, 1986; Alcala, 1988; Samoilys, 1988; Roberts and Polunin, 1992; Roberts and Polunin, 1993; McClanahan, 1994; Watson and Ormond, 1994; Watson *et al.* 2009). Particularly vulnerable species including game fish and larger-bodied fish may no longer be found on reefs subject to fishing pressure (Bohnsack, 1982; Goeden, 1982), and in some cases, become threatened with local or global extinction (Roberts and Hawkins, 1999; Hawkins and Roberts, 2004).

Mortality of reef fish is usually reflected rapidly in fisheries; due to their site attached nature and subsequent limited replacement capabilities via adult immigration (Munro and Williams, 1985). Initially, declines in catch per unit effort (CPUE) are exhibited in target species (Smith and Dalzell, 1993; Appledoorn *et al.* 1992), and may be followed by a decline in total catch if exploitation remains high (Gaut and Munro, 1983; Koslow *et al.* 1988). Similarly, spatial comparisons of areas subject to differing levels of fishing intensity indicate CPUE is higher at sites of higher fishing intensity (Wass, 1982; Kawaguchi, 1974; Lock, 1986*a*).

Various techniques used by fishers often target specific size ranges of particular fish species. Accordingly, populations subject to fishing may experience changes in size and age structure (Jennings and Lock, 1996). Both size and age are important indices for determining when a fish reaches maturity; which further influences population model estimates of sustainable harvest rates (Clark, 1991), and is an important predictor of the risk of overexploitation (Reynolds *et al.* 2005). Decreases in size have been attributed to fishing in cases following the opening of a reserve area or the development of a new fishery (Jennings and Lock, 1996). Comparisons of neighboring sites subject to varying levels of fishing pressure have also demonstrated decreases in size in response to increased pressure (Bell, 1983; Russ, 1985; Ayling and Ayling, 1986*a*; Ferry and Kohler, 1987; Beinssen, 1988, 1989; McClanahan and Muthiga, 1988; Samoilys, 1988; Buxton and Smale, 1989; Polunin and Roberts, 1993; Watson and Ormond, 1994). Fishing that is size selective may also affect the genetic structure of fish populations, leading to decreases in heterozygosity (Smith *et al.* 1991) and/or growth rates (Law and Grey, 1989; Sutherland, 1990; Law, 1991).

The effects of fishing are not limited to target species, but may encompass the overall reef fish community structure (Jennings and Lock, 1996). Not only catch rates (Munro, 1983; Koslow *et al.* 1988), but species richness (Russ and Alcala, 1989; McClanahan, 1994) may decline as well when subjected to intense fishing pressure. Changes invoked by gear selection or fish preference may include shifts in biomass or size-specific biomass of certain trophic groups or shifts in relative dominance of populations with specific life history stages (Adams, 1980). It has even been suggested that in some cases, the removal of just 5% of fish biomass could significantly alter the structure of reef fish communities due to depletion of preferential targeted species (Jennings and Polunin, 1996*a*). That said, relatively few studies have been undertaken to document changes in multispecies communities in response to fishing (Russ, 1985; Lock, 1986*b*; Koslow *et al.* 1988; Russ and Alcala, 1989).

Of the various fishing techniques employed, some will inflict direct impacts upon reef habitats (Jennings and Lock, 1996). Poisons can potentially cause chemical damage to corals and non-target fishes or invertebrates (Rubec, 1986; Eldredge, 1987; McAllister, 1988; Pyle, 1993); while physical damage to coral may be inflicted by activities such as drive netting (Carpenter and Alcala, 1977; Gomez *et al.* 1987), trapping (Munro *et al.*

1987) and explosives (Alcala and Gomez, 1987; Munro *et al.* 1987; Saila *et al.* 1993). Given that abundance of reef fishes has been shown to be positively correlated with habitat complexity, (Risk, 1972; de Boer, 1978; Luckhurst and Luckhurst, 1978; Carpenter *et al.* 1981; Thresher 1983; Kaufman and Ebersole, 1984; Patton *et al.* 1985; Roberts and Ormond, 1987; Grigg, 1994; Jennings and Polunin, 1995*c*; Beukers and Jones, 1997; Gratwicke and Speight, 2005; Coker *et al.* 2009), habitat degradation has the ability to affect fish yield by causing changes in the distribution of exploitable fish biomass and, in severe cases, by reducing the total productivity of the fishery (Jennings and Lock, 1996).

1.3 Conventional Fisheries Management: Implications for Fiji

The need for regulation of fishing practices has led to the establishment of fisheries management regimes, each with a wide variety of biological, economic, social, and political objectives (Clark, 1985). To accomplish fisheries objectives, managers have taken a range of actions, which can be broadly classified as catch controls, effort controls, and technical measures. Catch controls, also known as output controls, are among the most widely used management regulations. Their aim is to control fishing mortality by limiting the weight or amount of the catch that fishers can take. These include total allowable catch (TAC) or quotas (Q), which are limits to be taken from a specified stock, as well as individual quota (IQ) and vessel catch limits, where the TAC is divided between fishing units. Effort controls are a form of input control that limit the number of fishers or boats who work in a fishery, the amount, size and type of gear they can use, the duration the gear can remain in the water; as well as limits on the size or power of vessels (Jennings *et al.* 2001).

In many cases, conventional fisheries management tools have been unsuccessful in generating sustainable fisheries catches (Roberts and Polunin, 1991; Munro, 1996; Rodwell *et al.* 2002). The sustainable use of tropical marine resources is one of the most critical issues required to address the conservation of species diversity and ecosystem services. (McClanahan, 1995) This is especially problematic in tropical developing countries where exploitation is often for subsistence purposes and limited resources are available for management (Rodwell *et al.* 2002). Such is the case in the South Pacific region, where it has been estimated that 80% of the catch, an estimated 80,000 tons/yr, worth $160,300,000USD; from inshore fisheries is taken for subsistence purposes (Dalzell *et al.* 1996) Furthermore, coastal reef finfish are expected to remain the primary source of subsidence protein in most Pacific Island countries for the perceivable future (Dalzell and Adams, 1994; Kuster *et al.* 2006). Despite such assertions, low priority given to subsistence fisheries in the region has amounted to little documentation regarding both user patterns and status of inshore stocks (Dalzell *et al.* 1996; Zann and Vuki, 2000; Kuster *et al.* 2006), inhibiting proper application of fisheries management measures.

In Fiji, inadequate financial and human resources have continued to impede the assessment and efficient management of coral reef fisheries. Challenges, including lack of long term monitoring (Hand *et al.* 2005; Vuki *et al.* 2000) coupled with a shortage of technical expertise (Vuki *et al.* 2000), are the result of limited national priority; even though subsistence fisheries constitute the largest contributing sector to fisheries, with production at 50% of the total (Gillet and Lightfoot, 2002). Additionally, when combined with artisanal fisheries (inshore reef fisheries), they make up 60 to 85% of Fiji's total catch. Such figures help stress the critical role of coral reefs in maintaining Fiji's fisheries and ultimately, food security (Teh *et al.* 2009).

1.4 Fiji's Artisanal and Subsistence Reef Fisheries

Subsistence and artisanal (small scale commercial) fishing constitute the dominant spheres of influence within Fiji's coral reef fisheries. Subsistence fishing is defined as fishing mainly to supply one's own food requirements, with the sale of fish occurring infrequently and only in cases of surplus catch. Conversely, artisanal fishing is done primarily for sale; often with a small portion of the catch being kept for personal consumption (Teh *et al.* 2009). The distinction between the two has been progressively blurred with increasing monetisation of subsistence fishing, particularly on the larger, more populated islands; with significant proportions of subsistence catches being supplied to municipal markets, hotels, supermarkets, fish shops, and other outlets (Hand *et al.* 2005).

Several studies have attempted to try and quantify the economic contribution of Fiji's subsistence and artisanal fishing sectors. In 2003, Fiji's Department of Fisheries (DOF) estimated the total value of artisanal catch at F$27.6 million, while the Asian Development Bank (ADB) estimated the total value of subsistence catch to be equivalent to F$48.6 million (Hand *et al.* 2005). Additionally, the DOF's estimate of the subsistence fishery's economic value based on 'value to consumer' totaled US$7.2 million in 2003. Another study by Dalzell *et al.* (1996) valued 16,600 metric tons of subsistence fisheries production at US$45.8 million, with a total value of US$64 million when combined with artisanal fisheries. The actual economic significance of these fisheries is probably significantly underestimated, given the limited resources available for government monitoring (Teh *et al.* 2009). Indeed, re-estimation of these values has suggested a contribution 34% higher than that calculated by the DOF (Gillett and Lightfoot, 2002).

Fiji maintains a customary marine tenure system (CMT), built on local autonomy and self reliance (South and Veitayaki, 1998). The CMT system divides the inshore fishing grounds into 410 registered customary rights fishing areas, known as *iqoliqoli*, which range from one to 5,000 km² in area (Cooke and Moce, 1995). These *iqoliqoli* are an extremely important component of the institutional setting of Fiji's coral reef fisheries (Hand *et al.* 2005). They are generally located from the watershed, extending to the seaward slope of the reef (Mühlig-Hofmann, 2006). To fish a specific *iqoliqoli*, fishers must obtain a license from the Department of Fisheries, which requires the permission of the customary fishing right owners; including the district chief (Mühlig-Hofmann, 2006).

From 1981–2005, the number of registered fishing vessels operating in Fiji's inshore artisanal fishery reached their highest in 1989, with 2112 registered vessels (DOF, 1995); followed by a decline that reached its lowest in 2004 with 727 (DOF, 2004). In 2003, it was estimated that Fiji's fisheries sector provided employment for 2,137 artisanal inshore fishers. It was also estimated to provide employment for 3,000 subsistence fishers (Hand *et al.* 2005), but, in reality, this number is largely unknown (Teh *et al.* 2009). The Food and Agriculture Organisation (FAO, 2009) of the United Nations estimated that there are 30,000 subsistence fishers in Fiji, although the method by which this number was calculated was not documented. In addition, the FAO reported that half of all rural households participate in some form of subsistence fishing. If applied, this approach estimates there are approximately 43,000 subsistence fishers in Fiji (Teh *et al.* 2009).

Between 1981 and 1995, the annual number of crew accounted for by the Department of Fisheries averaged 3,964 fishers. This number fell to an average of 1,888 between 2001 and 2004. However, these numbers only reflect the number of fishers who obtain a license, neglecting those who bypass the fisheries department altogether by obtaining permission to fish directly from the village chief, or those fishers who fish within their own *iqoliqoli* (Veitayaki *et al.* undated). Indeed, it has been reported that many fishers are without a fishing license due to the high cost, and beaucratic red tape involved in obtaining one (Teh *et al.* 2009).

Fishing inshore usually occurs over shallow coral reef and reef associated habitats, using small non-powered or outboard power canoes and dinghies. For communities on outer islands, fishing occurs almost exclusively over coral reefs, while lagoons and fringing reefs are the main habitats utilised throughout coastal villages on Viti Levu. Common fishing methods used for targeting reef fish include the use of hand line, spear, gillnet, seine net, reef gleaning, and diving with surface supplied air, known as hookah (Rawlinson *et al.* 1995). Small-scale pelagic fishing is also carried out via trolling along the reef edge, and by pole and line. Additionally, less widespread, illegal methods still occur such as dynamite fishing and the use of *duva*, a traditional method that utilizes plant derived poison to stun fish (Vuki *et al.* 2000). Numerous individual site studies of subsistence and artisanal fishing activities are available for both remote and more populated regions in Fiji (Turner *et al.* 2007; Rawlinson *et al.* 1995; Veitayaki *et al.* undated; O'Garra, 2007; Jennings and Polunin, 1996*b*; van Beaukering *et al.* 2007; Kuster *et al.* 2005; Yeeting *et al.* 2001).

Numerous coral reef fish species including herbivorous, carnivorous and omnivorous families make up the primary food fish catch in Fiji (Richards *et al.* 1994). Reef fish catches are comprised primarily of large predatory fish such as *Lethrinidae*, *Serranidae*, and *Scrombridae*; making up 18%, 13%, and 19% of the catch, respectively (Rawlinson *et al.* 1995, Dalzell *et al.* 1996). On Kadavu's Great Astrolabe Reef, Jennings (1998) reported that over 68% of the fisheries yield consisted of partly piscivorous fishes, predominately groupers (*Epinephelidae*) and emperors (*Lethrinidae*). Likewise, groupers and emperors were among the top five most frequently reported target species from fringing reefs around Viti Levu (Rawlinson *et al.* 1995; Dalzell *et al.* 1996), along with tuna and mackerels (*Scrombidae*), mullet (*Mugilidae*), and snappers (*Lutjanidae*). For the live reef fish food trade, important target species include the coral grouper (*Plectropomus areolatus*, *P. leopardus*, and *P. laevis*), other groupers (*Epinephelus* spp.) and the now protected humphead wrasse (*Cheilinus undulatus*, *Labridae*). Key species targeted for the ornamental fish trade include butterflyfish (*Chaetodonidae*), damselfish (*Pomacentridae*), and angelfish (*Pomacanthidae*). Pelagic fish often caught along the reef edge during trolling include species such as Spanish mackerel (*Scomberomorus commerson*), dogtooth tuna (*Scrombidae*), rainbow runners (*Elagatis bipinnulata*, *Carangidae*) and queenfish (*Scomberoides* spp.). An in depth summary of Fiji's catch composition can be found in Dalzell *et al.* (1996).

1.5 Socio-economic Aspects of Fiji's Coral Reef Fisheries

Heavy reliance on marine resources for subsistence and livelihoods has always been an integral part of Fijian society (Teh *et al.* 2009). Fish is an important source of nutrition in many villages, with daily consumption among indigenous Fjian housholds estimated to be 23.4% (FAO, 2009). Depending on location, fish consumption can range from 187g person $^{-1}$ day $^{-1}$ in urban areas (Rawlinson *et al.* 1995) to 280-470g person $^{-1}$ day $^{-1}$ on smaller outer islands. Annual fish consumption is estimated to range from 40kg capita $^{-1}$ year $^{-1}$ to 56kg capita $^{-1}$ year $^{-1}$, with 46% being supplied by the subsistence fishery (Gillet and Lightfoot, 2002).

Fishing is common activity in many households throughout Fiji. Like fish consumption, the degree of fishing participation varies with location. Rawlinson *et al.* (1995) found that approximately 91% of households in coastal villages along Viti Levu were involved in fishing, with 35% doing so for subsistence, while 64% fished mainly for artisanal purposes. Likewise, on the outer islands, 40% reported to fish for subsistence, with a further 57% doing so for artisanal purposes. Jennings and Polunin (1996*b*) found that 12.5–50% of total catch was sold among households in the outer islands, while 54% was sold as marine products in coastal villages on Viti Levu (Rawlinson *et al.* 1995) .Conversely, the majority of fishers in some remote areas in the Bua Province of Vanua Levu (Yeeting *et al.* 2001) and Lau Islands (Turner *et al.* 2007) reportedly fished primarily for subsistence.

Even with high levels of participation, fishing still has variable influence among Fijian households. Interviews carried out with villagers in the Lau Islands found that fishing was considered of low importance compared to other occupations such as farming and *tanoa* carving, despite 80-100% of all households being involved in fishing (Turner *et al.* 2007). However, a study conducted in northeast Macuata found that fishing supplied households with 59% of their income (Veityaki *et al.* undated). While access to markets to sell fish will influence its importance as an economic priority, the quick financial return fishing provides in comparison to planting crops has led to its practice in most households in rural coastal areas (Teh *et al.* 2009).

One of the most critical gaps in knowledge regarding socio-economic impacts on Fiji's fisheries is the role of women in these fisheries. This is true of fisheries assessments throughout the Pacific islands, even as women dominate in the subsistence fishing sector. A 1995 study by Rawlinson *et al.* found that over 50% of the subsistence catch in rural areas of Viti Levu, Fiji, were taken by women. Additionally, increasing monetary pressures exerted on the subsistence sector have strengthened women's involvement in the local artisanal fishing sector; more so than in many other parts of the world (Adams, 1998; Vunisea, 2005). Despite such assertions, official statistics and documentation fail to recognize their contributions. Both the acquisition of vast amounts knowledge held by women, and the ability to provide developmental assistance or external information to them, are greatly inhibited by societal norms and commonly male dominated fisheries departments (Adams, 1998). Acknowledging and incorporating further knowledge of the trends underway among fisher women throughout Fiji will be crucial to the success of future coral reef fishery management; especially those which are community-based.

1.6 Exploitation of Fiji's Coral Reef Fisheries

Today, the coral reef fisheries of Fiji are intensively exploited to meet the subsistence needs of a growing population and commercial demands of expanding coral reef associated trades (Teh *et al.* 2009). Intense fishing pressure has been prevalent near major population centers since the early to mid 1980's (Lal, 1984; Lewis, 1985). The demands of resource owners in an increasingly monetary goods and services oriented economy have intensified exploitation; even on reefs removed from population centers, capable of high finfish yield (Jennings and Polunin, 1995*b*), and generally regarded as being not overfished (Jennings and Polunin, 1996*b*; Kuster *et al.* 2005). These attributes are compounded by the continued acquisition of modern fishing methods and technologies (Matthews *et al.* 1998). High fishing pressure as well as low economic yield have been suggested for observed declines in artisanal catch from 1996-2002. Similar figures for subsistence fisheries are less easily quantified, given spatial and temporal variations in sale and consumption (Raj and Evans, undated).

There are 385 marine and 25 freshwater *iqoliqoli* in Fiji. The resources from these provide livelihoods for approximately 300,000 people living in coastal villages (Aalbersberg *et al.* 2005). Of the 410 recognized *iqoliqoli*, roughly 70 are over-exploited; while 250 are fully exploited (Hand *et al.* 2005). Consequently, resource owners throughout Fiji have acknowledged the depletion of certain marine resources (Johannes, 2002; Naqasima-Sobey and Vuki, 2002). Local depletion of species such as mullet (Mullidae) rabbitfish (Signidae), coral grouper, and bumphead parrotfish (*Bolbometapon muricatum*) have been recorded since the early 1990's (Richards *et al.* 1994). Simultaneously, catches of emperor were declining and stocks possibly over exploited in the late 1980's and early 1990's (Teh *at al.* 2009). More recently, the live fish food trade has increased pressure on vulnerable grouper stocks and humphead wrasse (Sadovy *et al.* 2003*a*, *b*). Fijian communities have faced increased socio-economic hardships due to declines; including diet changes and loss of income (Turner *et al.* 2007).

1.7 The role of Marine Protected Areas (MPAs) in fisheries management

In recent years, marine protected areas (MPAs) have gained considerable attention in tropical coastal areas as "an instrument to preserve vagile fauna and habitat from detrimental effects of fishing (Claudet *et al.* 2006);" and hence is an important tool for reef fisheries management (Francini-Filho and Moura, 2008). This recognition can be attributed to the global overexploitation of many reef fish stocks, as well as the difficulties of using conventional management strategies for the management of such complex, multi-species fisheries (Roberts and Polunin, 1993; Russ and Alcala, 1996; Francini-Filho and Moura, 2008).

The classification scheme devised by the International Union for the Conservation of Nature (IUCN), recognizes an MPA as "any area of inter-tidal or sub-tidal terrain, together with its overlying water and associated flora, fauna, historical or cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (Kelleher, 1999). These areas can provide refuge where exploited species can recover and habitats modified by fishing can regenerate (Apostolaki *et al.* 2002). By protecting both resident species and their biophysical environments, MPAs offer an ecosystem-based approach to conservation and fisheries management that is distinctive from traditional single-species based management regimes (Lubchenco *et al.* 2003).

1.8 MPA Design and Implementation

MPAs are established for a wide variety of functions, including protecting marine species and habitats, conserving genetic diversity, restoring fisheries stocks, managing tourism activities (Roberts, 2007), providing scientific baseline data and educational opportunities; protecting cultural heritage, and minimizing conflict between diverse user groups (Parks *et al.* 2004). An MPA's management strategy will depend upon the objectives of the area in question. In some cases, conservation will be the primary motivating force, while some may be devoted to traditional use; others may focus on the sustainable use of a particular resource, particularly fisheries; and still others may be focused on a combination of these (Agardy and Staub, 2006). These objectives should be reflected in the physical design and governance strategy.

The various objectives of MPA implementation highlight some of the fundamental questions that continue to impede their efficiency. These include reserve size, both individually and as a proportion of the total management area; and reserve placement; both individually and as a network (Halpern and Warner, 2003). For conservation oriented reserves, the optimal size would be as large as possible, minimizing spillover, and promoting self sustainability. The design should hope to incorporate as much representative habitat of the area in question as possible (Agardy and Staub, 2006), as well as genetically diverse populations of the local and regional species (Halpern and Warner, 2003). In contrast, MPAs designed to cater more toward fisheries goals would be optimally designed as small as practically possible to maximize the export of larvae and adults of target species into relevant fishing grounds (Botsford *et al.* 2003). Most MPA planning must ultimately reach a compromise between benefiting the interests of conservation and fisheries.

Variations in location and life histories of different resident species makes it difficult to advocate a specific MPA or MPA network design of given size and spacing (Halpern and Warner, 2003). Fortunately, existing research suggests that reserves encompassing 30-50% of the total stock area can achieve conservation goals for most species (Turpie *et al.* 2000; NRC, 2001; Airame *et al.* 2003). Similarly, evidence suggests that density and biodiversity of benthically associated organisms can be protected and maintained in networks of reserves of moderate size (10–100 km^2) and variable spacing (Murray *et al.* 1999; NRC, 2001; Roberts *et al.* 2001, 2003; Allison *et al.* 2003; Halpern and Warner, 2003). The placement and spacing of these individual reserves would need to consider the biodiversity representation and connectivity within the network, as well as dispersal patterns affecting the self-sustainability of individual reserves and potential threats facing the network area (Roberts *et al.* 2003; Halpern and Warner, 2003).

While current developments are being undertaken to apply a more systematic approach to MPA planning, MPAs thus far have generally been identified and established on an *ad hoc*, opportunistic basis (NRC, 2001; in Agardy and Staub, 2006). Some reserves were established due to their location in relation to adjacent areas of significance such as military installations, sub-tidal anthropogenic structures (ie. oil rigs, communication cables, etc.) or dramatic natural features (Didier, 1998; Johnson *et al.*

1999; Halpern and Warner, 2003). Other reserves were implemented in response to the intial signs of local fisheries collapse (Russ and Alcala, 1996; Murawski, 2000) or to provide areas of “natural” habitat for scientific investigation (Ballantine and Gordon, 1979; Castilla and Dura´n, 1985). For a time, political processes focused on economics, logistics, and public acceptance were the determinant factors is marine reserve designation and design. More recently, MPA planners have tried to consider the complex ecological and biological implications that are crucial to successful reserve establishment (Halpern and Warner, 2003).

The actual level of protective regulation within MPAs varies considerably, which in turn, is likely to influence their utility. That said, fully protected marine reserves are an important subset of MPAs that are increasingly being advocated as a tool for marine conservation and management (Halpern, 2003; Halpern and Warner, 2003; Palumbi *et al.* 2003; Leslie, 2005; Claudet *et al.* 2008; White *et al.* 2008; Lester *et al.* 2009). These reserves are characterized as being “completely protected from all extractive and destructive activities, with explicit prohibitions against fishing and the removal of or disturbance of any living or non-living marine resource, except as necessary for monitoring or research to evaluate reserve effectiveness (Lubchenco *et al.* 2003).” While MPAs with less restrictive management regulations are important management tools in their own right, those which are fully protected offer the greatest protection for marine resources and ecosystems (Lester *et al.* 2009).

1.9 Effects of MPAs

Ecologically, the benefits gained through MPA implementation are typically measured as an increase in the biomass, abundance, and body size (age) of species that exist within the protected area (Jennings, 2000; Halpern, 2003; Kaiser *et al.* 2007) These effects have been demonstrated for a range of phyla (Kaiser *et al.* 2007), including fish (Watson and Ormond, 1994; Buxton and Smale, 1989; Buxton, 1993; Cowley *et al.* 2002; Russ and Alcala, 1996; Bohnsack and Ault, 2002; Willis *et al.* 2001), mollusks (Gell and Roberts, 2003; Murawski, 2000; Tawake *et al.* 2001), crustaceans (Babcock *et al.* 1999), echinoderms (NRC, 2002), and corals (Mumby and Harborne, 2010).

An increase in biomass within the MPA may support the recovery or sustainability of critical spawning stock (Halpern and Warner, 2002), which in turn may be beneficial to adjacent fishing grounds via exportation of biomass (Gell and Roberts, 2003; Francini-Filho and Moura, 2008). Exportation of biomass from MPAs may occur through two mechanisms: 1) net emigration of juveniles and adults, or spillover; and 2) net export of pelagic eggs and larvae, or recruitment subsidy (Gell and Roberts, 2003; Francini-Filho and Moura, 2008). Spillover is expected to operate over small spatial scales ranging from tens to a few hundred metres, depending on the mobility and habitat connectivity of the species in question. Conversely, recruitment subsidy is expected to be more widespread, covering broad spatial scales that generally range tens of kilometers. This process will depend on the dispersal capability of the pelagic larvae and physical hydrodynamic processes (Francini-Filho and Moura, 2008). The MPA-induced benefits to fisheries due to spillover may be minimal compared to those expected from recruitment subsidy (Abesamis *et al.* 2006). However, due to the difficulties associated with evaluating recruitment subsidy, most studies have focused on the effects of spillover when assessing MPA performance (Gell and Roberts, 2003; Sale *et al.* 2005; Francini-Filho and Moura, 2008).

An abundance of case studies have been documented and synthesized which demonstrate how given fish species' abundance, biomass, size, species richness, reproductive potential, and/or community structure may be benefited by reserve protection (Halpern and Warner, 2002; Palumbi, 2002; Gell and Roberts, 2003; Halpern, 2003; Micheli *et al.* 2004; Lester *et al.* 2009). Of these, the most apparent benefits are observed through comparisons of numerical density and biomass of exploited species inside and outside, and/or before and after reserve protection (Côté *et al.* 2001; Gell and Roberts, 2003; Halpern, 2003; Lester *et al.* 2009).

While the utility of MPAs has been widely recognized, their effects may be diverse in direction and magnitude (Halpern and Warner, 2002). Sometimes, biological responses including: abundance, density, biomass, average size, and diversity of organisms; have consistently increased within the reserve over time (Russ and Alcala 1996), showed little change over time (Denny and Babcock, 2004), or initially increased but then decreased back to original levels (Dufour *et al.* 1995). Consequently, reserve effects depend on the objectives responsible for the reserve's establishment, whether the reserve is part of a network, the location, size, habitat quality, and duration of protection; as well as the species under consideration (Jennings, 2000; Mosqueira *et al.* 2000; Côté *et al.* 2001; Jones *et al.* 2004; Micheli *et al.* 2004; Kaiser, 2005; Claudet *et al.* 2008; Lester *et al.* 2009). In many cases, resource managers and stakeholders mistakenly expect major benefits in the short term after MPA establishment, without considering the variable nature of ecosystem complexity and biodiversity (Cakacaka, 2008).

Biological responses to protection are also highly variable among fish taxa (Claudet *et al.* 2008). Generally, species targeted for exploitation benefit more from MPA establishment than non-target species (Côté *et al.* 2001), even in the case of recreational fishing (Westera *et al.* 2003). The individual life history or trophic level of a species will also be a determinant in the magnitude of biological response (Polacheck, 1990; Carr and Reed, 1993; Rowley, 1994; Russ and Alcala, 1998*a*; Jennings *et al.* 1999*a*, *b*). Species which take many years to mature are likely to respond less quickly to protection than short lived species with fast growing life cycles (Jennings *et al.* 1999*a, b*). Additionally, species that are relatively sedentary and spend most of their lives in the reserve area limit their risk of mortality by fishing, and thus often exhibit the most significant response to protection (Halpern and Warner, 2003).

1.10 MPA Assessment and Monitoring

Most approaches to fishery management are based upon results from stock assessment (Shertzer and Williams, 2008). However, data-limited, multi-species fisheries; such as in tropical developing countries like Fiji, may not conform to conventional assessment methods, necessitating the need for alternative approaches to management (Kruse *et al.* 2005).

One possible approach with data-limited stocks is to assign them to individually managed units, or assemblages. Ideally, an assemblage could be assessed using one or more data rich species that could serve as an indicator for similar species. Assemblages may be defined by similarities in biological characteristics such as life history, trophic behavior, or home range (Shertzer and Williams, 2008). In the case of fisheries management, an assemblage assessed on a particular indicator species should be composed of species caught together; in order to benefit all assemblage members. This approach must assume that population trends of the chosen indicator will reflect those of the other species in the assemblage. The stock status of the indicator is used to infer the status of other species in the assemblage, as well as other stocks of the same species (Shertzer and Williams, 2008).

Numerous attempts have been made to identify species that might serve as biological indicators of a given ecosystem, or management area (Stork and Samways, 1995; Mouillot *et al.* 2002; Sosa-López *et al.* 2005, Claudet *et al.* 2006). The concept has been used extensively in water management (Bain *et al.* 2000), but less frequently in marine ecology (Mouillot *et al.* 2002; Sosa-López *et al.* 2005). The use of indicator species as reference points for managing assemblages is arguably a small, but practical step in the direction of ecosystem-based management (Shertzer and Williams, 2008); the same management strategy being implemented currently in many MPAs (Roberts and Hawkins, 2000). Thus, using indicator species can be a valuable tool both in assessing current MPA effectiveness and implementing management strategies to maximize efficiency of future MPA establishment (Nielsen *et al.* 2001).

The effectiveness of MPA implementation and enforcement is highly dependent upon adequate ecological and socio-economic monitoring; as both provide important information for assessing and implementing effective MPA management strategies (Wilkinson *et al.* 2003). That said, indentifying and utilizing appropriate biological and socioeconomic monitoring techniques will be essential to the assessment and dissemination of findings among the reserve’s various stakeholders (IUCN, 2004).

Common biological parameters monitored to assess effects of area protection on a given species include abundance, biomass, organism size, species richness, and species diversity (Lester *et al.* 2009). The majority of recent surveys focused on monitoring species found in shallow subtidal reefs have involved sampling by underwater visual census (UVC) (Edgar *et al.* 2004). The utility of UVC techniques has been recognized (Labrosse *et al.* 2002) particularly in assessments of pelagic or semi pelagic stocks consisting of a limited number of species (Labrosse *et al.* 2002). However, like other field census methods, UVC surveys exhibit bias with respect to such factors as size, appearance and behaviour of target organisms. Differential visibility of organisms, as well as observer behaviour, experience, and subjective decision-making may all contribute to bias (Edgar *et al.* 2004), and consequently make UVC methods poorly adapted for assessments of highly diverse benthic populations (Labrosse *et al.* 2002) .

Another common indicator used in the assessment of fishery stocks as a measure of relative fish abundance is catch per unit effort (CPUE) (Haggarty and King, 2006). Despite its widespread use, CPUE is often considered to assume a strict proportionality with abundance (Harley *et al.* 2001). The most common instance of nonproportionality occurs when CPUE remains high even as abundance decreases, referred to as hyperstability. Factors including: increases in fishing efficiency (catchability), changes in the behaviour of fishers, or fish behavior itself (ie. hyperaggregation) may all contribute to hyperstability (Haggarty and King, 2006), and can lead to overestimation of biomass and underestimation of fishing mortality (Harley *et al.* 2001). This bias is often included in data collected from commercial and recreational fisheries; where fishing efficiency and fisher behavior varies.

Research data may offer control of bias, to some degree, through the use of standardized methods such as pre-determined fishing locations, durations, and gear. Haggarty and King (2006) found that CPUE derived from standardized surveys may not easily be converted into a fish density, but could be used in a relative sense to monitor whether a population is increasing, decreasing, or stable. In such instances, CPUE is capable of being used as a measure of relative abundance, and consequently can be used to monitor fluctuations in a fish population.

Gathering CPUE data also has practical applications in monitoring where limited financial resources are available. While UVC techniques offer a non-lethal method of monitoring species and life stages otherwise inaccessible to certain gears, they can also be expensive; requiring specialized training and/or equipment (ie. SCUBA) (Labrosse *et al.* 2002; Haggarty and King, 2006). Hook and line surveys have potential as a simple, cost effective, and non-lethal method to assess the effects of protection on relative abundance of target species; as well as biological parameters including size, sex, and maturity (Haggarty and King, 2006).

1.11 MPAs in the traditional Fijian context

Government interaction with fisheries management began fairly recently in the Pacific Islands, with the oldest colonial Government fisheries services being established in the late 1950’s (Adams, 1998). The continued trend toward commercialisation of Fiji’s inshore fisheries has influenced the Fisheries Department to focus more on product development rather than resource management (Hand *et al.* 2005). Despite varying levels of government involvement, many local communities have continued efforts to manage local fisheries, particularly the finfish subsistence and small scale artisanal fisheries that consitute most of the catch from Pacific Island coastal waters such as those in Fiji (Dalzell *et al.* 1996). Unsuccessful attempts at implementing Western based management strategies have led to increased Government interest and recognition of traditional management schemes. Unlike government alone, with its fairly limited financial and human resources, community marine tenure systems provide an avenue to assimilate information and apply appropriate management measures at the village level. This grassroots approach offers greater potential to address the issues of such complex multi-species fisheries (Adams, 1998).

Fiji and the entire Pacific island region have long been recognized for their rich heritage of marine use and management (Bartlett *et al.* 2009*a*). Many coastal communities have developed traditional resource management techniques similar to those of modern fishery managers, including restrictions on gear, season, reef areas, species, size, and ownership of marine resources (Cinner *et al.* 2005). Originally, these management techniques were rooted in cultural ceremonies, religion, dietary restrictions, and other traditions rather than explicit practiced for conservation purposes (Foale and Manele, 2004; Cinner *et al.* 2005). However, many of these practices, whether directly or unintentionally, continue to influence resource utilization to this day; especially in rural areas (Matthews *et al.* 1998).

The most important resource management strategy in Fiji is the traditional ownership of the land and coastal waters, which along with its indigenous people form a single indivisible unit; referred to as *vanua*. Fiji's circumstances for such management are particularly noteworthy, as its well established system of customary marine tenure (CMT), or *iqoliqoli*, are given some legal recognition and appear to be the only case worldwide where such tenure has been mapped (Cooke *et al.* 2000). Along with the ability to allow or restrict fishing access of those outside the customary landowner units, village chiefs may also use their traditional government structure to implement policies such as the banning of the use of certain fishing gears, the taking of certain species, or fishing during particular periods of time (Teh *et al.* 2009).

Fiji's customary tenure systems aid in regulating outsider access to fishery resources, but do little to address the issue of regulation among local resource owners themselves. Increasingly, semi-subsistence practices have been incorporated into the lives of customary resource owners to meet the immediate needs introduced by the cash economy. While chiefs have the ability to regulate this type of fishing effort, many do not live within the village, and even those that do rarely concern themselves with such daily supervision. Communities are acknowledging these issues, and the need to develop operational rules involving both those in authority and the fishers who maintain custody over the resource (Ram-Bidesi, 1997).

One customary management strategy that is being applied to supplement current management goals is the declaration of a *tabu* or "no take" area. Traditionally, *tabu* was declared for a 100-day period over certain fishing areas following the death of a high chief. When the *tabu* was lifted, villagers would harvest fish again and hold a feast to commemorate the end of the mourning period. This concept is being adapted to protect spawning or overexploited areas in order to increase fish stocks. The incorporation of traditional customs and knowledge with scientifically based contemporary techniques and knowledge, as well as involvement from the state and resource owners (Ram-Bidesi, 1997), has led to the establishment of the locally managed marine areas, or LMMAs (Aalbersberg *et al.* 2005).

1.12 The Fiji Locally Managed Marine Area Network

The Fiji Locally Managed Marine Areas Network (FLMMA) is a prime example of how community based management regimes can and have been successfully mainstreamed to meet the goals for sustainable development of inshore fisheries (Cakacaka, 2008). Approximately 87 percent of the land in Fiji is native land (NLTB, 2010), owned by local villages, communities or clans under customary resource tenure. These landowners are also entitled to customary fishing rights in their corresponding inshore areas (Aalsbersberg *et al.* 2005). The involvement of these stakeholders in the management and implementation process is critical to conservation, articulation of policies, and fisheries management goals.

Initial work into LMMA establishment in Fiji began during the early 1990’s when village residents of *Ucunivanua* in *Verata, Tailevu* (Viti Levu) sought to recover kaikoso clam (*Anadara antiquata*) stocks (Aalbersberg *et al.* 2005; Tawake *et al.* 2001). Working in conjunction with the University of the South Pacific (USP), the village decided to experiment by setting up a 24-hectare *tabu* on the mudflat and seagrass beds directly in front of the village. The project was a success, with clam abundance and CPUE increasing substantially both within and outside the *tabu* area. In the process, this also generated increased incomes for the village. Originally intended for three years, the *tabu* at Ucunivanua has been extended indefinitely (Veitayaki *et al.* 2003). Since then, similar benefits such as increased incomes, fish and invertebrate sizes and abundance after the establishment of *tabu* areas have also been documented in the *Yavusa Navakavu* LMMA near Suva (van Beukering *et al.* 2007, Cakacaka, 2008).

Success stories like these eventually led to the formation of the Fiji Locally Managed Marine Area Network (FLMMA) in 2000 (Aalbersberg *et al.* 2005). It was established to bring together and inform the diverse range of stakeholders about how community based conservation could best be done (Veitayaki *et al.* 2003). The network consists of 205 participating villages, working in conjunction with 18 partner organisations, including: government departments, regional bodies such as the Secretariat of the Pacific Community (SPC), conservation non-government organisations (NGOs) and the University of the South Pacific (USP). The combined efforts of these participatory bodies have led to the formation of 217 LMMA sites in 116 of Fiji's *iqoliqolis*; an area equivalent to 8,425 km² (Teh *et al.* 2009).

As a part of the greater LMMA network, the main aim of FLMMA is to make the community based learning process more efficient. FLMMA members have agreed to adhere to a common strategy in order to achieve three common goals:

- Implementing more effective projects;
- Systematically learn about the conditions under which this strategy works best and why; and
- Improving the capacity of the members of the portfolio to practice adaptive management.

(LMMA, 2002)

The approach taken by FLMMA enables local communities to take an active role in devising best management techniques of their inshore fisheries, assuming that more effective management will be more beneficial for the improvement of community livelihoods (Cakacaka, 2008). Using a Participatory Learning and Action (PLA) method, FLMMA asserts that “the local people should plan, design, implement and evaluate resource management strategies because this is the group best schooled in the ecological settings and socio-political capabilities of the community (van Beukering *et al.* 2007)”. This usually begins with resource management workshops, in which community members identify difficulties they face in managing their resources, recognize root causes of these difficulties, and ways these problems can be addressed. Partnership with other organizations within the network allows for communities to receive the added benefits of outside expertise and funding; ensuring that sufficient means are available to carry out the planned objectives (Veitayaki *et al.* 2003). Once the management plans have been implemented and followed, further training is conducted such as the biological monitoring and socio-economic training workshops. These enable communities to monitor their no-take zones and analyze results that can help them understand changes that are taking place. Most importantly, this will allow them to identify those management strategies that work, those that don’t; and the ability to communicate these findings with other communities within the network.

1.13 Research Aims and Objectives

This research study aimed to assess the effects of *tabu* areas on the abundance, biomass, size, species composition and and proportion of mature fish within four locally managed marine areas (LMMAs) along the Coral Coast of Viti Levu, Fiji. These LMMAs were initiated and are being monitored by the local communities with assistance from the Institute of Applied Science (IAS) at USP (University of the South Pacific) and Reef Explorer Fiji Ltd. The research conducted aimed to assess whether the MPA’s were providing any of the expected fisheries benefits. Specifically, the objectives were:

- to compare CPUE of species inside each *tabu* with those outside as a proxy for abundance;
- to compare size class of species caught inside each *tabu* with those caught outside;
- to compare the biomass of the catch being obtained at each *tabu* with that being caught outside; and
- to compare the percent of fish caught that had reached reproductive maturity inside each *tabu* with those caught outside

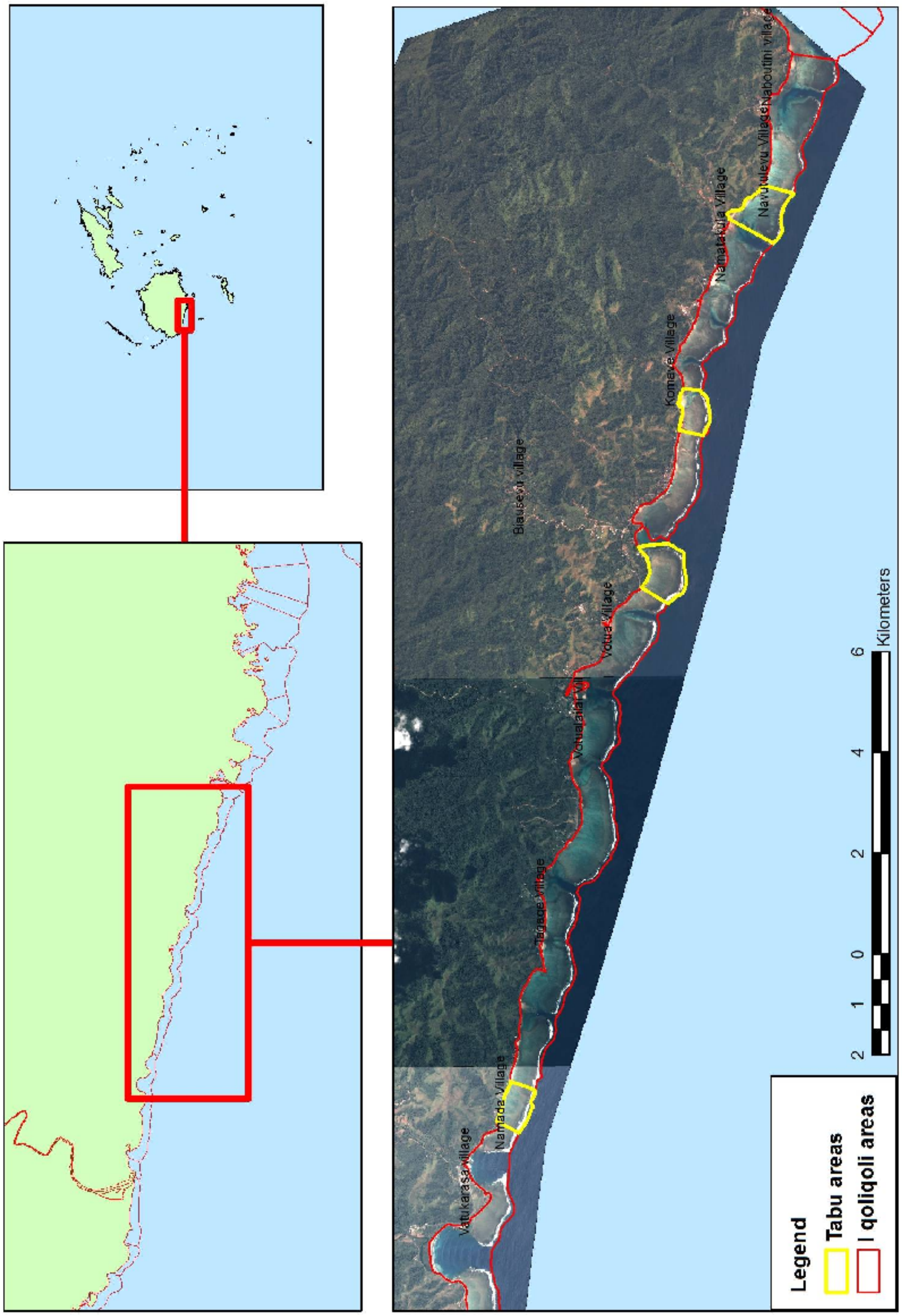
Chapter 2: Methodology

2.1 Site Description

The study area is situated on the Coral Coast; approximately 95 km west from Suva city along the southern coast of Viti Levu, Fiji. This coast is characterised by a chain of fringing reefs, with shallow intertidal platforms up to 700 metres wide; separated by a series of channels located at the discharge points of terrestrial waterways. Most of these reefs are accessible via wading at low or falling tide from the shore all the way to the reef crest that defines their seaward margins. Reefs are comprised of varying degrees of coral, rubble, sand, and algal flats; interspersed with moats and tidal channels. A more in depth account of these reefs can be found in Morton and Raj (1980).

Specifically, the study took place within *tabu* and *tara* areas of four villages: Namatakula, Komave, Votua, and Namada. All are located in the province of Nadroga/Navosa, with Namatakula and Komave both belonging to the *Vanua* (clan) Vusu in the *Tikina* (district) Komave; while Votua and Namada fall within the *Vanua Davutukia* in the *Tikina* Korolevu-i-wai, directly adjacent to *Tikina* Komave (Figure 2.1.1).

Figure 2.1.1: Location of the study site in the Fiji Islands. *Top Right*: Map image of the entire Fiji Island group; Top Left: Enlarged map image of the Coral Coast along the Southern coast of Viti Levu; *Bottom*: Satellite image of the Coral Coast with an overlay of the *tabu* area study sites at Namada, Votua, Komave and Namatakula villages.



While the two districts comprise some of the smallest in country, they also exhibit some of the most densely populated rural areas in Fiji (Fong, 2006). As of 2002, the villages of Namatakula (including Vucilevu), Komave, Votua, and Namada had populations of 300, 200, 260, and 300; respectively. Additional villages located within the *Tikina* Komave include Biausevu and Navola with populations of 100 and 158, respectively. Likewise, another two villages exist within *Tikina* Korolevu-i-wai; Votualailai and Taqage; with corresponding populations of 154 and 260 (Thaman *et al.* 2002).

Over the past 30 years, the population of these districts has changed significantly (Fong, 2006), with a combined population of approximately 7,787 as of 2007 (Fiji Islands Bureau of Statistics). These changes are due in large part to increased tourism (Fong, 2006). Despite the coups of 1987, 2000, and 2006; tourism on the Coral Coast experienced marked growth in the 1980's, 1990's, and into the 2000's, with as many as 100,000 tourist arrivals as of 2003 (Thaman *et al.* 2006). Several resorts and guesthouses or smaller backpacker accommodations can be found within the Komave and Korolevu-i-wai *Tikina*, including Mango Bay Resort, Warwick Resort and Spa, the Beachouse, Naviti Resort, Valase Beach Resort, Rydges Hideaway Resort, and Tambua Sands Resort (personal observation). Settlements, such as Votua housing, have gradually been established to cater for new residents immigrating to the area in search of work, directly or indirectly as a result of the tourism industry. Others including: Qalito, Nasi, Uciwai, Jafau, Korolevu Settlement, Vatu-o-lailai, and Nagasau were established to cater for those who have married into families of local village residents (Fong, 2006). Along with increased population, the growth of tourism has altered lifestyles throughout the Coral

Coast from a mainly subsistence focus of farming and fishing to a mixed income subsistence lifestyle, with many people from the villages working at nearby resorts (Thaman *et al.* 2002).

A wide range of issues influence coastal areas within the two districts including: tourism, depletion of fisheries, solid waste management, sewage pollution, coral harvesting, storm surge and flooding, agriculture, forestry, small-scale sand mining; as well as siltation from coastal and inland erosion. In turn, the area has a history of participation in environmental planning and management regimes. Numerous environmentally related workshops have been held throughout the local communities to address these issues; with attempts to involve local villagers as well as outside partners, including the nearby resorts (Thaman *et al.* 2002). It was these types of workshops that led to the development of marine resource management plans, including the establishment of LMMAs *Vanua Vusu* in *Tikina* Komave, and *Vanua Davutukia* in *Tikina* Korolevu-i-wai. Plans included the designation of small *tabu* areas, as well as environmental committees and community biological monitoring of target species. With the help of IAS and the Nadroga Provincial Office, these strategies ultimately culminated in the establishment of no take MPAs within the *Vanua Davutukia* LMMA during 2002-2003 and *Vanua Vusu* LMMA in 2004. Along with the other villages of their respective *Tikina*, each of the four villages involved in this study have designated their own *tabu* area within their *iqoliqoli* as their protected area management device (Thaman *et al.* 2006).

Tikina Komave: Namatakula and Komave Villages

Namatakula, along with the incorporated Vucilevu, are the first villages in the *Tikina* Komave while traveling from Suva to Sigatoka along the Coral Coast of Viti Levu. They are located adjacent to a small bay, less than 1 kilometer along the coast from Mango Bay; a resort where several of the villagers are employed. Fishing is an important activity in village life. Singh (2005) found that 57% of the households fish 1-2 times per week while 33% of the households fish 3-4 times per week. The majority of fishers spend between 1-4 hours for a typical fishing excursion, with the most targeted habitats being the shoreline and lagoon (shallow and deep water). Fish consumption within the village is high as well, with 1995 survey reporting rates of approximately 2kg/person/meal/day and a high of 5.8kg/person/meal/day on Sunday (Rawlinson *et al.* 1995).

Their LMMA was established in 2004 in cooperation with other villages in the *Tikina*, and is approximately $4.89\,km^2$ in total area. A subsequent *tabu* was also demarcated. The *tabu* extends from the beachfront to the reef crest and from the reef passage transversing the inshore area in front of the village, to approximately 1.2 kilometers down the coast, past Mango Bay Resort's beachfront. The total area of the *tabu* is approximately $0.8\,km^2$ or 16.4% of the total LMMA.

Komave village, after which the district is named, is situated on a small bay along the coast approximately 2.81 km past Namatakula and is the last village before reaching the Warwick Hotel and Spa, an important source of employment and development opportunities for the village. They share the same LMMA as Namatakula and have also demarcated their own *tabu* in front of the village. At $0.46\,km^2$, or approximately 9.4% of the total LMMA, the *tabu* extends over .75 km from the channel west of the village toward the Warwick Resort. Komave's *tabu* was recently opened for fishing once more during December 2008, and remains so to date (Josaia Totonavosa, personal communication).

LMMA baseline biological monitoring took place in both villages during 2005 and 2007 with three fish species groupings; parrotfish (*bune*), snapper (*damu*) and emperor (*kabatia*); and two invertebrate groupings: octopus (*hulua*) and *Tripneustis gratilla* (*cawaki*) being used as biological indicators. Baseline data pertaining to socio economic standards was also conducted during July 2007. Follow up biological and socio economic monitoring was conducted in 2009.

Tikina Korolevu-i-wai: Votua and Namada Villages

Votua village is the first village in the Korolevu-i-wai district when traveling from Suva to Sigatoka along the Coral Coast, located approximately 5.5 km farther northwest of Komave village along the Queen's Road. Established in 2003, the LMMA of which Votua and the other villages of Korolevu-i-wai share is approximately $9\,km^2$, encompassing the entire *iqoliqoli*. Of this, Votua's *tabu* area is approximately $0.78\,km^2$ or 8.7% of the total LMMA. It extends from the beachfront to the base of the reef crest, and is linearly demarcated by five buoys on opposing ends. The *tabu* area is located adjacent to the area locals refer to as '*Namahara*,' directly in front of renowned Vilisite's tourist accommodation and restaurant, roughly 0.75 km to the southwest of the village. Biological monitoring began in 2003, with follow up surveys conducted in 2004, 2005, and 2007 by USP and Coral Cay Conservation. Community monitoring has also been taking place on a yearly basis. Main community indicators selected included *Scaridae* spp. (*Ulavi*) and live corals (*Lahe*).

Namada lays at the most western end of the *Tikina* Korolevu-i-wai, roughly 10.5km from Votua Village. Separating the two are several resorts including Naviti, Hideaway, and Tambua Sands Resorts as well as Valase Beach Resort, Maui Bay Estates. Namada's designated *tabu* area is located across the road adjacent to the village, extending from the beach front to reef crest, with a total area of approximately $0.47\, km^2$ or 5.2% of the total LMMA. Baseline biological surveys were conducted in 2003 using indicators chosen by the community including: *Lethrinus* spp. (*Kabatia*), *Scaridae* spp. (*Ulavi*), *Tripneustis gratilla* (*Cawaki*) and hard coral (*Lase Bula*). Several follow up surveys have been conducted, with the latest taking place in 2009.

Unlike Komave, the *tabu* areas at Namatakula, Votua, and Namada have officially remained completely no-take, with only one temporary opening occurring for a few hours in Namada for a special village function. Despite additional help from the Department of Fisheries in training and designating community members as fish wardens (Thaman *et al.* 2006), poaching has been noted in every village (Victor Bonito personal communication).The occurrence and intensity of these events are difficult to quantify, with instances taking place as recent as during this study (personal observation).

2.2 Preparation and Data Collection

Prior to carrying out field work, each village was approached and consulted by members of IAS and other FLMMA partners. This included the presentation of *sevusevu* to the village, as well as an information and discussion period for each village to clarify the aims of the research to be carried out. The decision was made to assess the respective *tabu* areas using traditional fishing methods that would be temporally and financially feasible, as well as relevant for assessing fish species important to the villagers.

Due to the *tabu* areas' locations along intertidal fringing reef, the women, who most commonly fish such areas, were sought to participate in the study. Women were consulted about appropriate dates, times, and locations to conduct fishing, as well as compensation for their participation. The decision was made to pay each women participating $2.50 for each hour fished.

Fishing took place over an 8 month period; from March through May and August through December. Over the 47 days that fishing was carried out, reasonable distribution of effort was given to both protected areas (*tabu*) and areas open to fishing (*tara*) at all the four village *iqoliqoli* (Table 2.1). *Tara* sites were geographically proximate and contiguous to *tabu* sites, taking place across the intertidal flats of fringing reefs.

Line-fishing, a common method employed by local women, was the sole fishing method utilised for the study. This method entailed the use of standard fishing line, wrapped around a plastic bottle, with a size #13 fishing hook. Baits included hermit crabs (*kasikasi*), octopus (*hulua)*, and occasionally canned mackerel. Attempts were made to fish at varied locations both within the *tabu* and *tara* areas. These locations were chosen by the local women either because they are commonly fished areas, or were based on subjective decision making of a given day. During excursions, women were free to move about the reef flat to fish where they perceived fit. Additionally, attempts were made to restrict fishing to clear, sunny days when weather was considered optimal for fishing.

Along with the fishers present, most fishing excursions included at least one villager serving as a runner; bringing fish in a plastic bucket for identification, measurement, tagging, and release. In most cases, this process was completed within five minutes of capture, and within one minute of being removed from water. Fish caught were identified as accurately as possible to the species level, as well as their local Fijian name. In any case where fish identification was questionable, a photograph was taken for later review with the aid of fish identification books such as Allen *et al.* (2003). The size of each fish was also recorded using fork length (cm) as the standard measure. Fish were then tagged with color coated, numbered, FLOY T-bar tags; documented; and released. Any additional information, such as fish death due to deep hooking (i.e. hook lodged in either the throat or gut), was also recorded.

2.3 Data Analysis

Following each fishing excursion; relative abundance in the form of mean CPUE was calculated by dividing the number of fish caught (n) by the CPUE effort coefficient (f). f was calculated by multiplying the number of fishers by the number of hours spent fishing. Therefore, CPUE can be written as $^{n}/_{f}$. Likewise, the biomass of given fish was determined by first obtaining the weight (W) of each fish through the equation $W = aL^b$; where L is the recorded length/size, a is the intercept, and b is the slope of the fitted linear relationship. Weight calculation was limited to those fish for which a and b length-weight metrics could be obtained (Appendix A). In the case of multiple a and b metrics for a given species, priority was given to those studies with the greatest number of fish (n) analysed or the study conducted in the geographical locale closest to Fiji. For those species where no fork length, length-weight metrics were available; *a* and *b* length-length metrics were used to convert the fork length of individuals to the appropriate length type. This involved substituting the relevant values into one of two equations: $TL = a + bFL$ or $SL = \frac{FL - a}{b}$; where FL is the fork length, TL the total length, SL the standard length, *a* the intercept, and *b* the slope of the fitted linear relationship. The sum of the weights of fish caught were then further divided by the CPUE effort coefficient *f*, or W/f, to determine the mean *W* in grams of fish caught per person per hour. Upon completion of work in the field, the CPUE, biomass and size data recorded were input into Microsoft Excel, separating fish caught by village site, area, date, species and Fijian taxonomic name. In addition to the four village sites, a combined "MPA-effect" variable was included which lumped data from the three villages whose MPAs have remained closed since their inception; Namada, Votua, and Namatakula. This data was exported to the JMP statistical software package (Ver. 5.0.1.2) for further analysis.

Preliminary analysis via JMP summary tables allowed for determining sample size (*n*) of individual species at each area and site. It was decided that species with an *n* of less than 10 at each site area would not be large enough for sufficient analysis of abundance, biomass, or size; leading to the exclusion of all species from individual analysis except: *Epinephelus hexoganatus*, *Epinephelus merra*, *Lethrinus harak*, and *Lethrinus obsoletus*. However, due to the *n* of *Lethrinus harak* and *Lethrinus obsoletus* at individual sites, only the Namada and combined "MPA-effect" data for each were included. For analysis at the group level, all *Epinephelus* spp. and *Lethrinus* spp. were included regardless of the *n* of individual species. The only exception to these were

Epinephelus bleekeri and *Epinephelus tauvina*, which were excluded from group size analysis due to their exceptionally larger size capability compared to the other species of their grouping being analysed (Allen *et al.* 2003). These groups were classified in accordance with their local Fijian names; as all *Epinephelus* spp. were associated with the name *Kahala* and all *Lethrinus* spp. with *Kabatia*.

The mean CPUE, as well as standard error of the mean, were calculated for the total, group level, and individual species by site and area based on the number of days excursions took place. The same was calculated for mean biomass. Similarly, mean size and standard error of the mean were calculated at group and individual species level by site and area. CPUE, biomass, and size were checked for normality of mean distribution using a Shapiro-Wilk Test for normality (*P*= 0.05). While most authors agree that this is the most reliable test for non-normality for small to medium sized samples (Conover, 1999), it does not guarantee normality of distribution, and therefore must be used prudently. Thus, all corresponding *tabu* and *tara* area distributions indicating normality were plotted in a histogram and checked for the size of their respective sample size (*n*) value. Those with *n* values lower than five were excluded. All others were considered on a case by case basis, with subjective decision making based on the distribution of the population displayed in their respective histogram.

Pair-wise comparisons were used due to the inherent differences in habitat type and quality, reserve size and placement, and varying levels of fishing intensity across sites. Species and group variables with normally distributed means between *tabu* and *tara* areas were checked for significant differences between these areas using a one-way analysis of variance (ANOVA) parametric test. Due to the statistical power of this type of test, all area/site samples lacking normal distributed means were transformed in an attempt to normalise the data. This was done using Log_{10} for the size data and $Log_{10}+1$ for CPUE and biomass data. The Shapiro-Wilk Test for normality, followed by examination of *n* value and histogram distribution, were repeated for all transformed data. Any site/area groupings with subsequent normal distributions were compared using one-way ANOVA. Those groupings still lacking normal distributions were then analysed using a non-parametric Wilcoxon rank-sum Test, with an alpha level of 0.05.

Summary statistics of the species catch composition were calculated for each area of all sites. Species were grouped into six categories: *Epinephelus* spp., *Lethrinus* spp., *Cheilinus* spp., *Balistidae* spp., *Lutjanus* spp. and other spp. These values were plotted in a stacked bar graph with the help of Microsoft Excel 2003.

Additionally summary statistics were calculated for each species for which the size at sexual maturity (SSM) was known. A review of the literature was conducted to find the estimated maturity size of as many species as possible. The mean length at which fish of a given population become sexually mature for the first time (L_m) is often expressed as the length at which 50% of the population is estimated to be mature (L_{50}). L_{50} was used extensively, but was substituted for higher metrics (ie. L_{90}) if available. Preference given to the latter was to help decrease the probability of underestimating the percentage of immature fish. In the case of more than one SSM with the same metric for a given species, priority was given to those studies with the greatest number of fish (*n*) analysed or the study conducted in the geographical locale closest to Fiji. For those species where *Lm* was not available for fork length, *a* and *b* length-length metrics were used to convert the fork length of individuals to the appropriate length type. This involved substituting the relevant values into one of two equations: $TL = a + bFL$ or $SL = \frac{FL - a}{b}$; where FL is the fork length, TL the total length, SL the standard length, *a* the intercept, and *b* the slope of the fitted linear relationship.

The percentage of fish caught at each site area that were of SSM was calculated by dividing the number of fish at SSM by those that were not. As with the analysis of abundance, size, and biomass data, it was decided that species with an $n \geq 10$ at each site area would not be large enough for sufficient summary statistics, leading to the exclusion of all species from individual analysis except: *Epinephelus hexoganatus*, *Epinephelus merra*, *Lethrinus harak*, and *Lethrinus obsoletus*. However, due to the *n* of *Lethrinus harak* and *Lethrinus obsoletus* at individual sites, only the “combined MPA effect” data for each were included. Again, as with the other parameters analysed, higher taxonomic groupings were used and classified in accordance with their local Fijian names: *Kahala* for all *Epinephelus* spp. and *Kabatia* for all *Lethrinus* spp. For analysis at the group level of *Kahala* and *Kabatia*, percentage of fish at SSM for all *Epinephelus* spp. and *Lethrinus* spp. were calculated for each individual species of the grouping and summed together for the group percentage.

Chapter 3: Results

Fisher participation varied daily across sites, ranging from two to thirty fishers, with a mean of eight. Similarly, time fished varied from 1.5-5.5 hours, with most excursions commencing within three hours of low tide. Effort, f, as a measure of fisher participation and duration, or number of fishers multiplied by the number of hours fished, varied across sites from approximately 66 up to approximately 266 effort hours.

Table 3.1.1: The total number of days and hours fished within *tabu* and *tara* sites of each village site

Village	**Total days fished in *tabu***	**Total hours fished in *tabu***	**Total days fished in *tara***	**Total hours fished in *tara***
Komave	7	265.89	4	148.25
Namada	4	209.42	3	145
Namatakula	8	206.75	3	137.39
Votua	11	126.33	7	61.33

3.1 Catch Composition

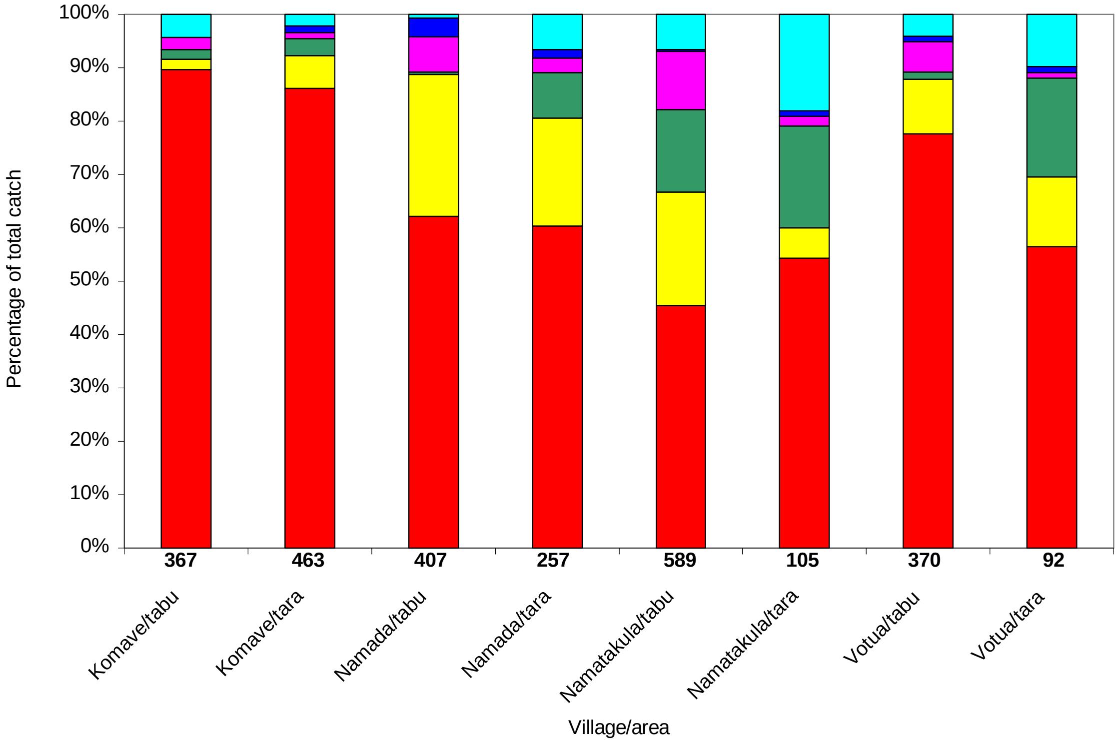
During the forty seven days fishing was carried out, 2,650 fish, consisting of 55 species, were caught and documented. Catch composition in each area of the four sites was grouped into six categories: *Epinephelus* spp., *Lethrinus* spp., *Cheilinus* spp., *Balistidae* spp., *Lutjanus* spp., and other spp. (Figure 3.1.1)

Epinephelus spp. exhibited the greatest proportion of the catch at all village sites. This trend was most pronounced at Komave, where catch of *Epinephelus* spp. composed 89.6% of the total catch in the *tabu* area and 86.2% in the adjacent *tara* area. On the other hand, catch of *Lethrinus* spp. at Komave was fairly low, composing only 1.9% of the catch in the *tabu* and 6% in the adjacent *tara*.

The *tabu* areas at Namada, Namatakula, and Votua all exhibited catch composed mainly of *Epinephelus* spp., with 62.2%, 45.5%, and 77.6%; respectively. Similar figures were exhibited within the adjacent *tara* areas of these three sites, ranging from 54.3% at Namatakula to 60.3% at Namada. Unlike Komave, the *tabu* areas at these three sites had from 10.3-26.5% of the catch consisting of *Lethrinus* spp. The *tara* areas of each site also had greater percentage of *Lethrinus* spp. catch than the *tara* area at Komave. Lower percentages of *Lethrinus* spp.were also caught at the *tara* area of Namada and Namatakula sites than at their corresponding *tabu*. Votua had a slightly greater catch composition of *Lethrinus* spp. (2.7%) within its *tara* as opposed to its *tabu*.

Most of the catch from the other three family-based species groupings composed less than 10% of the total catch from most area at most sites. The only exception was *Cheilinus* spp., which composed 15.4% of the catch in the *tabu* area of Namatakula and 19.0% of the catch in the adjacent *tara* area. *Cheilinus* spp. also composed 18.5% of the catch in the *tara* area adjacent to the Votua’s *tabu*. Additionally, most of the catch from the “other spp.” grouping composed less than 10% of the total catch from most areas at most sites, except in the *tara* area of Namatakula, where it made up 18.1% of the total catch.

Figure 3.1.1: Catch composition of the four village sites by percentage caught of each species groupings. Percentages of the total caught at each site area are represented by color-shaded bars, where red denotes the percentage of *Epinephelus* spp., yellow denotes the percentage *Lethrinus* spp., green denotes the percentage of *Cheilinus* spp., pink denotes the percentage of Balistidae spp., blue denotes the percentage of *Lutjanus* spp., and turquoise denotes the percentage of other species. Number of total fish caught is listed below each area column.

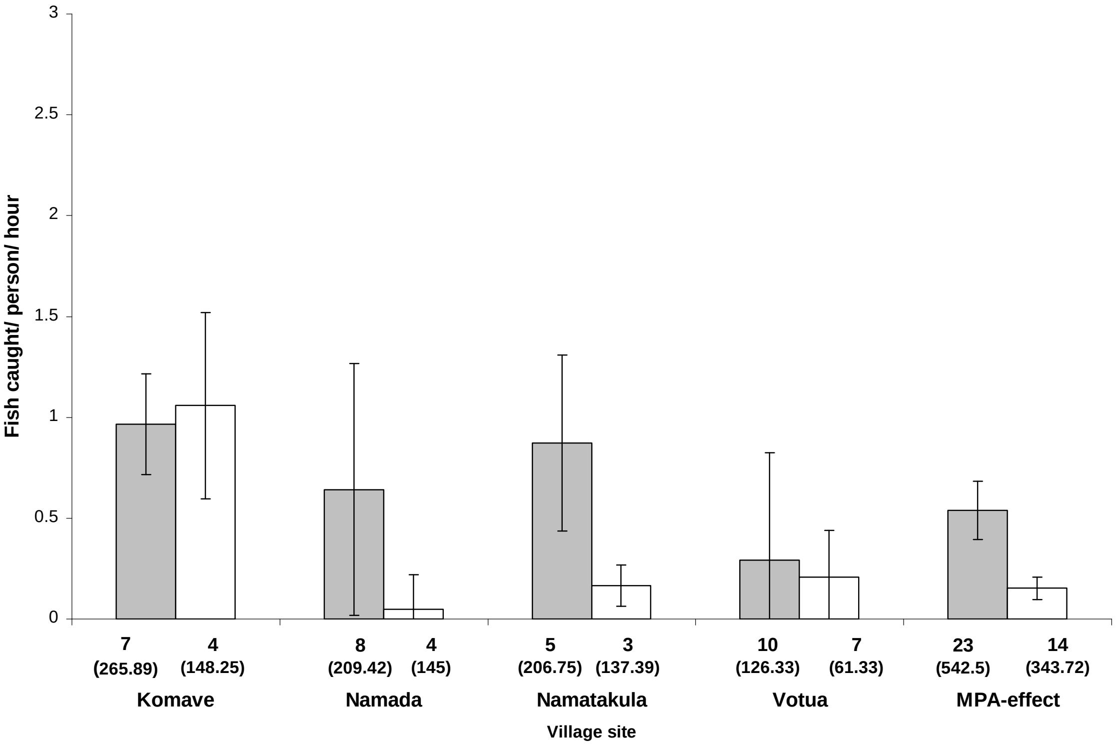


3.2 CPUE Analysis

Of the 55 species, only four: *Epinephelus hexagonatus*, *Epinephelus merra*, *Lethrinus harak*, and *Lethrinus obsoletus*; had a sample size per area (*n*≥10) suitable for CPUE analysis at nearly all village and combined village sites. Consequently, the same was true of the taxonomic groupings Kahala (*Epinephelus* spp.) and Kabatia (*Lethrinus* spp.). Of these species and groupings, none demonstrated normal variance, making them unsuitable for a parametric ANOVA test.

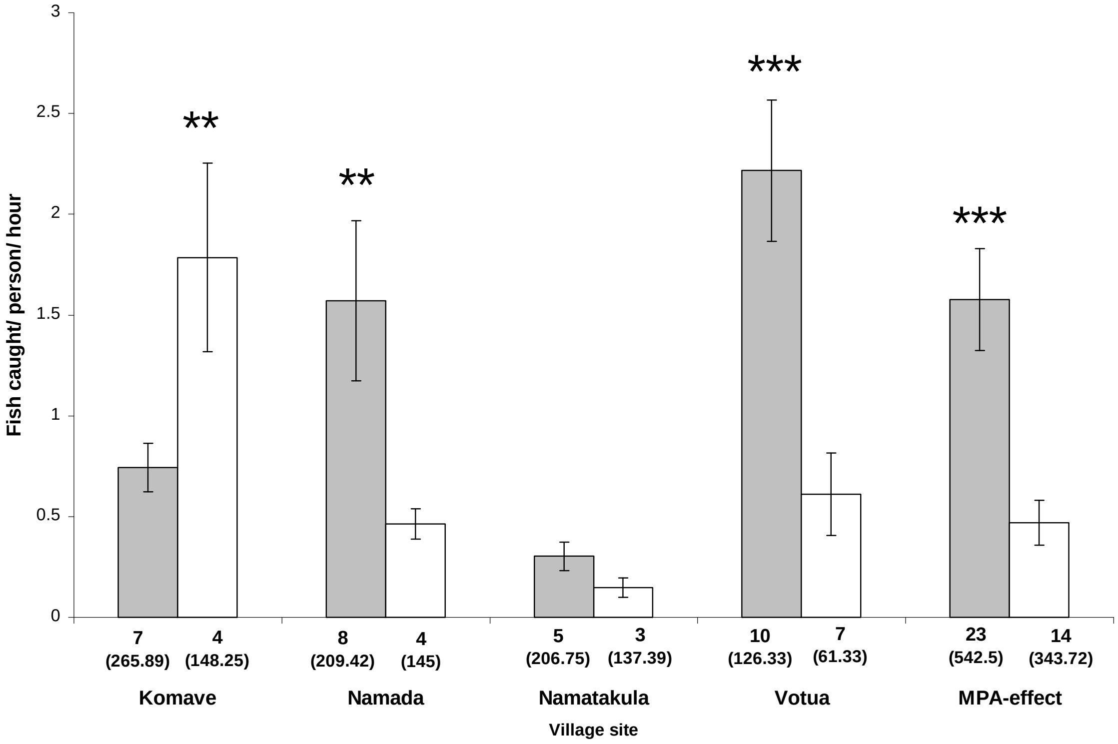
While the mean CPUE of *Epinephelus hexagonatus* (Figure 3.1.1) was greater in the *tabu* areas of the Namada (*Wilcoxon rank-sum test*, *Z*= -1.29668, *P*= 0.1947) Namatakula (*Wilcoxon rank-sum Test*, *Z*= -1.49071, *P*= 0.1360), Votua (Wilcoxon rank-sum test, *Z*= 0.56671, *P*= 0.5709) and combined “MPA-effect” (*Wilcoxon rank-sum Test*, *Z*= -1.20055, *P*= 0.2299) than at adjacent fished areas, differences between areas at each site were not significant. Similarly, mean *E. hexagonatus* CPUE was greater within the *tara* area of the Komave LMMA, but not significantly greater than the *tabu* area (*Wilcoxon rank-sum Test*, *Z*= -0.66144, *P*= 0.5083).

Figure 3.2.1: Results of Wilcoxon rank-sum Test on mean CPUE of *Epinephelus hexagonatus* for the four village sites and the combined "MPA-effect" group. Number of days and total hours (in parentheses) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars.



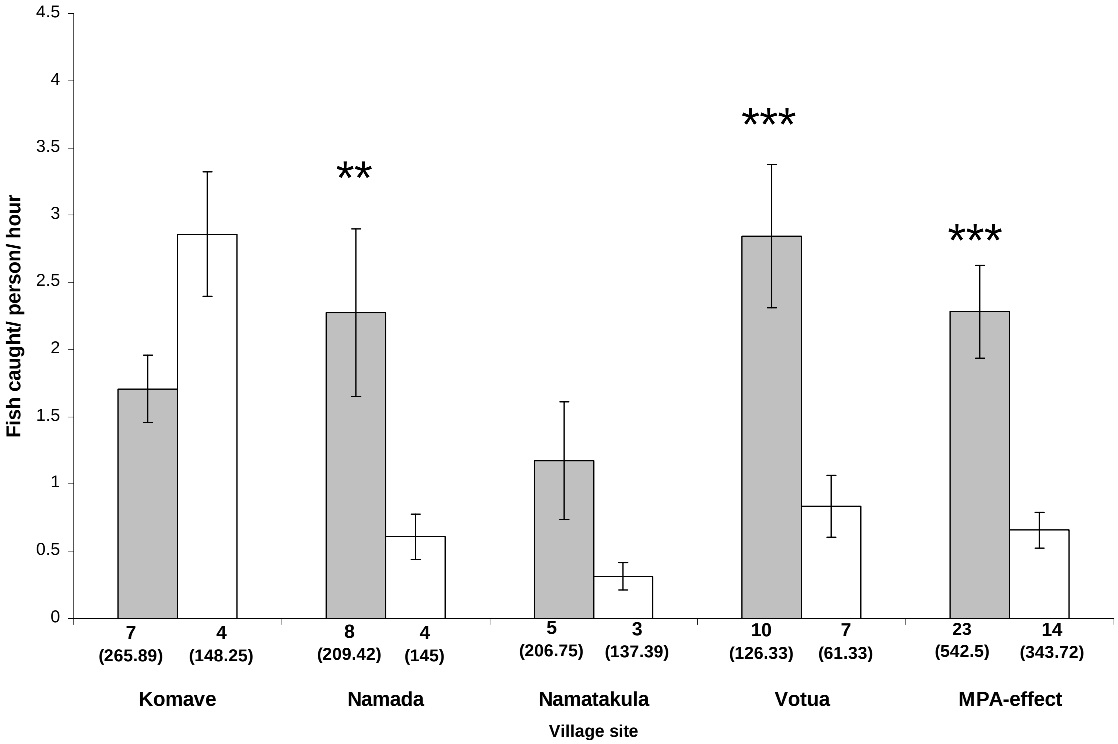
Mean CPUE of *Epinephelus merra* (Figure 3.1.2) was significantly greater within the *tabu* area of Namada (*Wilcoxon rank-sum Test, Z*=-2.12674, *P*= 0.0334), as well as highly-significant within the *tabu* areas of the Votua (*Wilcoxon rank-sum Test*, *Z*=-2.78132, *P*= 0.0054) and for the combined "MPA-effect" *(Wilcoxon rank-sum Test, Z*= -3.24143, *P*= 0.0012) group. In the Komave LMMA, mean *E. merra* CPUE was significantly greater within the *tara* area as opposed to the *tabu* area (*Wilcoxon ranked-sum Test*, *Z*=1.98431, *P*= 0.0472).

Figure 3.2.2: Results of Wilcoxon rank-sum Test on mean CPUE of *Epinephelus merra* for the four village sites and the combined "MPA-effect" group. Number of days and total hours (in parentheses) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference (0.01≤*P*≤ 0.05), *** denotes a highly-significant difference (*P*< 0.01)



When grouped to the taxonomic level, the mean CPUE (Figure 3.1.3) of *Epinephelus* spp. at the Namada *tabu* was significantly greater than its *tara* area (Wilcoxon rank-sum Test, *Z*= -2.29286, *P*= 0.0219). Greater mean CPUE within the *tabu* area when compared to *tara* was highly-significant at both the Votua site (*Wilcoxon rank-sum Test*, *Z*= -2.58614, *P*= 0.0097) and the combined "MPA-effect" group (*Wilcoxon rank-sum Test*, *Z*= -3.64835, *P*= 0.0003).

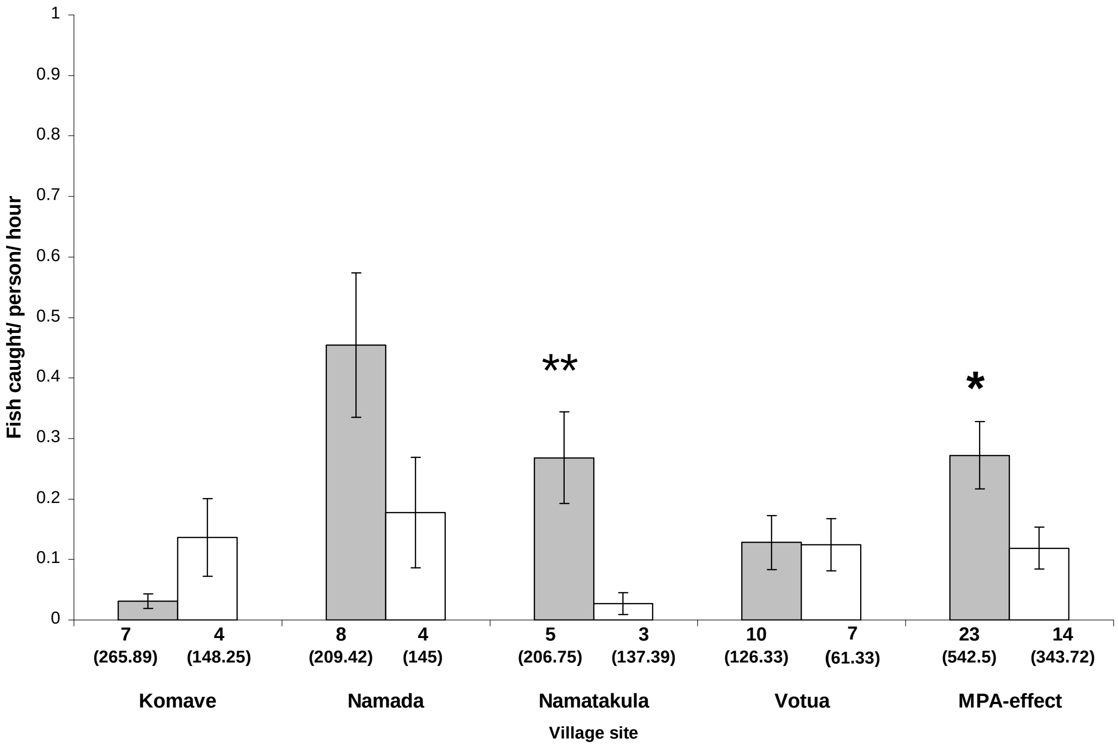
Figure 3.2.3: Results of Wilcoxon rank-sum Test on mean CPUE of *Epinephelus* spp. for the four village sites and the combined "MPA-effect" group. Number of days and total hours (in parentheses) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference (0.01≤*P*≤ 0.05), *** denotes a highly-significant difference (*P*< 0.01)



Mean CPUE of *Lethrinus harak* (Figure 3.1.4) was greater within the *tabu* areas at the Namada and Votua, but not significantly greater. Only Namatakula had significantly greater mean CPUE within its *tabu* area compared to the adjacent *tara* area (*Wilcoxon rank-sum Test, Z*= -2.08700, *P*= 0.0369). A marginally-significant difference was also exhibited for the combined "MPA-effect" group (*Wilcoxon rank-sum Test, Z*= -

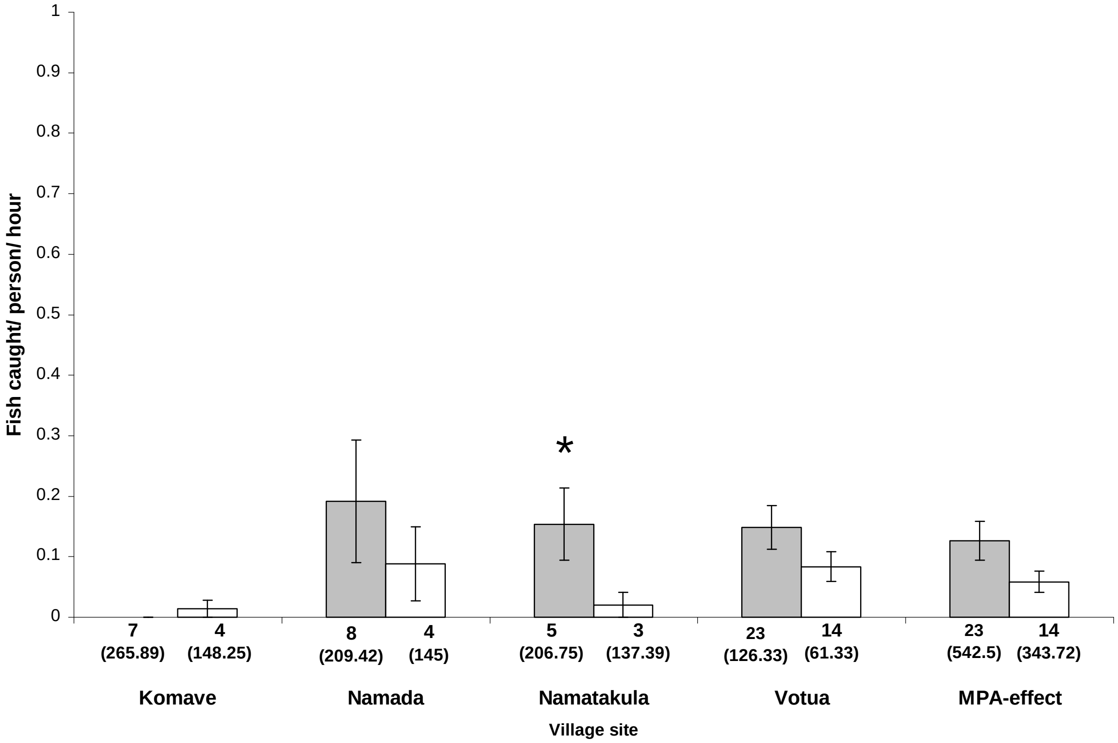
1.91690, *P*= 0.0553). Mean CPUE within the *tara* area of the Komave site, while greater, was not significant.

Figure 3.2.4: Results of Wilcoxon rank-sum Test on mean CPUE of *Lethrinus harak* for the four village sites and the combined "MPA-effect" group. Number of days and total hours (in parentheses) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. * denotes a marginally- significant difference (0.1>*P*>0.05), ** denotes a significant difference (0.01≤*P*≤ 0.05)



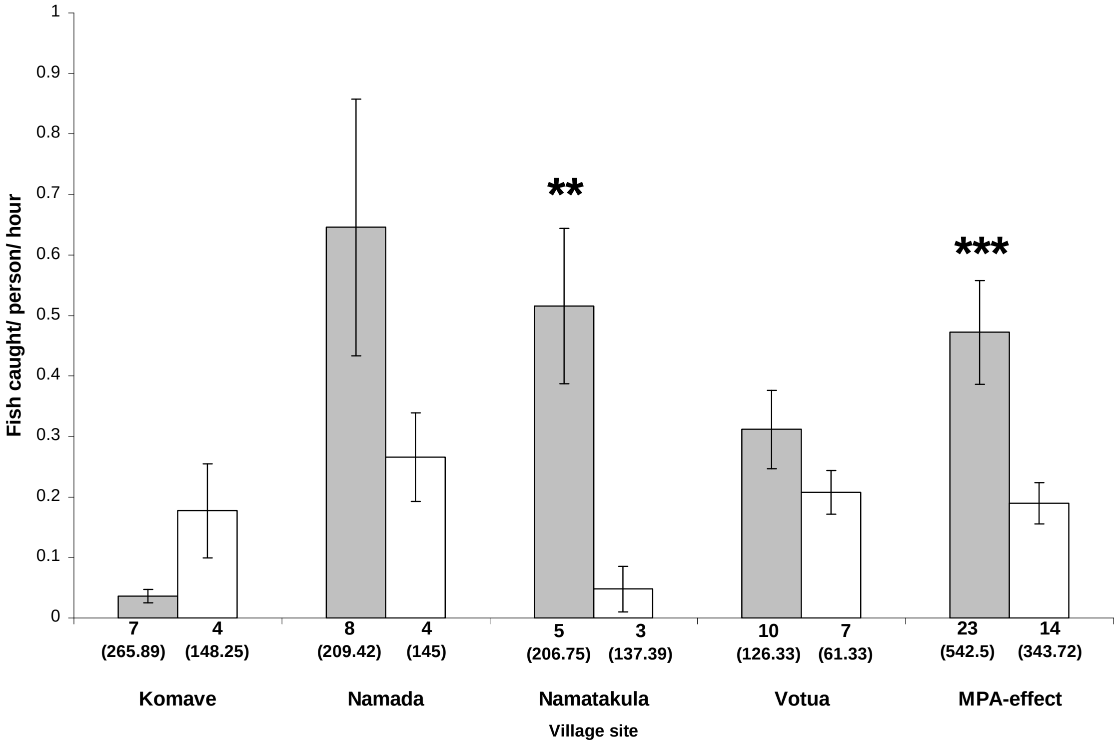
Mean CPUE of *Lethrinus obsoletus* (Figure 3.1.5), while greater in the *tabu* areas of Namada, Votua, and the "MPA-effect" group compared to their respective *tara* area, the difference was not significant. Only the greater mean CPUE of the *tabu* area at Namatakula was considered marginally-significant when compared with its respective *tara* area (*Wilcoxon rank-sum Test*, *Z*=-1.79960, *P*= 0.0719). At the Komave site, *tara* area mean CPUE was greater than that in the *tabu*, but was not significantly different.

Figure 3.2.5: Results of Wilcoxon rank-sum Test on mean CPUE of *Lethrinus obsoletus* for the four village sites and the combined "MPA-effect" group. Number of days and total hours (in parentheses) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. * denotes a marginally- significant difference (0.1>*P*>0.05)



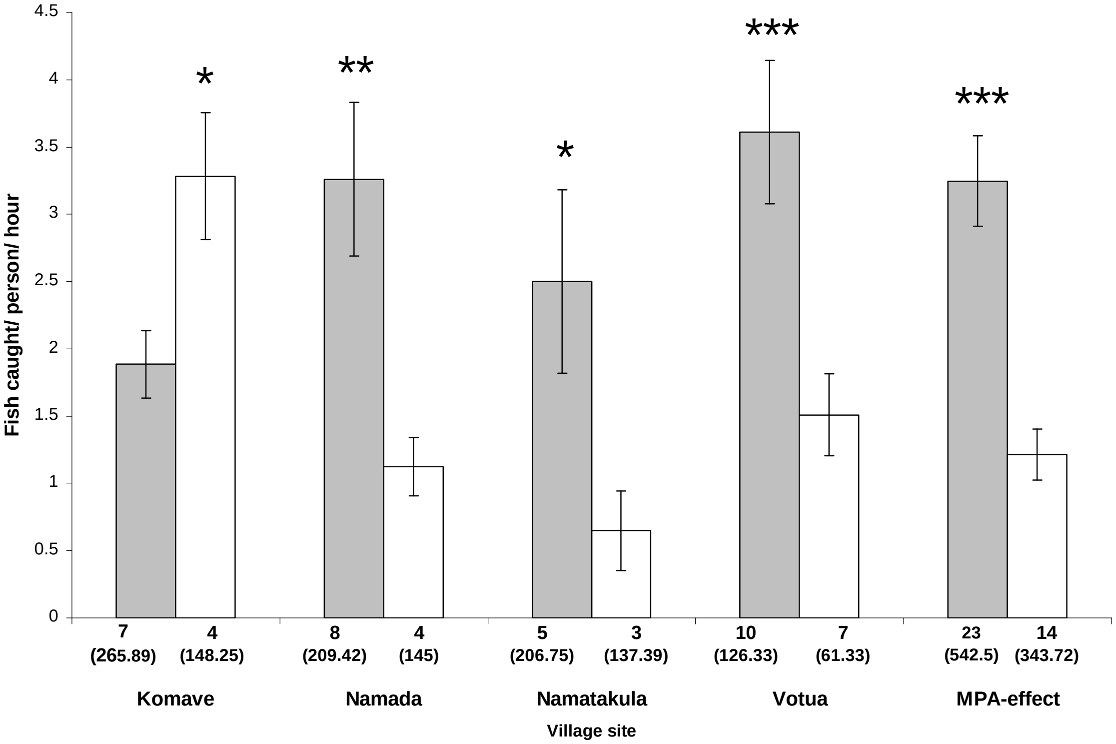
When grouped to the taxonomic level, the mean CPUE (Figure 3.1.6) of *Lethrinus* spp. at the Namada and Votua sites was greater within their *tabu* area than the adjacent *tara* area, but not significantly so. However, the mean CPUE of the Namatakula *tabu* was significantly greater than its *tara* area (*Wilcoxon rank-sum Test*, *Z*=-2.09953, *P*= 0.0358). This difference was highly-significant when combined for an overall "MPA-effect" (*Wilcoxon rank-sum Test*, *Z*= -2.75633, *P*= 0.0058). The *tara* area at Komave displayed greater mean CPUE than the *tabu* area, but the difference was not significant (*Wilcoxon rank-sum Test*, *Z*= 1.42060, *P*= 0.1554).

Figure 3.2.6: Results of Wilcoxon rank-sum Test on mean CPUE of *Lethrinus* spp. for the four village sites and the combined "MPA-effect" group. Number of days and total hours (in parentheses) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference (0.01≤*P*≤ 0.05), *** denotes a highly-significant difference (*P*< 0.01)



Analysis of the total catch (Figure 3.1.7) showed greater mean CPUE in the *tabu* areas of Namada, Namatakula, Votua, and the combined “MPA-effect” group compared to their adjacent *tara* areas. Of these, highly-significant differences were observed at Votua (*Wilcoxon rank-sum Test*, *Z*= -2.58614, *P*= 0.0097) and the “MPA-effect” (*Wilcoxon rank-sum Test*, *Z*= -2.84300, *P*= 0.0045) grouping, as well as a significant difference between *tabu* and *tara* at Namada (*Wilcoxon rank-sum Test*, *Z*= -2.12302, *P*= 0.0338). Greater mean CPUE within the *tabu* area was also marginally-significant at Namatakula (*Wilcoxon rank-sum Test*, *Z*= -1.78885, *P*= 0.0736). Komave was the only site where mean CPUE was greater within the *tara*, a difference that was marginally-significant (*Wilcoxon rank-sum Test*, *Z*=1.79533, *P*= 0.0726).

Figure 3.2.7: Results of Wilcoxon rank-sum Test on mean CPUE of the total catch for the four village sites and the combined “MPA-effect” group. Number of days and total hours (in parentheses) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. * denotes a marginally-significant difference (0.1>P> 0.05), ** denotes a significant difference (0.01≤P≤ 0.05), *** denotes a highly-significant difference (P< 0.01)

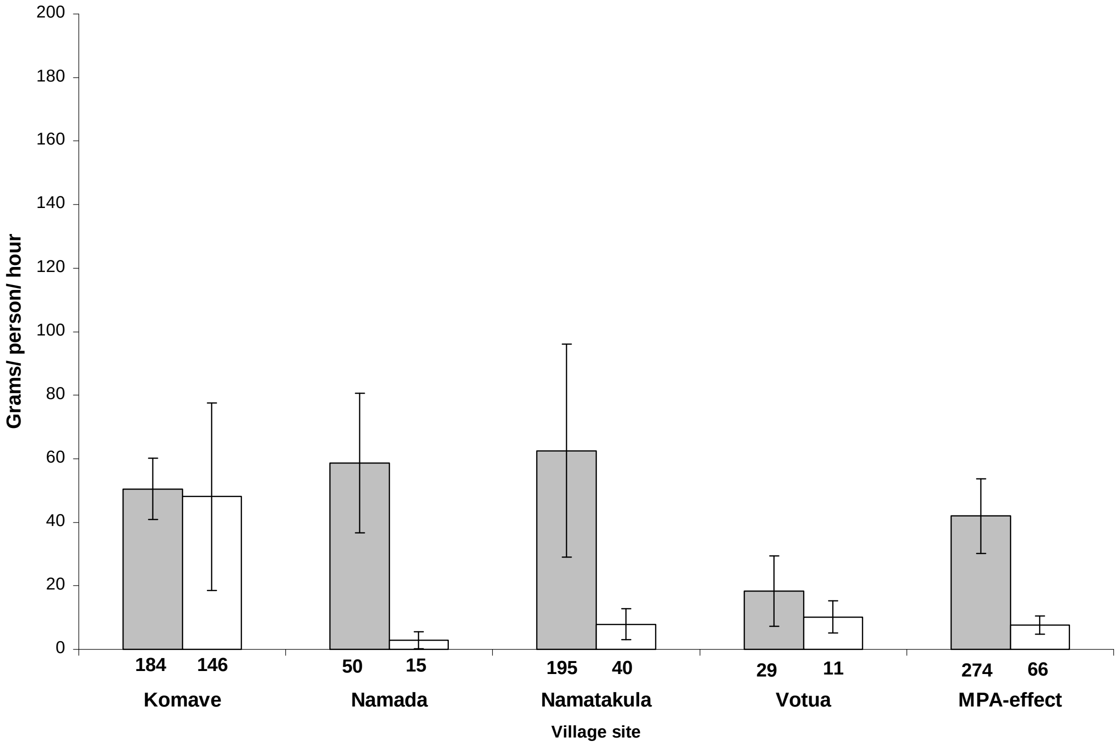


3.3 Analysis of Biomass

Length-weight relationships were obtained for 41 of the 55 species caught in this study, which amounted 74.5% of the species caught, or 98.5% of the total individual fish caught. Only four of the 55 species caught: *Epinephelus hexagonatus*, *Epinephelus merra*, *Lethrinus harak*, and *Lethrinus obsoletus*; had a sample size (n≥10) at each sampling location that was suitable for individual analysis at nearly all village sites. The same was true for the taxonomic groupings of *Epinephelus* spp. and *Lethrinus* spp. Of these species and groupings, none demonstrated normal distributions, making them unsuitable for a parametric ANOVA test.

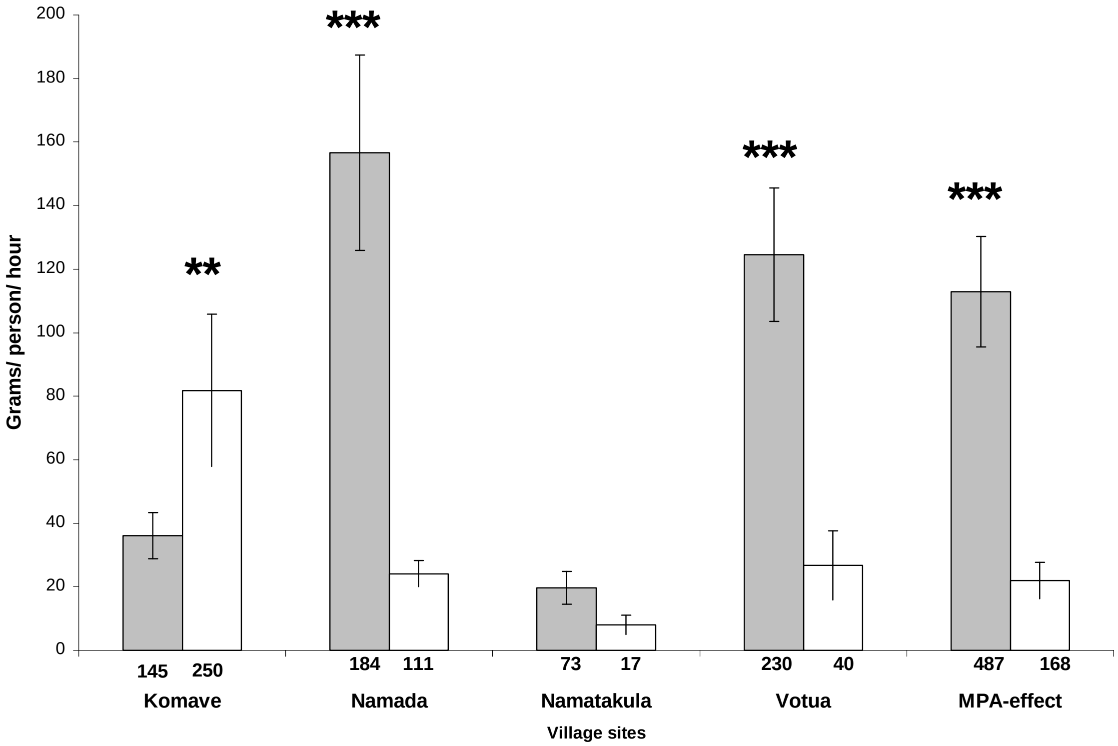
Mean biomass of *Epinephelus hexagonatus* (Figure 3.4.1) caught per person per hour was greater, but not significantly so, in the *tabu* areas of the Namada (*Wilcoxon rank-sum test*, *Z*= -1.29668, *P*= 0.1947), Namatakula (*Wilcoxon rank-sum Test*, *Z*= -1.49071, *P*= 0.1360), Votua (Wilcoxon rank-sum test, *Z*= 0.56671, *P*= 0.5709) and the combined "MPA-effect" group (*Wilcoxon rank-sum Test*, *Z*= -1.48868, *P*= 0.1366). Mean biomass was greater within the *tara* area at Komave compared to the adjacent *tabu* area, but was not significant (*Wilcoxon rank-sum Test*, *Z*= -0.66144, *P*= 0.5083).

Figure 3.3.1: Results of Wilcoxon rank-sum Test on mean biomass of *Epinephelus hexagonatus* for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars.



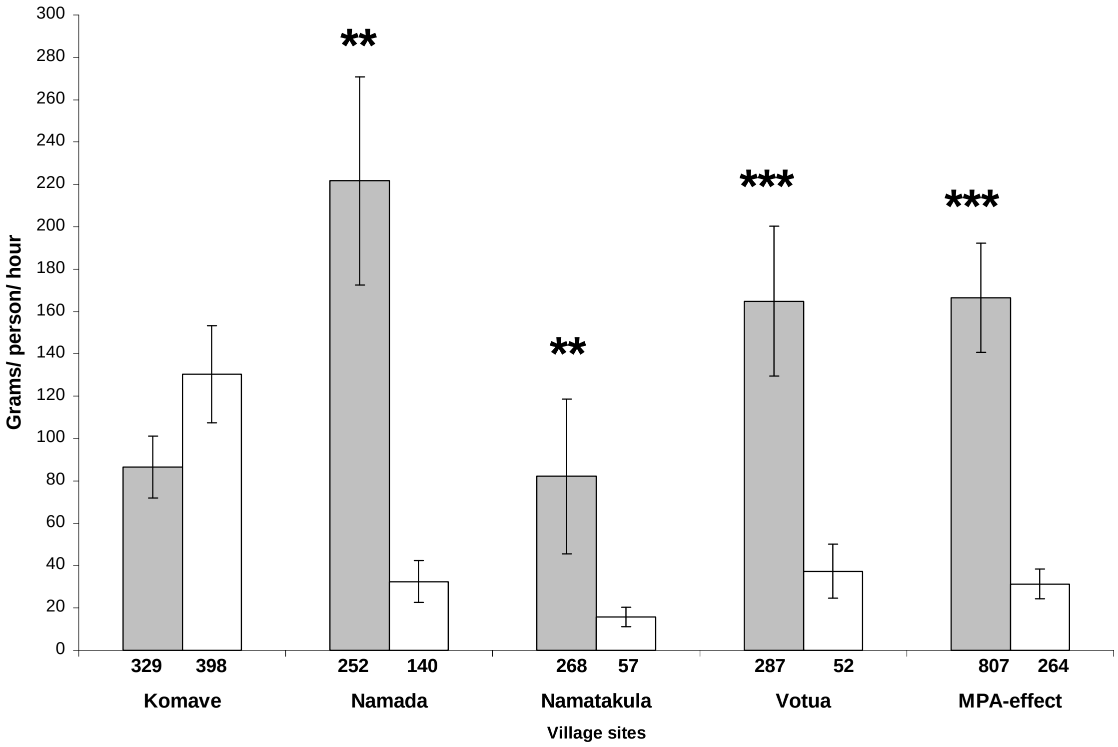
Highly-significant differences in mean biomass of *Epinephelus merra* (Figure 3.4.2) were exhibited within the *tabu* area of Namada (*Wilcoxon rank-sum Test, Z*= -2.63254, *P*= 0.0085), Votua (*Wilcoxon rank-sum Test*, *Z*= -2.87891, *P*= 0.0040) and the combined "MPA-effect" *(Wilcoxon rank-sum Test*, *Z*= -3.74230, *P*= 0.0002) grouping. In the Komave LMMA, mean biomass was significantly greater within the *tara* area as opposed to the *tabu* area (*Wilcoxon ranked-sum Test, Z*= 1.98431, *P*= 0.0472).

Figure 3.3.2: Results of Wilcoxon rank-sum Test on mean biomass of *Epinephelus merra* for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference (0.01≤*P*≤0.05), *** denotes a highly-significant difference (*P*< 0.01)



When grouped to the taxonomic level, the mean biomass caught of *Epinephelus* spp. (Figure 3.4.3) was significantly greater in the *tabu* area of Namada (*Wilcoxon rank-sum Test*, *Z*= -2.46270, *P*= 0.0138) and Namatakula (*Wilcoxon rank-sum Test, Z*= -2.08700, *P*= 0.0369) than their adjacent *tara* area. Greater mean biomass within the *tabu* area when compared to *tara* was highly-significant at both the Votua (*Wilcoxon rank-sum Test*, *Z*= -2.68373, *P*= 0.0073) and the combined "MPA-effect" grouping (*Wilcoxon rank-sum Test*, *Z*= -4.14941, *P*= <0.0001).

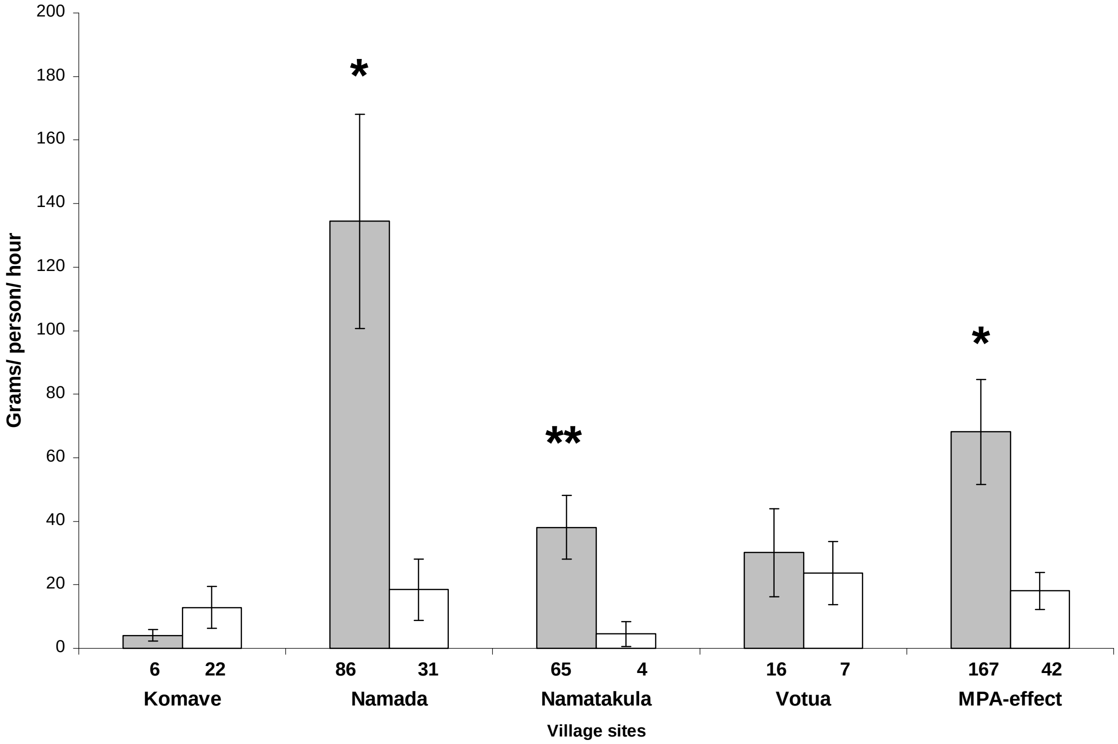
Figure 3.3.3: Results of Wilcoxon rank-sum Test on mean biomass of *Epinephelus* spp. for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference (0.01≤*P*≤0.05), *** denotes a highly-significant difference (*P*< 0.01)



Mean biomass of *Lethrinus harak* (Figure 3.4.4) was greater within the *tabu* area at the Votua site, but not significantly-greater. Namatakula had significantly-greater mean biomass caught within its *tabu* area (*Wilcoxon rank-sum Test*, *Z*= -2.08700, *P*= 0.0369). The same was true when sites were combined for an overall "MPA-effect" (*Wilcoxon rank-sum Test*, *Z*= -2.12090, *P*= 0.0339). Greater mean biomass within the *tabu* was marginally-significant when compared with its corresponding *tara* at the Namada site

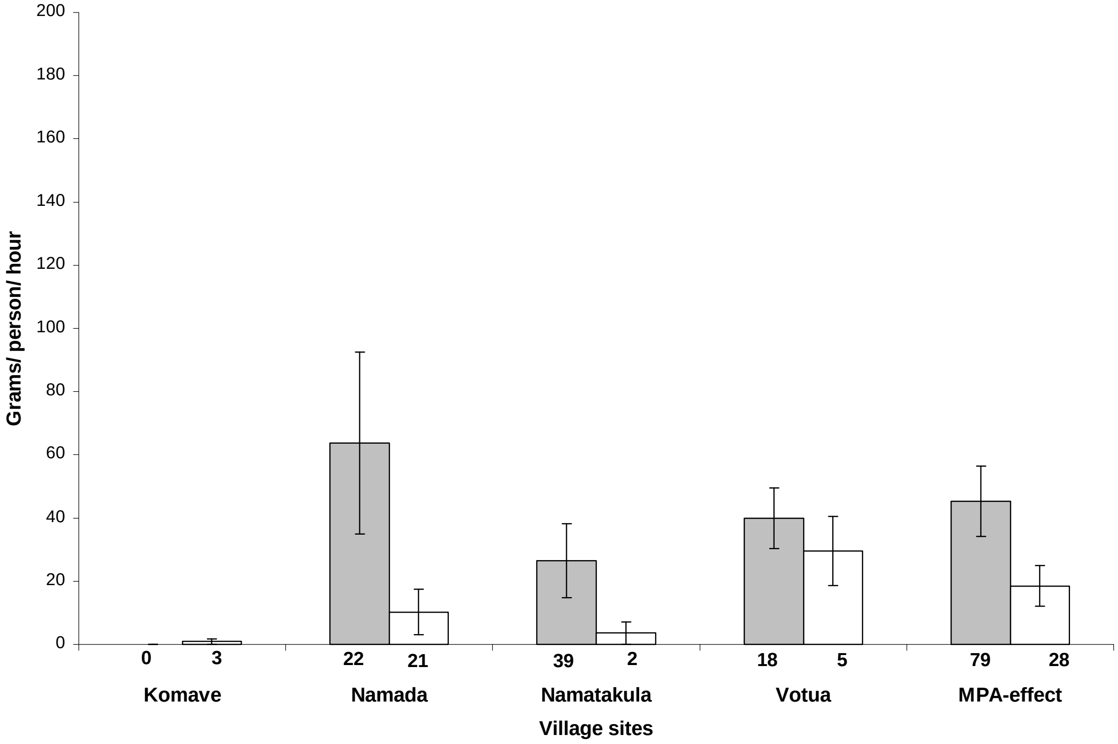
(*Wilcoxon rank-sum Test*, *Z*= -1.95318, *P*= 0.0508). Mean biomass caught within the *tara* area of the Komave site, while greater, was not significant.

Figure 3.3.4: Results of Wilcoxon rank-sum Test on mean biomass of *Lethrinus harak* for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. * denotes a marginally-significant difference (0.1>*P*>0.05), ** denotes a significant difference (*P*<0.05)



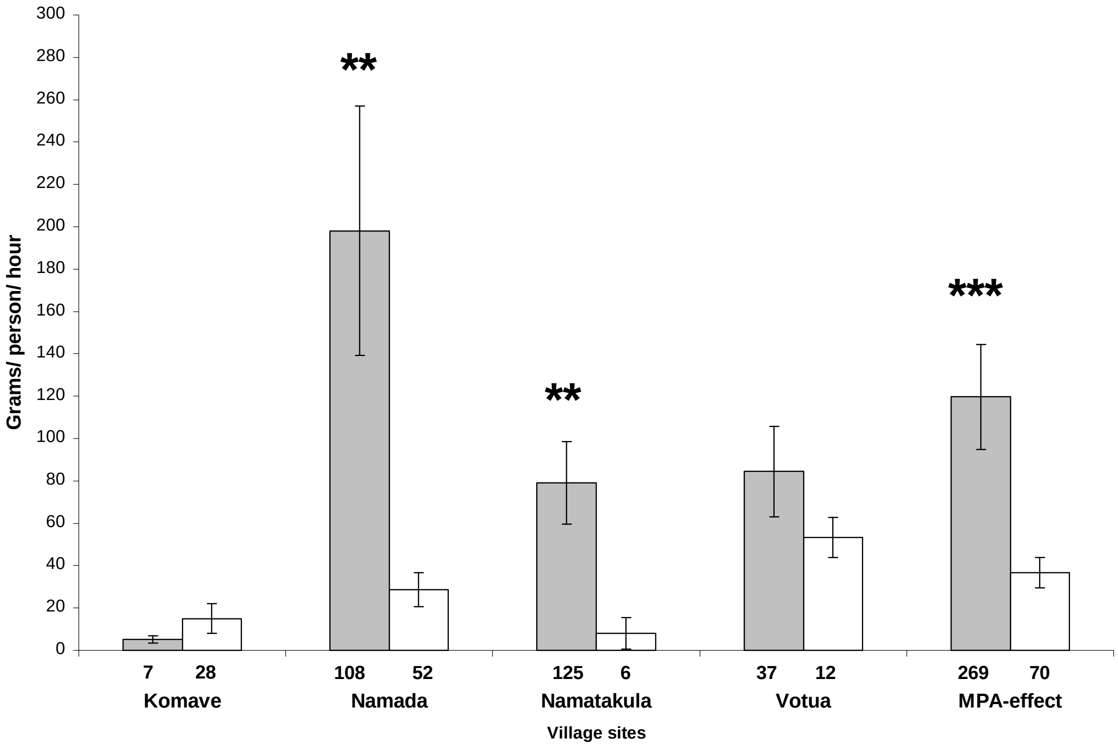
Mean biomass of *Lethrinus obsoletus* (Figure 3.4.5), while greater in the *tabu* areas of Namada, Votua, Namatakula, and "MPA-effect" sites, was not enough to be considered significant. At the Komave site, *tara* area mean CPUE was greater than that in the *tabu*, but was not significantly so.

Figure 3.3.5: Results of Wilcoxon rank-sum Test on mean biomass of *Lethrinus obsoletus* for the four village sites and the combined "MPA-effect" group. Number of days (hours in parentheses) (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars.



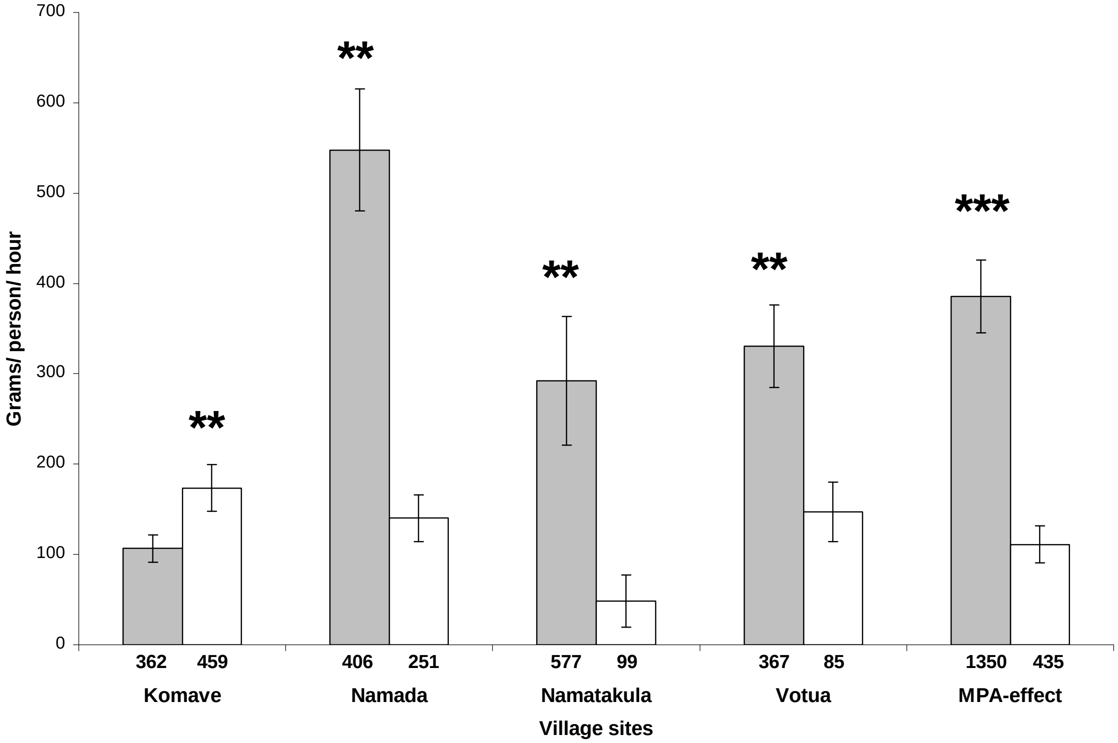
When grouped to the taxonomic level, mean biomass of *Lethrinus* spp. (Figure 3.4.6) was significantly greater in the *tabu* than the *tara* of Namatakula (*Wilcoxon rank-sum Test*, *Z*= -2.08700, *P*= 0.0369). Greater mean biomass in the *tabu* compared to the adjacent *tara* was highly-significant at Namada (*Wilcoxon rank-sum Test*, *Z*= -2.63254, *P*= 0.0085) and the combined "MPA-effect" (*Wilcoxon rank-sum Test*, *Z*= -2.94391, *P*= 0.0012) group. The *tara* area at Komave displayed greater mean CPUE than the *tabu* area, but was not significantly greater (*Wilcoxon rank-sum Test*, *Z*= 0.85236, *P*= 0.3940).

Figure 3.3.6: Results of Wilcoxon rank-sum Test on mean biomass of *Lethrinus* spp. for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference ($0.01 \leq P \leq 0.05$), *** denotes a highly- significant difference ($P < 0.01$)



Research yielded *W* values for 45 of the 55 species in the study, or approximately 81.8% of all species caught. This translated to 98.6% of all the fish caught in this study. Analysis of the total catch (Figure 3.4.7) showed greater mean biomass in the *tabu* areas of Namada, Namatakula, Votua, and the combined"MPA-effect" group. Significant differences between *tabu* and *tara* were exhibited at Namada (*Wilcoxon rank-sum Test*, Z= -2.55209, P= 0.0107), Namatakula (*Wilcoxon rank-sum Test*, Z=-2.08700, P= 0.0369), and Votua (*Wilcoxon rank-sum Test*, Z= -2.48855, P= 0.0128). These differences were highly-significant when combined for the overall “MPA-effect” (*Wilcoxon rank-sum Test*, *Z*= -4.24336, *P*= <0.0001). Komave was the only site where mean CPUE was significantly greater within the *tara* area rather than the *tabu* (*Wilcoxon rank-sum Test*, *Z*=1.98431, *P*= 0.0472).

Figure 3.3.7: Results of Wilcoxon rank-sum Test on mean biomass of the total catch for the four village sites and the combined “MPA-effect” group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference 0.01≤*P*≤ 0.05), *** denotes a highly- significant difference (*P*< 0.01)

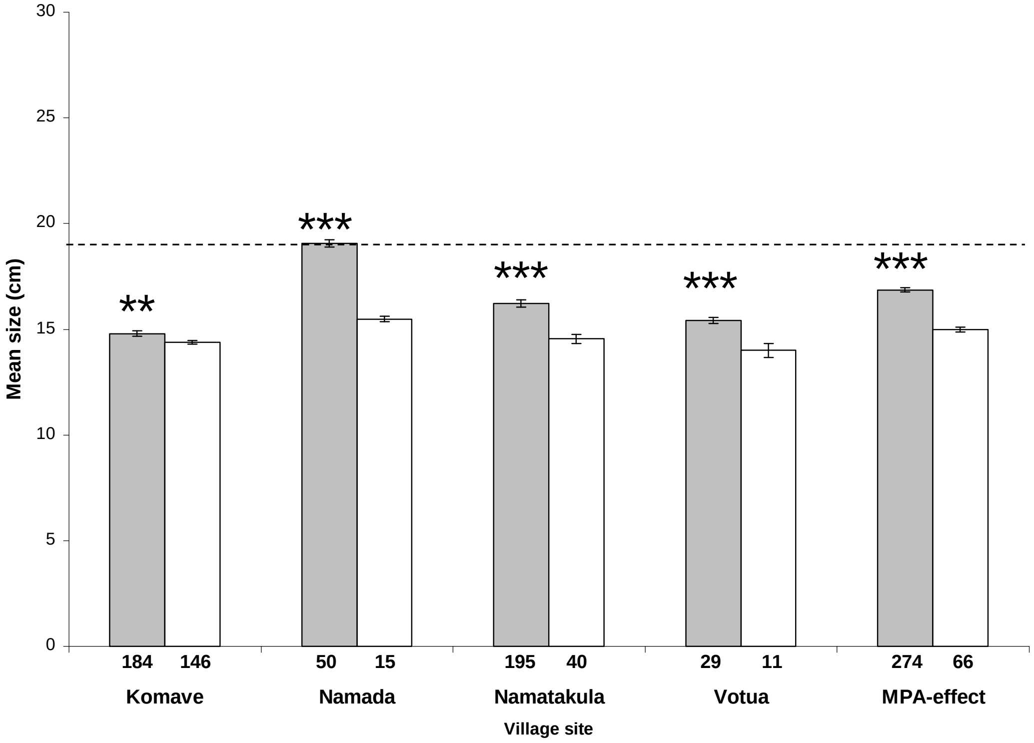


3.4 Mean Size Analysis

Of the 55 species caught, only four: *Epinephelus hexagonatus*, *Epinephelus merra*, *Lethrinus harak*, and *Lethrinus obsoletus*; had a sample size ($n \geq 10$) suitable for size analysis at nearly all village and combined village sites. The same was true for the taxonomic groupings of *Epinephelus* spp.and *Lethrinus* spp. Of these species and groupings, none demonstrated normal variance, making them unsuitable for a parametric ANOVA test.

The non-parametric Wilcoxon rank-sum Test demonstrated significantly greater mean size of *Epinephelus hexagonatus* (Figure 3.3.1) within the *tabu* area at Komave compared to the adjacent *tara* (*Wilcoxon rank-sum Test, Z*= 2.03004, *P*= 0.0423). Greater mean size of *E. hexagonatus* in the *tabu* compared to the adjacent *tara* was highly-significant at Namada (*Wilcoxon rank-sum Test*, *Z*= -12.4714, *P*= 0.0000), Namatakula (*Wilcoxon rank-sum Test*, *Z*= -4.34333, *P*= <0.0001), Votua (*Wilcoxon rank-sum Test*, *Z*= -3.60569, *P*= 0.0003) and the combined sites for “MPA-effect” (*Wilcoxon rank-sum Test*, *Z*= -9.26940, *P*= 0.0000). Differences ranged across sites from lower than 0.5 cm at Komave to over 3.5 cm at Namada. Only the *tabu* at Namada village had a mean size greater than or equal to the length at maturity (*Lm*) for *E. hexagonatus*. Compared to the Komave *tabu*, mean sizes were greater within the *tabu* areas of Namada, Namatakula, Votua and the combined “MPA-effect” group, with differences of 4.3 (28.8%), 1.4 (9.7%), 0.6 (4.2%), and 2.1 (14%) centimetres, respectively. Mean sizes at the Namada and combined “MPA-effect” sites were also greater within their *tara* areas when compared with the Komave *tabu*, with differences of 0.7 (5.8%) and 0.2 (1.3%) centimetres.

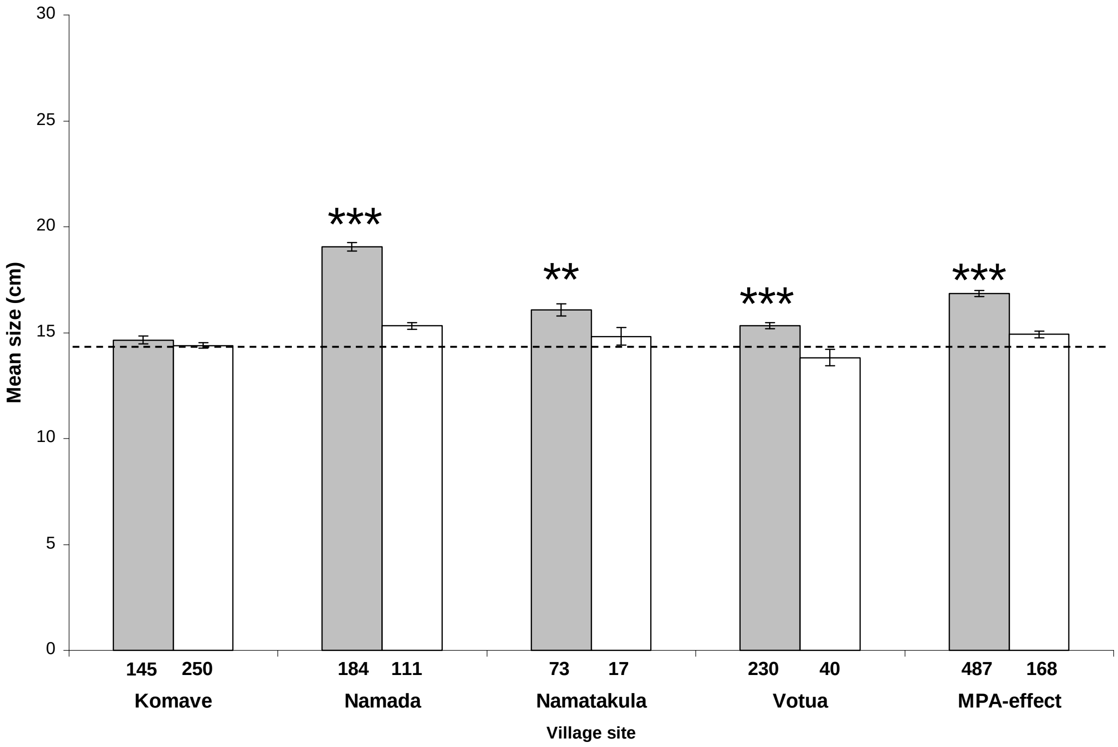
Figure 3.4.1: Results of Wilcoxon rank-sum Test on mean size (cm) of *Epinephelus hexagonatus* for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference ($0.01 \leq P \leq 0.05$), *** denotes a highly-significant difference ($P < 0.01$), **----** denotes the length at maturity (*Lm*)



Mean size of *Epinephelus merra* (Figure 3.3.2) was significantly greater within the *tabu* area compared to *tara* area at Namada (*Wilcoxon rank-sum Test*, *Z*= -10.6054, *P*= 0.0000), Namatakula (*Wilcoxon rank-sum Test, Z*= -2.07883, *P*= 0.0376), Votua (*Wilcoxon rank-sum Test*, *Z*= -3.22598, *P*= 0.0013) and the combined-sites for "MPA-effect" (*Wilcoxon rank-sum*, *Z*= -3.22598, *P*= <0.0001). Namada, Namatakula, and the

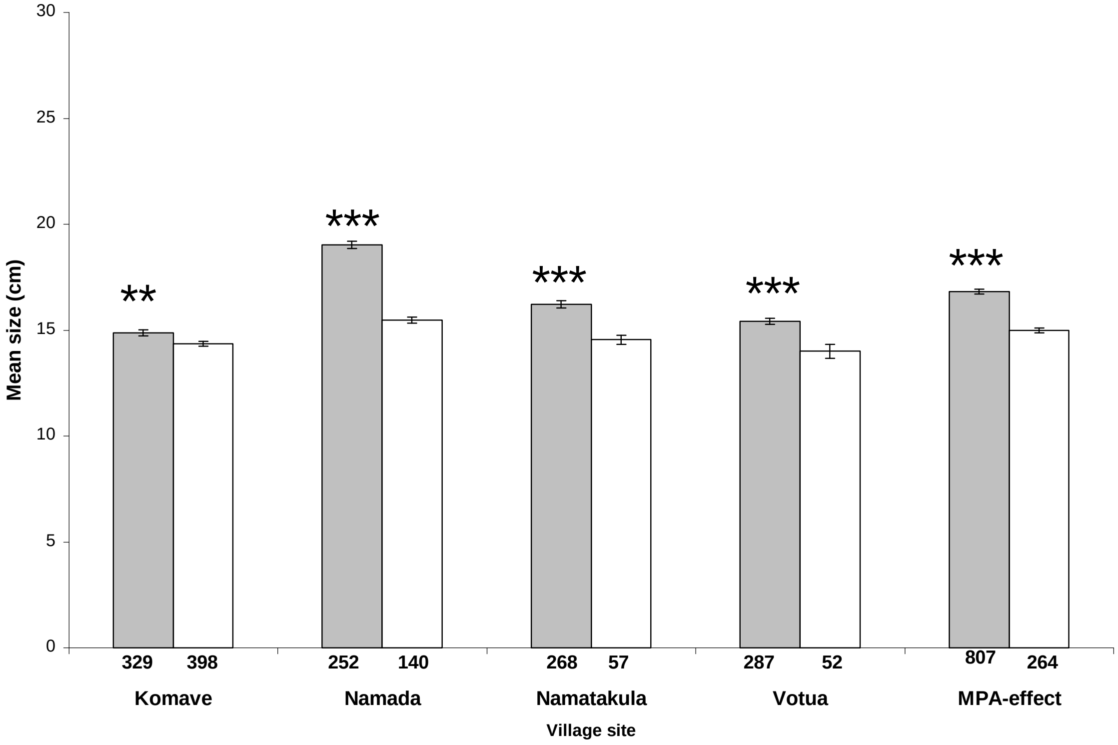
"MPA-effect" group all had mean sizes greater than or equal to the *Lm* of *Epinephelus merra* within their respective *tabu* area. Mean sizes were greater within the *tabu* areas of Namada, Namatakula, Votua and "MPA-effect" sites than in the Komave *tabu*, with differences of 4.4 (30.1%), 1.4 (9.7%), 0.7 (4.6%), 2.2 (15%) centimetres, respectively. Mean sizes of Namada, Namatakula and the "MPA-effect" group were also greater within their *tara* areas when compared with the Komave *tabu*, with differences of 0.7 (4.6%), 0.2 (1.1%), and 0.3 (1.8%) centimetres. Mean size for fish in the *tabu* areas of all sites was greater than the SSM.

Figure 3.4.2: Results of Wilcoxon rank-sum Test on mean size (cm) of *Epinephelus merra* for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference ($0.01 \leq P \leq 0.05$), *** denotes a highly-significant difference ($P < 0.01$), ---- denotes the length at maturity (*Lm*)



When combined to the taxonomic level, mean size of *Epinephelus* spp. caught (Figure 3.3.3) was significantly larger within the *tabu* area of Komave (*Wilcoxon rank-sum Test*, *Z*= 2.03004, *P*= 0.0424), Namada (*Wilcoxon rank-sum Test*, *Z*= -12.4763, *P*= 0.0000), Namatakula (*Wilcoxon rank-sum* Test, *Z*= -4.34333, *P*= <0.0001), Votua (*Wilcoxon rank-sum Test*, *Z*= -3.68072, *P*= 0.0002), and the combined "MPA-effect" group (*Wilcoxon rank-sum Test*, *Z*= -9.13136, *P*= 0.0000) when compared with the corresponding *tara* areas. Compared to the Komave *tabu*, mean sizes were greater within the *tabu* areas of Namada, Namatakula, Votua and the combined "MPA-effect" group, with differences of 4.1 (27.9%), 1.3 (9%), .5 (3.6%), and 2.0 (13.1%) centimetres, respectively. Mean sizes at the Namada site and for the "MPA-effect" group were also greater within their *tara* areas when compared with the Komave *tabu*, with differences of 0.6 (4%) and 0.1 (.8%) centimetres.

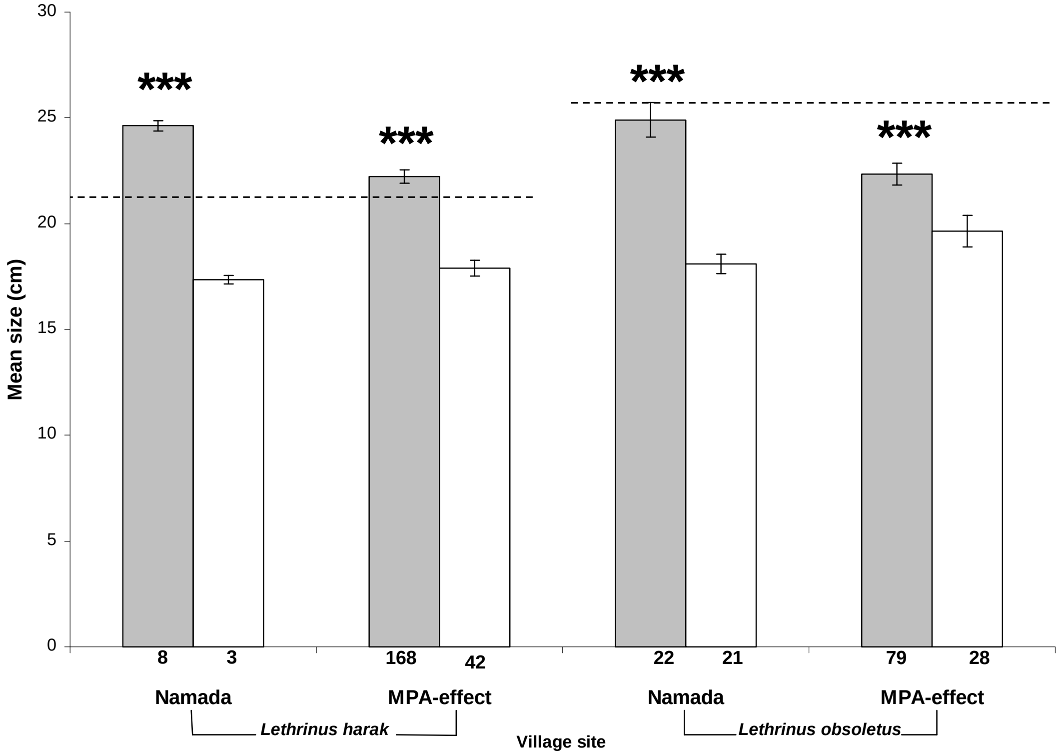
Figure 3.4.3: Results of Wilcoxon rank-sum Test on mean size (cm) of *Epinephelus* spp. for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference (0.01≤*P*≤0.05), *** denotes a highly-significant difference (*P*<0.01)



Insufficient size of *n* at all sites except for Namada and the combined “MPA-effect” group led to their exclusion in analysis for both *Lethrinus harak* and *Lethrinus obsoletus*. Mean size of *Lethrinus harak* (Figure 3.3.4) caught was significantly greater within the *tabu* than the within the *tara* of Namada (*Wilcoxon rank-sum Test*, *Z*= -8.25546, *P*= <0.0001), and the combined “MPA-effect” group (*Wilcoxon rank-sum Test*, *Z*= -5.46044, *P*= <0.0001). Mean size of *Lethrinus harak* was greater than or equal to the species’ *Lm* at both Namada and the “MPA-effect” group *tabu* areas.

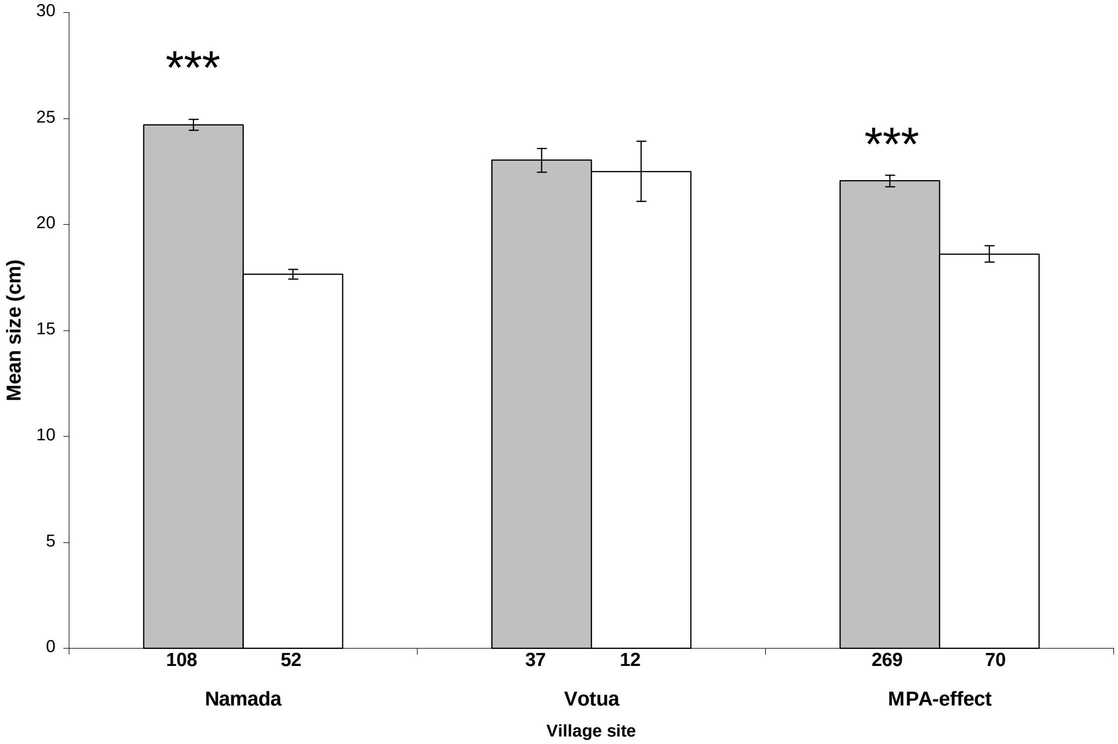
Lethrinus obsoletus also exhibited significantly greater mean size (Figure 3.3.4) within *tabu* area at Namada (*Wilcoxon rank-sum Test*, *Z*= -5.05135, *P*= <0.0001) and the “MPA- effect” group (*Wilcoxon rank-sum Test*, *Z*= -2.87164, *P*= 0.0040).

Figure 3.4.4: Results of Wilcoxon rank-sum Test on mean sizes (cm) of *Lethrinus harak* and *Lethrinus obsoletus* for the Namada site and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. ** denotes a significant difference (0.01≤*P*≤ 0.05), *** denotes a highly-significant difference (*P*< 0.01), ---- denotes the length at maturity (*Lm*)



For the *Lethrinus* spp. grouping, only Namada, Votua, and the "MPA-effect" group had a sufficient sample size for analysis (Figure 3.3.6). Mean size of fish caught within the *tabu* area was significantly greater at Namada (*Wilcoxon rank-sum Test*, *Z*= -10.0267, *P*= 0.0000), and for the combined "MPA-effect" group (*Wilcoxon rank-sum Test*, *Z*= -5.80491, *P*= <0.0001) compared to their corresponding *tara*.

Figure 3.4.5: Results of Wilcoxon rank-sum Test on mean size (cm) of *Lethrinus* spp. for the Namada and Votua village sites, and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars. *** denotes a highly-significant difference (*P*< 0.01)

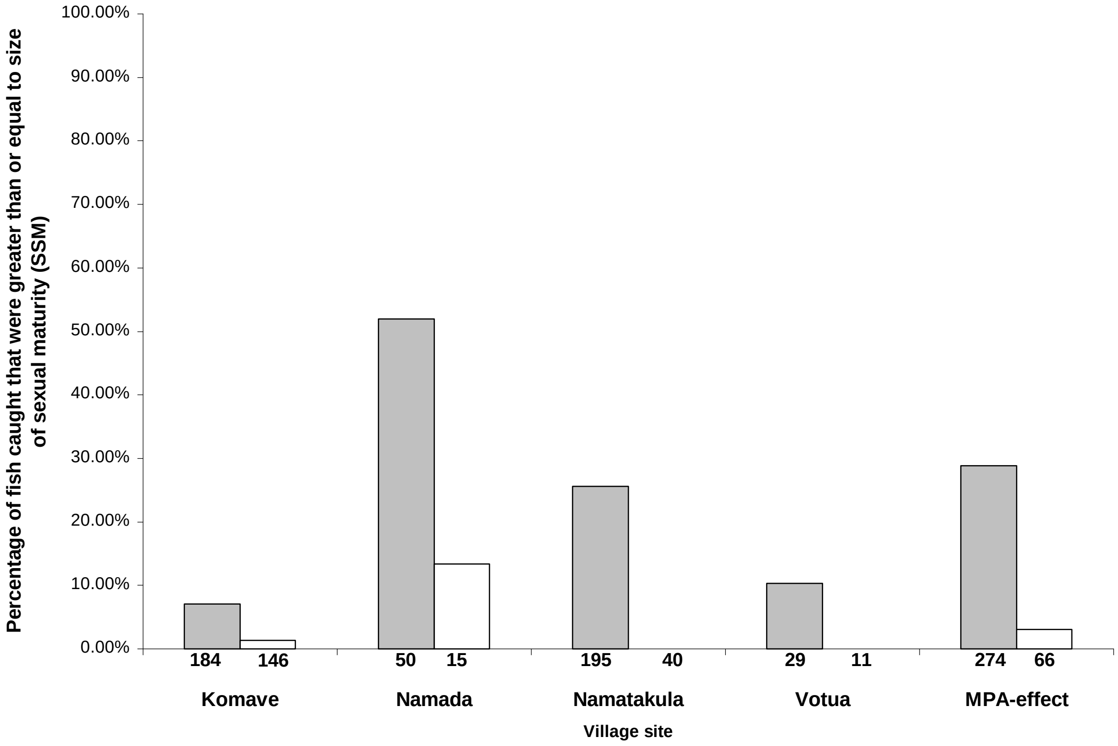


3.5 Percentage of catch greater than or equal to size of sexual maturity (SSM)

Comparisons of fish at SSM included all the species and taxonomic groupings previously analysed for CPUE and mean size due to their adequate sample size ($n \geq 10$). An additional comparison was made between areas of the total number of fish for whom SSM was known.

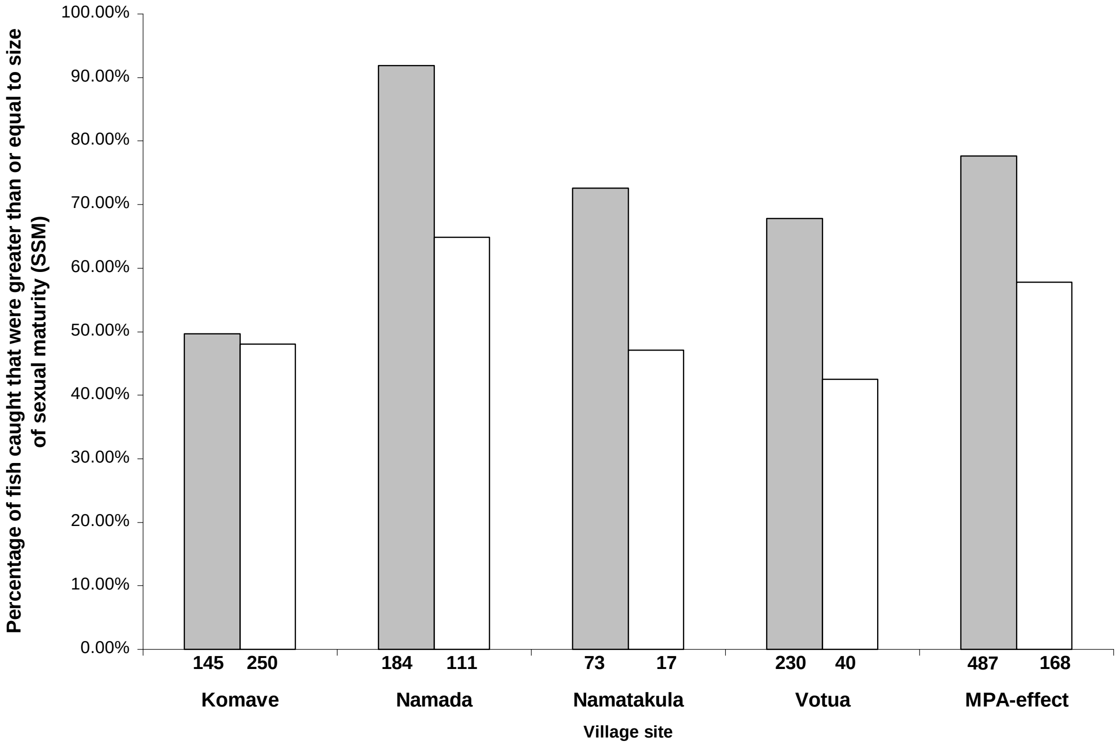
At all five sites: Komave, Namada, Namatakula, Votua, and the combined "MPA-effect;" catch of *Epinephelus hexagonatus* exhibited a greater percentage of individuals greater than or equal to SSM when each site's *tabu* area was compared with its corresponding *tara* area (Figure 3.5.1). Differences between these areas ranged between sites; from 5.7% at Komave to 38.8% at Namatakula. All sites except Komave exhibited differences of over 10%, and with the exception of Votua, differences were over 25%. At Komave, 90% of the catch for both *tabu* and *tara* areas was below *Lm*.

Figure 3.5.1: Percentage of *Epinephelus hexagonatus* greater than or equal to SSM for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars.



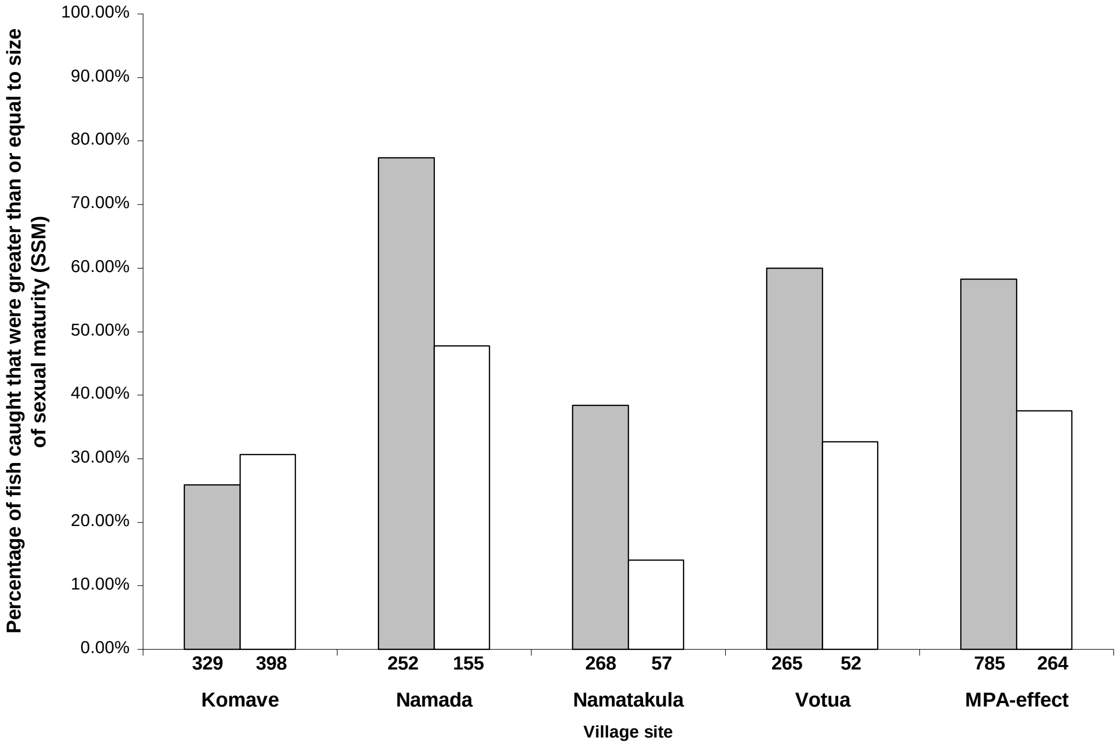
Epinephelus merra at all five sites exhibited a greater percentage of individuals caught that were greater than or equal to SSM within the *tabu* areas when compared to their respective *tara* (Figure 3.5.2). Percentage differences between *tabu* and *tara* areas ranged from 2.6% at Komave to as much as 53.8% at Namada. All sites, except for Komave, demonstrated differences of at 17% or more. The *tabu* areas of Namada, Namatakula, Votua and "MPA-effect" sites displayed catch with a greater percentage of fish greater than or equal to SSM of the *tara* area of Komave; a difference of 11% or greater, and with Votua excluded, was a difference of at least 26% or more. Similarly, the *tara* areas of Namada, Namatakula, and the combined "MPA-effect" group had greater percentages of fish greater than or equal to SSM for both the *tabu* and *tara* area of Komave.

Figure 3.5.2: Percentage of *Epinephelus merra* greater than or equal to SSM for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars.



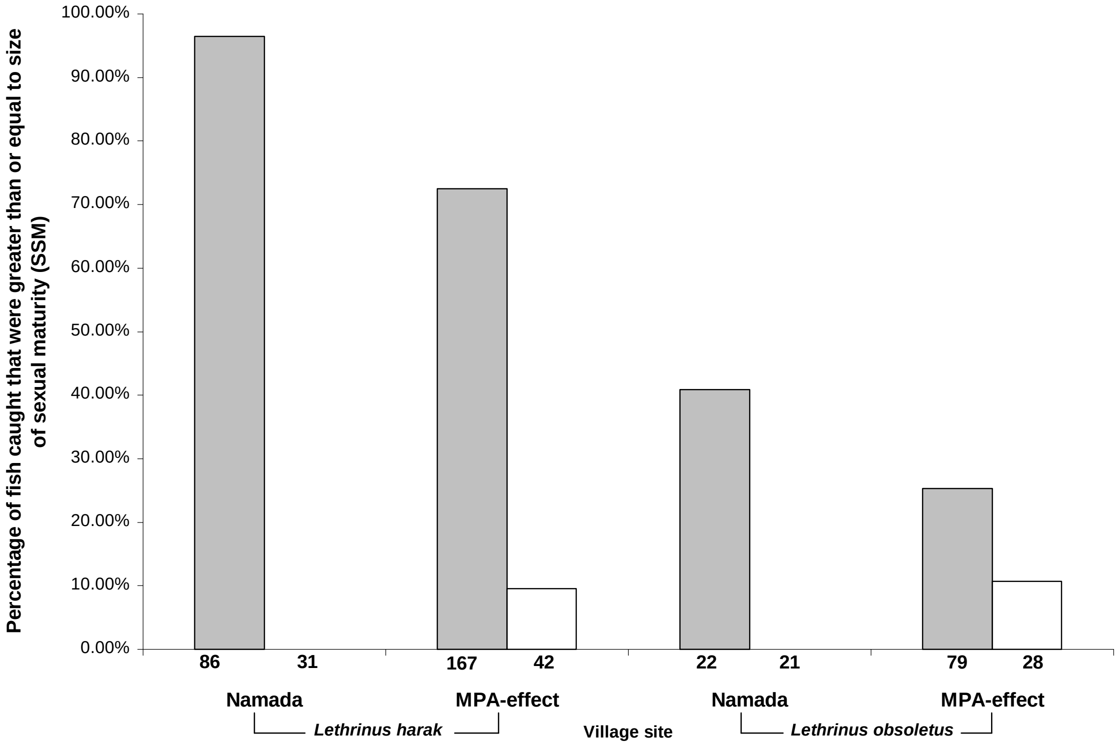
Epinephelus spp. (Figure 3.5.3) showed a greater percentage of individuals greater than or equal to SSM in the *tabu* area of the Namada, Namatakula, Votua, and "MPA-effect" group, with differences between *tabu* and their corresponding *tara* area being 48.9%, 24.6%, 19.2%, and 27.7%, respectively. In contrast, catch at the Komave LMMA exhibited a greater percentage of fish greater than or equal to SSM in the *tara* area when compared to the adjacent *tabu*, with both areas exhibiting catch of 80% or more of immature size. The *tabu* areas of Namada, Namatakula, Votua and the combined "MPA-effect" grouping displayed catch with a greater percentage of fish greater than or equal to SSM compared to the *tabu* area of Komave; a difference of at least 16% or greater. Similarly, the *tara* areas of Namada, Namatakula, and the combined "MPA-effect" group had greater percentages of fish greater than or equal to SSM compared to both the *tabu* and *tara* area of Komave.

Figure 3.5.3: Percentage of *Epinephelus* spp. greater than or equal to SSM for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars.



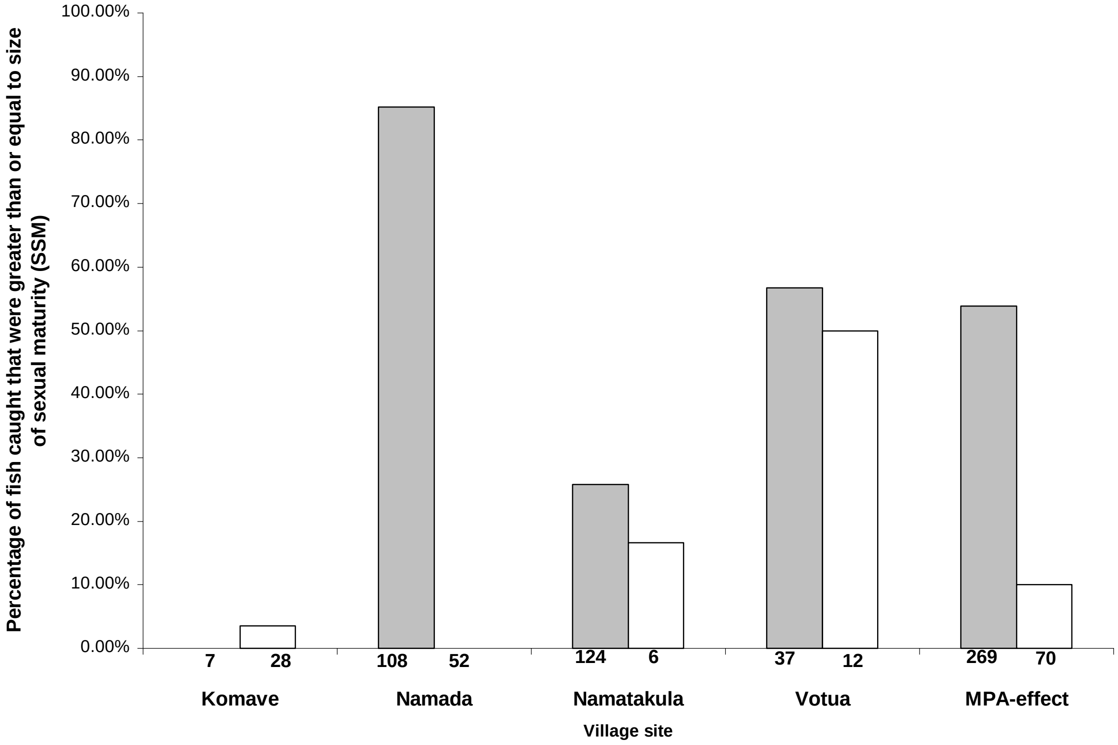
Due to insufficient sample size, analysis of *Lethrinus harak* and *Lethrinus obsoletus* was limited to Namada and the combined "MPA-effect" group (Figure 3.5.4). A greater percentage of *Lethrinus harak* greater than or equal to SSM composed the catch in the *tabu* areas of both Namada and the "MPA-effect" group, with differences of 96.5% and 62.9%, respectively. A greater percentage of *Lethrinus obsoletus* greater than or equal to SSM was caught within the *tabu* areas of Namada and the combined "MPA-effect" group, with differences of 40.9% and 14.6%, respectively.

Figure 3.5.4: Percentage of *Lethrinus harak* and *Lethrinus obsoletus* greater than or equal to SSM for the Namada site and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars.



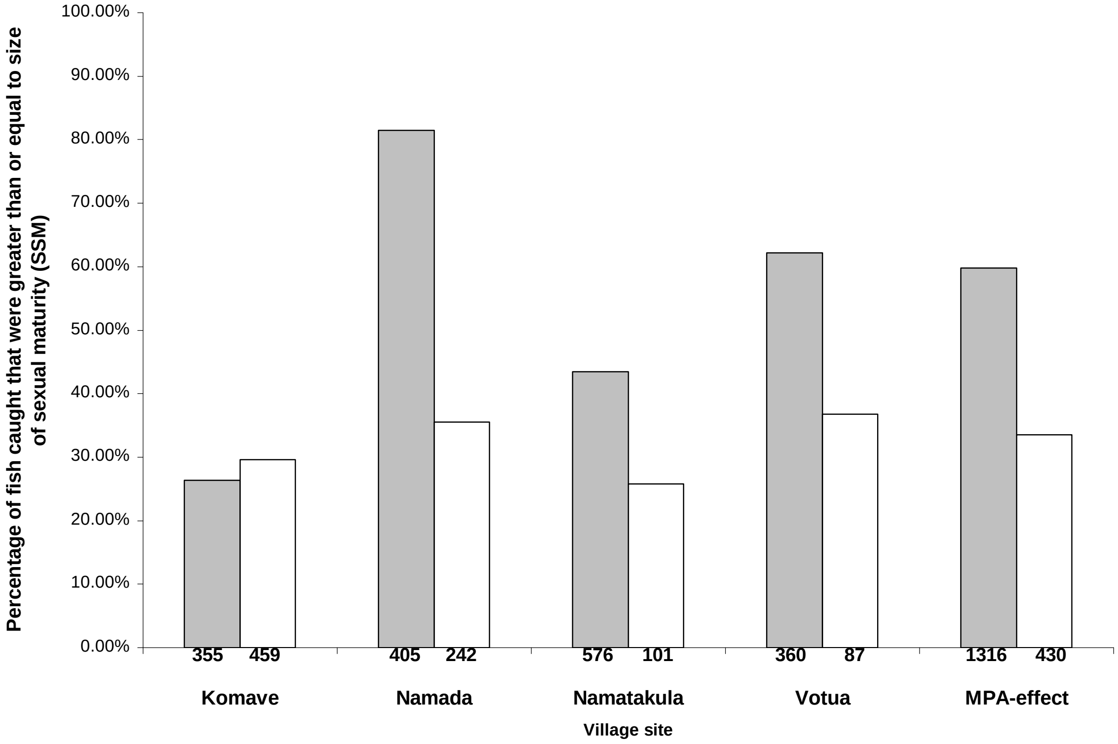
Sample sizes of *Lethrinus* spp. (Figure 3.5.5) for Namada, Votua, and the combined "MPA-effect" group were the only ones with an *n* greater than or equal to ten. Both Namada and the combined "MPA effect" group demonstrated a greater proportion of fish caught within the *tabu* being of SSM, with differences of 85.2% and 52.8%, respectively. On the contrary, the *tara* area of the Votua LMMA had a 1.6% greater amount of fish caught that were of SSM when compared to the *tabu* area.

Figure 3.5.5: Percentage of *Lethrinus* spp. greater than or equal to SSM for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars.



Research yielded SSM values for 31 of the 55 species in the study, or approximately 56.4% of all species caught. This translated to 97.4% of all the fish caught in this study. When these species were combined together for a total SSM comparison (Figure 3.5.6), Namada, Namatakula, Votua, and the combined "MPA-effect" group all showed a greater percentage of fish greater than or equal to SSM within their respective *tabu* as opposed to the adjacent *tara*. Percentage differences between areas at each site varied from 16.9% at Votua, to as high as 58.2% at Namada. Komave was the only site where percentage of catch of *Lm* was greater within the *tara* area, with a difference of 3.6%. Both *tabu* and *tara* at Komave had approximately 80% or more of the catch consisting of individuals of immature size. Still, *tabu* areas of Namada, Namatakula, Votua, and the "MPA-effect" group exhibited greater percentages of fish that were greater than or equal to SSM compared to the Komave *tara* area, with differences ranging from 21.4-60.5% higher. Additionally, *tara* areas of Namatakula, Namada, Votua, and the combined "MPA-effect" group also demonstrated greater percentages of fish that were greater than or equal to SSM when compared to *tara* area at Komave.

Figure 3.5.6: Percentage of fish of all species with known SSM that were greater than or equal to SSM for the four village sites and the combined "MPA-effect" group. Number of fish caught (*n*) are listed below each area column. *Tabu* areas are displayed with shaded bars and *tara* areas with clear bars.



Chapter 4: Discussion

The results of this study highlight several major findings. Primarily, the use of small no- take *tabu* areas as a management tool can have positive effects on the mean CPUE, size, biomass, and percentage of sexually mature target fish species within a LMMA.

4.1 Catch Composition

Initial inspection of the catch in *tabu* and *tara* areas of each respective site reveals that the majority of catch at all sites appears to be composed primarily of *Epinephelus* spp., followed by a mixed proportion of the other species groupings. While it is difficult to draw conclusions without statistical comparisons between area sites, the findings suggest a more heterogeneously-distributed community composition at the three sites where MPA's are still being enforced than at Komave. In Kenya, McClanahan *et al.* (2010) found less variation in catch community composition of reef fish among fishing grounds than in adjacent closures, and suggested that fishing had homogenised catch composition. A similar occurrence may have taken place at Komave, where fishing pressure exerted on both the *tabu* and adjacent *tara* may have contributed to the more homogenised catch composition apparent in both areas. Since many fishery-dependent assessments report modest changes, these small differences in catch composition may reflect larger ecological differences (McClanahan *et al.* 2010).

Understanding changes in the composition and value of the catch is important to the proper analysis of multispecies CPUE (Jennings and Polunin, 1995*c*). While *Epinephelus* spp. and total species CPUE in the Komave *tara* were both nearly equivalent or greater than that of the *tabu* in other villages, this could have attributed to the catch composition at the Komave *tara,* which was composed of 86% *Epinephelus* spp. In contrast, *Epinephelus* spp. composed 59% of the catch in the *tabu* of the protected sites when combined for an "MPA-effect". CPUE of *Epinephelus* spp. alone in the *tara* area of Komave was also relatively equivalent when compared to other the *tabu* areas of protected sites, but was not significantly greater than its own *tabu*.

Of the three *Epinephelus* spp. species caught in Komave's *tara*, 62.7% were *Epinephelus merra*. This species has the lowest SSM (Appendix A) of any of the target *Epinephelus* and *Lethrinus* species. *E. merra* is also known to exhibit a relatively high growth rate (Pothin *et al.* 2004), and a has a high rate of resilience to fishing pressure, with a minimum population doubling time of less than 15 months (Froese and Pauly, 2010). It is plausible that the shift in pressure away from the *tara* to the no longer enforced *tabu* area alleviated fishing pressure in the *tara* enough to allow stocks of species in the *Epinephelus* spp. grouping to rebound over the following months. Unsworth *et al.* (2007) found *Epinephelus merra* to be the most abundant fish in their study comparing abundance of different species of groupers in a small MPA with adjacently fished areas after 5 years of protection. *E. merra* was one of the only species in the study to not exhibit significant differences between sites. They suggested that the population doubling time of the species might have been responsible for increased spillover from the MPA, and hence the lack of differences observed between sites in relation to other longer-lived species.

The second most caught species (36%) at Komave's *tara* was *Epinephelus hexagonatus*, which coincidently has the second lowest SSM (Appendix A) of any of the main target *Lethrinus* or *Epinephelus* spp. Rapid initial growth has also been suggested for this species, but requires further confirmation (Kulbicki *et al.* 2008). Indeed, the importance of these fish to island fisheries has been attributed to their abundance in shallow waters (Heemstra and Randall, 1993), yet studies on their basic ecology are limited (Unsworth *et al.* 2007).

On the other hand, *Lethrinus* spp. only comprised 2% of the total catch in the Komave *tara*, and had abundance of approximately the same levels as those of the *tara* areas of the other protected sites. In contrast, *Lethrinus* spp. composed 20% of the total catch in the *tabu* area of "MPA-effect" site grouping. *Lethrinus* spp. are one of the most important food fishes in Fiji, with a high average landed price (around FJD $3.50 to $4 kg $^{-1}$) (Teh *et al.* 2009), and a high preference for consumption among fishers; more so than the small bodied *Epinephelus* spp. caught in this study (personal observation). Indeed, Rawlinson *et al.* (1995) found *Lethrinus harak* to be more targeted than *Epinephelus* spp. on fringing reefs, and the most targeted of all species surveyed along the shoreline and inside the lagoon (<10m) among rural coastal villages around Viti Levu. Additionally, known maximum observed sizes are larger, and growth rates comparatively slower for target *Lethrinus* spp. (Carpenter and Allen, 1989) than those of known for the most commonly caught *Epinephelus* spp.; such as *Epinephelus hexagonatus* and *Epinephelus merra* (Heemstra and Randall, 1993; Kulbicki *et al.* 2008).

Theoretical analyses have suggested that the response or vulnerability of a species to exploitation may depend upon their life history; with large, slow growing and late maturing species suffering greater declines in population for a given mortality rate (Pope *et al.* 2000). Growth, age at maturity, reproductive output, and natural mortality are all negatively correlated with fish maximum size, which is closely related to observed maximum size (Charnov, 1993; Sadovy, 1996; Pauly, 1998). For data deficient species, like most tropical reef fish, easily measured life history parameters such as observed maximum size may be used to make preliminary assessments of vulnerability.

The vulnerability of late-maturing and larger species to fishing suggests that small and early maturing species would increase in relative abundance in an intensively exploited multispecies community. In addition to more advantageous life history traits, smaller species may also benefit from lower fishing mortality simply because they are less desirable and may escape through fishing gears such as meshes in nets and traps. Under increasing fishing pressure, community structure is expected to shift from initial decreases in abundance of larger individuals of all target species and larger individuals as a proportion of the total abundance (Haedrich and Barnes, 1997), to ultimately being dominated by smaller individuals and smaller species. These types of changes in community structure are often interpreted from changes in catch (Jennings *et al.* 2001). In Jamaica, Koslow *et al.* (1988) found that the largest fish caught with traps, the main fishing method, virtually disappeared from catches at all heavily fished study sites in a 20-year period while catch of less desirable species was able to persist. In the Philippines, Russ and Alcala (1998*a*,*b*) examined reefs opened and closed to fishing following management successes and failures, and found that large species, including species of emperor (*Lethrinidae*) and grouper (*Serranidae,* Subfamily: *Epinephelidae*) with slow life histories declined in abundance most rapidly when fished and recovered the slowest following protection. Additionally, larger species with slower life histories were less abundant on reefs subject to heavy fishing intensity compared to lightly fished and non-fished reefs in the Seychelles (Jennings *et al.* 1996) and Fiji (Jennings and Polunin, 1996*a*; Jennings *et al.* 1999b).

The findings of this study are indicative of these types of shifts, with faster growing, maturing, and reproducing *Epinephelus* spp. species making up more of the catch than the slower growing, maturing, and reproducing *Lethrinus* spp. This trend is exhibited the most in areas open to fishing. At Komave, the trend is more pronounced in the recently opened *tabu* area, and as mentioned earlier, may be due to a shift in fishing pressure formerly allocated to the *tara* area. Even so, catch of *Epinephelus* spp. at both areas composes 26% or more of the total catch than that of the *tabu* or *tara* areas of the other three sites when combined in the "MPA-effect" group. Similarly, less of the catch in the Komave *tabu* was composed of *Lethrinus* spp. compared to its *tara* area. Both areas were still lower in composition than either *tabu* or *tara* of the "MPA-effect" grouping, with differences of 9% or more. The same is true of the other species category, with differences of 13% or more.

Comparisons of vulnerability between different genera, or between different populations within a species, are more appropriate than between those of different taxonomic groups, since they eliminate the amount of variables species have in common, and are more likely to yield evolutionarily independent data (Harvey and Pagel, 1991). Still, the relevance of different levels of vulnerability among target species and its influence on the catch composition such as these are important to consider in assessments.

4.2 Mean CPUE

CPUE showed a general trend toward greater abundance in the *tabu* area of the three sites where protection has been maintained. Even on an individual species level, *Epinephelus merra*, *Lethrinus harak*, and *Lethrinus obsoletus* all showed significantly greater levels of CPUE within the *tabu* area of at least one or more sites still enforcing their *tabu* area. Differences were demonstrated among individual species even with low *n* values present in the study, and are consistent with other studies involving CPUE as a measure of relative abundance (Attwood, 2000; Kaunda-Arara and Rose, 2004). These findings suggest that fish abundance in *tabu* areas is greater than that of fish in adjacent *tara* areas, and thus are providing protection to target fish taxa.

E. merra was the most abundant fish accounted for in this study, and subsequently had the most significant differences between *tabu* areas among sites, as well as significantly greater abundance in the *tara* area of Komave, the only site whose *tabu* is no longer protected. Research conducted in Indonesia by Unsworth *et al.* (2007) comparing a no-take area with three areas subject to varying fishing intensities revealed that *Epinephelus merra*, the most abundant species reported in the study, only displayed significant differences in abundance between the no-take and the most heavily fished area. This suggests that the sites from the present study are likely subject to enough fishing pressure to experience increased abundance due to protection.

Combining species to the taxonomic level groupings *Epinephelus* spp. and *Lethrinus* spp. allowed for a greater number of fish to be compared for the given sample size. Use of defined taxonomic groups such as these are preferred when dealing with CPUE analysis of multispecies fisheries, since assessing communities as a single unit may misrepresent changes that occur due to species interactions, and the variable catchability and desirability of different species (Jennings and Polunin, 1995*c*). The primary reason for using these taxonomic groupings was: 1) their shared habitat affinity; 2) virtual lack of fisher discrimination between individual species within the *Epinephelus* spp. and *Lethrinus* spp. groups, either in fishing practices or preferences; and 3) their status as some of the most targeted species among the coastal communities (Rawlinson *et al.* 1995).

Epinephelus spp., which made up 69.7% of the study's total catch, showed significantly greater CPUE within the *tabu* areas of three sites, two of which were highly-significant. Meanwhile, *Lethrinus* spp. composed about 14.2% of the total catch, but exhibited only one site with significantly greater CPUE in its *tabu* when compared to its *tara*. The findings, in accordance with other studies (Russ, 1985; Russ, 1991; Russ and Alcala, 1989; Alcala and Russ, 1990; Polunin and Roberts, 1993; Jennings and Polunin, 1995*b*; Jennings and Lock, 1996), support the validity of these taxonomic groups as adequate indicators of fishing pressure, as they account for over 80% of the total catch.

Taxonomic groups are preferred in analysis, but the analysis of total CPUE between areas at each site was also relevant, since increasing overall catch was the driving factor for communities establishing these MPAs. Additionally, the hook and line method limited analysis to only carnivorous fish. Though preference for these different fish varies, all have some level of catchability and if caught will almost always be taken back for personal consumption at the least. Decreases in catch of preferential *Epinephelus* and *Lethrinus* spp. over time may also lead to increased fishing pressure or catchability of other species vulnerable to the same method (Willis *et al.* 2000; McClanahan *et al.* 2010) making total CPUE of species subject to this method important for analysis in its own right. Not surprisingly, analysis at this level displayed some level of significance for all sites, with those still protected having greater CPUE within their *tabu* area, and thus probable greater abundance.

Of the comparisons between areas of total CPUE, Namatakula was the only site with continued protection that showed only a marginally greater CPUE within its *tabu* when compared with the *tara*. It was also the only site with no significant difference between *tabu* and *tara* areas in CPUE analysis of *Epinephelus merra* and the *Epinephelus* spp. grouping. These findings may be attributed to under reporting of catch that took place while fishing in the *tabu* area, in which several of the fishers were withholding their catch to take home for consumption. The incident was not reported until after fishing had concluded; and though exact numbers, sizes, or species of fish taken are uncertain; several witnesses reported seeing fishers "filling their bags full (Daniel Friedman, personal communication)." While the incident was only reported to have occurred on the third of five fishing excursions into the *tabu* area, similar instances that may have previously taken place are unknown. Subsequent village meetings helped to address and resolve the issue for the final two excursions.

Findings from total catch CPUE analysis also suggest that abundance was marginally greater in the *tara* area at Komave; the only site whose *tabu* area is no longer enforced. Following the opening of the Komave's protected area in December 2008, local community members observed extensive fishing pressure being shifted toward the *tabu* area. This response was attributed to widespread awareness of its previous protection status and the perceived surplus in marine resources it had accumulated over its nearly five year closure (Josaia Totonavosa, personal communication).

Foale and Manele (2004) discussed the limitations inherent with *tabu* areas as a management tool, stating that these areas are almost always opened after less than a year, after which accumulated stocks of many species are removed, many times with alarming efficiency (Foale and Day, 1997). They argued that if fishing pressure is high, stocks of most fished species can be severely depleted; slowing rates of recruitment and leading to declines in fishery production. Their consensus was that if the typical length of *tabu* was a year or less, then most species of reef fish and invertebrates are vulnerable to overfishing. However, the findings at Komave indicate that even extended closures, in this case nearly five years, can see rapid depletion of the protected area stock within a relatively short time period. Similar findings from long term monitoring at Sumilon Island Reserve in the Philippines reported that CPUE and total yield were significantly less for the entire reef after the reserve had been pulse fished (1985-1986) than during the period when the reserve existed (Russ and Alcala, 1994). Furthermore, the present findings suggest that under considerable fishing pressure these previously protected areas can even be depleted to levels below that of the area that has had no restricted activity.

Local expertise were highly valued and utilised in this study to determine fishing locations, and times for excursions to take place. Some time after field work had commenced, it became apparent from observation that differences in fisher efficiency existed amongst the women both within and among villages. While some women excelled quite well at catching sizeable amounts of fish during an excursion, others were not so successful. This was despite using the same bait and fishing equipment. Differences may be partially attributed to the level of fishing experience held amongst women, and may be reflected in manners such as fishers' subjective decision making in dispersing about the reef for their individual fishing "spot." Other factors may include varying levels of tangible "effort" provided by individual fishers, or simply the ability of some to excel over others in this fishing practice.

Additional record keeping of individual CPUE among the women's would not only have reduced the uncertainty of this variable when considering overall abundance of a given area, but would also have allowed for multiple sets of data for each fisher, increasing the size of *n* and strengthening the statistical power of the tests conducted. At the same time, the ideal approach for standardizing fishing "efficiency" would have involved using the same group of fishers during each excursion at all villages. Time and financial constraints, not only of the researchers, but of the fishers themselves made this approach unfeasible and impractical. Additionally, given the experience of fishers in their own fishing grounds, one could argue that the use of a single group of fishers with virtually no experience other than that in their own fishing ground would severely hamper fishing efficiency.

4.3 Biomass

As with the mean CPUE abundance, the average CPUE-derived biomass (grams/person/hour) was generally greater for several species within the *tabu* areas of sites still enforcing protection, and was most pronounced at the taxonomic grouping level. All three sites with continued protection displayed significantly greater biomass values for *Epinephelus* spp. species in their *tabu* areas, ranging from 3.4-5.8 times greater than that in their respective *tara* areas. Similiarly, *Lethrinus* spp. in Namada and Namatakula had significantly-greater biomass within their *tabu* areas, approximately 5.9 and 8.8 times greater than that of their *tara* areas, respectively. Again, the only site to display opposing trends was Komave, where *tabu* and *tara* values were nearly equivalent, slightly greater, or in the case of *Epinephelus merra*, significantly greater in the *tara*.

These trends were also consistent at the level of total catch, with all sites displaying significantly-different values. Mean biomass for the total catch ranged across sites from approximately 1.2-5 times greater in the *tabu* areas compared to their respective *tara*. When protected sites combined for an “MPA-effect,” the difference in mean biomass was approximately 2.5 times greater in the *tabu* areas of the protected sites compared to the unprotected *tara* areas. These findings substantiate those from numerous other case studies that MPAs can provide benefits through increases in target species biomass (For reviews, see Halpern, 2003; Lester *et al.* 2009).

The biomass values for both *Epinephelus* spp. and the total catch add perspective to the previously discussed issue regarding abundance at the Komave site; where CPUE was nearly equivalent or slightly greater than that of other sites still enforcing protection. While catch of individual fish was fairly high at Komave, the contribution these individuals made to the total weight of the catch indicates that, while numerous, the fish being caught were smaller than those being caught at sites with protection. Indeed, when combined for an “MPA-effect,” the three enforced *tabu* areas had mean biomass approximately 3.6 times greater than the respective *tara* area and 2.2 times greater than the *tara* of Komave.

During a series of openings and closures from 1974-1994 at Sumilon Reserve in the Phillipines, Russ and Alcala (1994) found that despite large numbers of predatory fish including snappers, groupers (*Epinephelus* spp.), and emperor (*Lethrinus* spp.) being observed after 4 to 5 years of protection (1988-1991), these fish consisted mostly of low biomass individuals. This was in contrast to the greater biomass recovery observed during 1983 after a 10-year closure, and was attributed to the longevity of many of these species. Similar circumstances appear to be present in this study, with the abundance of smaller individuals of *Epinephelids*, such as *Epinephelus merra*, masking the true status of the catch. Given that these *tabu* areas were set aside to increase stocks of target food-fish species, not only the number of fish, but the weight of fish obtained per unit effort is an important consideration for assessing whether these areas are fulfilling their management goals. Thus, biomass may be a better indicator of the benefits afforded by protection than simply comparisons of CPUE alone. Further studies are needed to assess other biological, physical, and anthropogenic influences between sites.

4.4 Mean Size

Size-selective fishing has the ability to affect fish assemblage structure and function, with the potential to influence the productivity and resilience of some stocks (Baskett *et al.* 2005; Watson *et al.* 2009). The increased size of target species in MPAs is extremely important as it correlates to greater biomass and fecundity, with length increases of 10% capable of producing 25 to 30% increases in biomass (Tetreault and

Ambrose, 2007). Mean size of target species and species groupings showed a general trend toward significantly greater sizes within the *tabu* areas of the sites where protection is still enforced, with most exhibiting highly-significant differences. These differences demonstrate that protection, even small MPAs such as those examined in this study, can lead to an increase in size of exploited species. This finding is consistent with other studies of MPA implementation on size of target fish species (Watson *et al.* 2009). A synthesis of 149 peer-reviewed scientific publications published between 1977 and 2006 demonstrated an overall positive, yet moderate, response to reserve protection on organism size; with average increases of 28% and median increases of 17% (Lester *et al.* 2009). In the present study, all sites with continued protection exhibited significantly-higher mean sizes in their *tabu*, with differences from 8.5-41.9% for individual species and 10.1-39.8% for taxonomic groups. These results are noteworthy, not only because of the limited sizes of these protected areas and duration of these closures, but because this parameter has a much lower scope for change relative to other parameters, such as density.

Mean sizes of *Epinephelus hexagonatus* and the *Epinephelus* spp. grouping were significantly greater in the *tabu* at Komave than in the *tara* area. Even so, these values were much lower than those at the other protected sites, with differences between *tabu* and *tara* areas of 1.8% for *Epinephelus hexagonatus* and 3.6% for the combined *Epinephelus* spp. Additionally, mean sizes for *Epinephelus hexagonatus* and combined *Epinephelus* spp.in the *tabu* of several sites, including the combined “MPA-effect,” were greater than in Komave. The same is true of the greater, though not significant, mean size of *Epinephelus merra* in the Komave *tabu*. It is important to point out that despite having

CPUE in the Komave *tara* nearly equivalent to that in other protected areas, mean fish size of *Epinephelus hexagonatus*, *Epinephelus merra*, and the entire *Epinephelus* spp. grouping are still less that found at other sites; even less than some *tara* areas including the combined “MPA-effect” grouping. Again, as with CPUE, underreporting of catch from excursions into Namatakula’s *tabu* may have distorted some of the results. Even so, results from *tabu* sites with continued enforcement suggest that these areas are enhancing survival of target species, which is expected to benefit adjacent fisheries through spillover and reproductive export of larvae. This will depend upon the species' biological characteristics (ie. mobility, dispersal capability of the pelagic larvae) and physical parameters of the reserve (ie. patterns of ocean currents, habitat connectivity) (Gell and Roberts, 2003).

4.5 Sexual Maturity of the Catch

Increases in target species abundance and size in protected areas are expected to translate into increased reproductive potential, thus enhancing recruitment (Gell and Roberts, 2003). A potentially more accurate representation of the contribution of protected areas to recruitment would be based on the proportion of the population capable of reproducing (Roberts and Hawkins, 2000). Virtually all villages still enforcing their *tabu* exhibited percentages of target fish species greater than or equal to SSM that were at least a 10% or greater compared to their adjacent *tara* area. Namada posted exceptional differences in percentages of fish greater than or equal to SSM, especially for *Lethrinus* spp. 100% of the catch was of sexual immature size within the *tara* area, while 85.2% in the *tabu* were of maturity size. At Komave, catch of all species of sufficient sample size exhibited majorities, in most cases vast, which were of immature length. Even for *Epinephelus merra*, a species highly resilient to fishing pressure, 51% or more of the catch was below maturity size in both areas. These values are comparable to those found in the *tara* areas at other sites (±16%). Meanwhile, the percentage of *Epinephelus merra* within the *tabu* of the three sites that remain protected were around 25% greater than within their corresponding *tara* areas, and were roughly 18-42% greater than in either area at Komave. These findings contrast to those of Unsworth *et al.* (2007), who examined maturity of grouper species in small-scale no-take MPA over 5 years following reserve establishment. When compared to adjacent areas subject to fishing, they found more mature populations of most groupers, except for *Epinephelus merra*. Despite being the most abundant fish recorded, 90% of the population of *Epinephelus merra* were below 20 cm in length, with 70% below 15 cm (half maximum size) (Froese and Pauly, 2010) at all sites, indicating immature populations. These findings are in agreement with those obtained in the current study from Komave, however, the results of this study suggest that small MPAs can provide benefit for this species in terms of more mature populations. Perhaps the most profound findings are the percentage of total species for which SSM is known. Only about 26.3-29.6% of the stock was of mature length in either area at Komave; slightly lower than at the *tara* area at most other sites, which when combined for an “MPA-effect” comprised about 33.5% of the catch. In comparison, the fish of mature size in the combined “MPA-effect” *tabu* of the three protected sites was 53.4% of the stock.

The use of SSM values are important management parameters for monitoring whether enough juveniles in an exploited stock mature and spawn (Jennings *et al.* 1998).

However, all approximations may underestimate the percentage of fish that are actually still immature even at these sizes. Indeed, the use of the most prevalent metric for species in this study, L_{50}, should equate to half of the population at a given size is immature. A more appropriate gauge of reproductive potential may be possible by taking a larger size along the size-at-maturity curve, such as the L_{90} (Purcell *et al.* 2009). In such a case, it is likely that the current SSM percentages for some species across area sites might be lower than those currently reported. All fish measurements in this study were rounded to the nearest centimetre, which may have also affected these figures.

Those sites maintaining their MPAs certainly exhibit larger and more fecund populations of several target species than the unprotected Komave site. It is assumed that this will strengthen their capability to produce more larvae with greater survival potential, which will in turn increase their potential to replenish stocks to areas open to fishing (Francini-Filho and Moura, 2008). Whether this occurs may determined based on MPA sizes, their spatial distribution, and interactions of the two with oceanographic conditions, larval dispersal and larval life history characteristics (e.g. time in the plankton) (Watson *et al.* 2009).

4.6 Study Considerations

Interestingly, sites still under protection seemed to respond regardless of their fairly small size. Despite having the smallest *tabu* of the four sites, Namada exhibited very high mean CPUE, size, biomass, and percentage of species greater than or equal to SSM; often with values 20% or greater than its *tara* area. This is in agreement with Halpern’s (2003) findings that the relative magnitude of the effect of a reserve on a biological measure appear to be independent of reserve size, as well as similar findings from empirical studies in extremely small reserves in St. Lucia (0.026 km^2) (Roberts and Hawkins, 1997) and Chile (Las Cruces: 0.044 km^2) (Castilla *et al.* 2007) that exhibited significantly larger values in the biomass and size of the organisms within the protected area compared to outside areas.

The exceptional performance of the *tabu* at Namada compared to those at other sites may have been attributed to the culmination of several factors. Not only has the closure at Namada been in place the longest, but it is also the only *tabu* of the four located directly in front of the village, hence allowing it to receive the best enforcement. Additionally, anecedotal evidence suggests that Namada is subject to lower fishing pressure compared to the *tabu* located around more populated villages such as Namatakula, or Votua with its nearby settlements.

The results of this study, particularly in regards to Komave, highlight the on-going debate regarding the efficacy of no take reserves versus those subject to periodic openings, rotations and/or partial restrictions. Modeling suggests that while permanent reserves will offer the greatest benefits; rotation or periodic harvest of marine reserves may still produce positive benefits for biomass and abundance of target species, depending on their life history characteristics (Gerber *et al.* 2003; Lester and Halpern, 2008). This has been supported by empirical evidence from other community-based MPAs in the Pacific, such as in Papua New Guinea (McClanahan *et al.* 2006) and Vanuatu (Bartlett *et al.* 2009*b*), and most often in areas with low population, low market connectivity, and high social capital. However, conflicting studies, including the present, have found that these types of reserve strategies often experience an immediate postharvest depletion of resources (Russ and Alcala, 1998*a*; Ferraris *et al.* 2005). Indeed, rotational strategies used in a small Hawaiian marine reserve proved ineffective because of harvested during rotations fish stocks were harvested to a degree that prevented long-term improvements (Williams *et al.* 2006).

The combined findings of several studies from Kenya (McClanahan, 2000; McClanahan and Graham, 2005) and the Philippines (Russ and Alcala, 2004) have determined that the no-take protected areas need to be maintained for at least two decades before coral reef fishes approach their full size and biomass. Thus, the capacity of different management strategies to enhance fish stocks would likely depend upon the controls over fishing intensity during openings (Williams *et al.* 2006), as well as the life histories of target species (McClanahan and Graham, 2005). For many Fijian villages located along the coast of Viti Levu, like the four addressed in this study, pressure from fishing may be too great to permit periodic openings of any given extended period, especially for more vulnerable target species such as the large, long-lived *Lethrinus* spp.

The incident discussed previously regarding the withholding of catch at Namatakula accentuates the importance of enforcement and compliance to MPA management. Any benefits gained from reserve areas depend on the degree of respect it is afforded by fishermen (Kritzer, 2004; Byers and Noonburg, 2007), and the occurrence of opportunistic behaviour is not implausible in common-property resource scenarios. Social and economic factors may influence individuals' decisions toward acting either out of self interest or for the common interest of all resource users (Ostrom *et al.* 1999; Pretty, 2003; McClanahan *et al.* 2006). Episodes like that at Namatakula stress the need to foster communication, participation, and education in the community-based management process. They also merit further investigation into current trends in Fijian community social organisation, how this affects perceptions of resource use (Johannes *et al.* 1993), and the implications for traditional management regimes.

Socio-economic assessments conducted by McClanahan *et al.* (2006) found that effective conservation was positively related to compliance, visibility of the reserve, and the length of time the management has been in place but negatively related to market integration, wealth and village population size. The demands introduced by an increasingly cash-based economy; along with changing values and the erosion of the traditional Fijian social structure may present challenges to the traditional management approach. Important areas of societal framework that should be considered include concepts in the term *vanua*, the chiefly authority system, social values, ideal Fijian behaviour including the act of being considerate (*veinanumi*), the preference of collectivism in relation to individualism, and compliance with authority (Fong, 2006).

Less obvious occurrences of noncompliance within the established *tabu* areas may have also contributed indirectly to the results obtained during the study. Compliance with community-based protected areas can be poor, especially in cases where surveillance is difficult (Crawford *et al.* 2004) and there is a lack of external assistance (Pollnac *et al.* 2001). Interestingly, the *tabu* at Namada consistently had some of the greatest values for mean abundance, size, and percentage of fish at SSM of all sites, and was also the only currently protected site with its *tabu* area located almost directly in front of the village. Still, poaching is common occurrence within all the *tabu* areas, with some incidents having taken place during the period of the study (personal observation).

The influence of poaching has been advocated as a factor to consider in reserve design (Kritzer, 2004; Byers and Noonburg, 2007), as it can have varying effects on fish abundance via the fraction of fish harvested inside the reserve, thus undermining the reliability of predicted consequences of a protection on fish population. By increasing fishing pressure, fishing mortality is also increased. Fishers also typically target larger individuals and species at higher trophic levels (Pauly *et al.* 1998; Wilson *et al.* 2008). As a result, one would also expect decreases in the expected life span and consequently the potential size of individual fish. Because fecundity typically increases with size, this chain of events may have further consequences beyond just the direct effect on abundance (Byers and Noonburg, 2007). That said, the frequency and intensity of poaching events at these *tabu* sites, and the magnitude of their influence not just on abundance, but on organism size, biomass, and percentage of fish at maturity size are unknown.

4.7 Study Limitations

The overall findings from mean CPUE, size, biomass and percentage of fish at SSM suggest that all four can benefit from protection. However, ecological systems are subject to inherent spatial and temporal variability due to changing rates of biotic (e.g. predation, competition, recruitment) and abiotic (e.g. habitat size/quality/disturbances, fishing intensity) factors (Guidetti, 2002), which may affect reef fish abundance, size, diversity, and distribution (Beukers and Jones, 1997; Guidetti, 2002; Kulbicki *et al.* 2007; McClanahan and Arthur, 2001; Steele, 1999; Wilson *et al.* 2008). These processes no doubt had varying degrees of influence between areas and among sites in this study. The relevant influence of protection in relation to these processes will most likely depend on the extent of and species' susceptibility to such processes (Wilson *et al.* 2008; Wilson *et al.* 2010). The nature of the study also limits the assumptions that can be drawn, as it is but a "snapshot" in time of the fishery's status. In the absence of baseline data, there is the possibility that differences between fished and unfished areas may have existed prior to reserve establishment. Temporal replication before and after MPA establishment would have been ideal to assess changes in average conditions, but were unrealistic given financial constraints and the nature of the reserves' establishment. The ability to determine if and to what extent the recruitment effect is occurring is also hindered by a lack of before and after data.

Furthermore, the nature of the sampling procedure was subject to variation due to different levels of catchabilty among species; such as large carnivore target species exhibiting fishing-related behavioural plasticity between sites (Cole, 1994; Jennings and Polunin, 1995*a*), time of day, and unpredictable weather conditions that occurred during fishing excursions. Even sample recording itself may have experienced minor variations, such as accurate species identification. Many species, such as individual species of *Epinephelidae* and *Lethrinidae*, may be hard to distinguish between other members of their taxonomic group, especially when caught as juveniles, as was the case with many of the fish in this study.

The results of overall biomass caught and percent of sexual mature fish caught may also have been subject to inaccuracies, since length-weight and length-length metrics were not available for some species. However, these variations are expected to be minimal given that those for whom metrics were available comprised the vast majority of the catch, and that most species caught were those that exhibit fairly negligible differences between fork and total length, or had conversion rates between the two length measures.

In regards to overall benefits to the community assemblage, the nature of assessment was subject those species susceptible to hook and line, and offers limited insight into the effects protection may have on other species of socio-economic and/or ecological importance (McClanahan and Mangi, 2004). Nevertheless, the findings presented here and the importance of the species assessed to local Fijian communities emphasises the need for no-take protected areas to promote sustainability of key fishery stocks.

Chapter 5: Conclusions and Recommendations

The findings of this study support the utilisation of permanent no-take *tabu* areas as a MPA management strategy for LMMAs; with the potential to benefit stocks of target species subject to elevated fishing pressure. These benefits may be manifest as increases in biological parameters including CPUE, biomass, size, and percentage of fish reaching sexually maturity. Evidence suggests that fishing pressure may be an important factor in management of these fish communities, and that even relatively-small no-take areas can still garner some benefits in terms of preserving a more abundant and reproductive stock. The findings from Komave suggest that even after protection, many of the benefits gained by MPA establishment can be quickly depleted, and that even temporary closure spanning a few years may be insufficient for protecting stocks of more vulnerable species.

Given the limited scope of this study, there is a need to design and carry out further studies that can unequivocally demonstrate that these MPA management strategies have produced tangible benefits toward the sustainability and enhancement of fishery yield in the surrounding *iqoliqoli*. Studies that address missing biological information for target species such as mobility, life-history, rates and patterns of settlement and recruitment, connectivity among neighboring populations, and the status of these populations as sources or sinks will be critical to assessing the suitable size and placement of reserves, as well as the use of particular species as short or long-term indicators. These studies could be complimented by those pertaining to physical parameters at the various sites such as the habitat, bathymetry, hydrodynamics and the role these various factors play in the dynamics of the fishery. Optimally, such studies should be carried out over an extended temporal scale to help account for natural variations that may affect the status of the stock. Additionally, follow up socio-economic monitoring should be conducted to assess the changing value, perceptions, and utilisation of fishery resources, and how these changes will influence sustainability of the resource. Successful management of nearshore coral reef fisheries will likely require a holistic, integrated-management approach, spanning the aforementioned factors to the incorporation of terrestrial management regimes. That said, the use of no-take *tabu* areas are a valuable first step in the right direction toward sustainable community-based fisheries management.

Appendix A

Reef Fish Database

Table A: Summary table of biological measures of maturity, length, and weight data for all species caught during the study. *Lm* indicates length at maturity; *Lm* metric indicates percentage at sexual maturity; FL indicates standard length, TL total length, SL standard length; *a* indicates the y-intercept, *b* the slope of the fitted linear relationship; n/a not available or unknown. [1] Froese and Pauly, 2010; [2] May and Robinson, 2004; [3] Ebisawa, 2006; [4] Sadovy, 1999; [5] Shakeel and Ahmed, 1996

				Length-Length			**Length-Weight**		
Species	***Lm***	***Lm* metric**	***L* type**	**a**	**B**	**Converted *Lm***	**a**	**B**	***L* Type**
Abudefduf sexfasciatus	--	--	--	--	--	--	0.0213[1]	3.152[1]	FL
Balistapus undulatus	13.8[2]	L50	n/a	0.000[1]	1.000[1]	--	0.0058[1]	3.554[1]	TL
Caranx melampygus	35[1]	L50	SL	0.150[1]	1.076[1]	37.810	0.0235[1]	2.920[1]	FL
Caranx papuensis	--	--	--	--	--	--	0.0249[1]	2.910[1]	FL
Cephalopholis argus	--	--	--	--	--	--	0.0093[1]	3.181[1]	FL
Cephalopholis urodeta	--	--	--	--	--	--	--	--	--
Cephalopholis urodeta	--	--	--	--	--	--	--	--	--
Cheilinus chlorourus	19.8[2]	L50	n/a	--	--	--	--	--	--
Cheilio inermis	21.7[2]	L50	n/a	--	--	--	0.0035[1]	3.082[1]	FL
Cheilinus trilobatus	19.8[2]	L50	n/a	--	--	--	--	--	--
Coris gaimard	--	--	--	--	--	--	--	--	--
Dascyllus trimaculatus	--	--	--	--	--	--	0.0313[1]	3.043[1]	FL
Epinephelus	30[4]	n/a		0.000[1]	1.000[1]		0.0183[1]	2.891[1]	TL

bleekeri									
Epinephelus hexagonatus	19[1]	L50	TL	0.000[1]	1.000[1]	19.000	0.0177[1]	2.930[1]	TL
Epinephelus lanceolatus	105[2]	L50	n/a	0.000[1]	1.000[1]		0.0173[1]	3.000[1]	TL
Epinephelus merra	14.2[2]	L50	FL	--	--	--	0.0158[1]	2.966[1]	FL
Epinephelus spilotoceps	24	n/a	--	--	--	--	0.0041[1]	3.346[1]	FL
Epinephelus tauvina	61.1[1]	L50	TL	0.000[1]	1.015[1]	60.197	0.0156[1]	2.930[1]	TL
Halichoeres hortulanus	12.8[1]	L50	TL	0.000[1]	1.000[1]	12.800	0.0119[1]	3.064[1]	TL
Halichoeres trimaculatus	12.6[2]	L50	n/a	--	--	--	0.0275[1]	2.736[1]	
Lethrinus ambioensis	29.2[2]	L50	n/a	--	--	--	0.0201[1]	2.964[1]	n/a
Lethrinus harak	21.1[3]	L90	FL	--	--	--	0.017[1]	3.043[1]	FL
Lethrinus lentjan	20[1]	L50	n/a	--	--	--	0.0197[1]	2.986[1]	FL
Lethrinus obsoletus	25.7[3]	L60	FL	--	--	--	0.0173[1]	3.026[1]	FL
Lethrinus ornatus	20[3]	L90	FL	--	--	--	0.0201[1]	2.964[1]	n/a
Lethrinus xanthochilus	29.9[2]	L50	n/a	--	--	--	0.0201[1]	2.964[1]	FL
Lutjanus argentimaculatus	51.9[1]	L50	TL	0.000[1]	1.018[1]	50.982	0.0336[1]	2.792[1]	TL
Lutjanus bohar	26.8[2]	L50	n/a	--	--	--	0.017[1]	3.035[1]	FL
Lutjanus ehrenbergii	15.8[2]	L50	n/a	0.000[1]	1.032[1]	--	0.0026[1]	3.335[1]	TL
Lutjanus fulvus	17.8[2]	L50	n/a	--	--	--	0.0243[1]	2.928[1]	FL
Lutjanus kasmira	13.9[2]	L50	n/a	--	--	--	0.0117[1]	3.136[1]	FL
Lutjanus monostigma	24.1[2]	L50	n/a	--	--	--	0.0222[1]	2.913[1]	FL
Lutjanus quinquelineatus	17[2]	L50	n/a	--	--	--	0.0296[1]	2.851[1]	FL
Lutjanus semicinctus	--	--	--	--	--	--	0.0224[1]	3.002[1]	FL
Mulloidichthys pflugeri	--	--	--	--	--	--	--	--	--
Monotaxis grandoculis	25.5[2]	L50	n/a	--	--	--	0.023[1]	3.022[1]	FL
Novachulichthys taeniourus	13.8[2]	L50	n/a	--	--	--	--	--	--

Parapercis hexophthalma	--	--	--	--	--	--	0.0068[1]	3.157[1]	FL
Parupeneus indicus	--	--	--	--	--	--	0.0142[1]	3.114[1]	FL
Parapercis millipunctata	--	--	--	--	--	--	--	--	--
Parupeneus multifasciatus	13.8[2]	L50	n/a	--	--	--	0.0114[1]	3.211[1]	FL
Parupeneus spilurus	--	--	--	--	--	--	0.0192[1]	3.022[1]	FL
Rhinecanthus aculeatus	14[1]	L50	TL	0.000[1]	1.000[1]	14.000	0.0522[1]	2.641[1]	FL
Scarus frenatus	--	--	--	--	--	--	--	--	--
Scarus spinus	--	--	--	0.000[1]	0.920[1]	--	0.0279[1]	3.06[1]	SL
Siderea picta	--	--	--	--	--	--	--	--	--
Sphyraena jello	55.4[2]	L50	n/a	--	--	--	0.014[1]	2.81[1]	FL
Synodus dermatogenys	--	--	--	--	--	--	0.0046[1]	3.346[1]	FL
Terapon jarbua	13[1]	L50	n/a	--	--	--	0.0154[1]	3.082[1]	FL
Thalassoma hardwicke	8.5[1]	L50	n/a	--	--	--	0.0178[1]	2.978[1]	FL
Tylosurus crocodilus	--	--	--	--	--	--	0.0008[1]	3.205[1]	FL
Zenarchopterus dispar	--	--	--	--	--	--	--	--	--
unidentified *Labridae* sp.	--	--	--	--	--	--	--	--	--
unidentified *Labroides* sp.	--	--	--	--	--	--	--	--	--
unindentified *Congridae* spp.	--	--	--	--	--	--	--	--	--

Appendix B

List of Relevant Fijian terms

Iqoliqoli- customary Fijian fishing ground that extends from the high tide water mark along the shoreline to the most outer reef crest (can be either fringing or barrier reef)

Tabu- a Fijian term used for referring to the type of Marine Protected Areas assessed in this study. Intended to link traditional prohibition on marine resource harvest to modern-day MPA strategies; literally means forbidden, prohibited.

Tara- the area within the traditional fishing grounds where fishing is allowed; literally means lawful, not *tabu*

Kabatia- local term generally used to refer to species of emperor fish (*Lethrinus* spp.)

Kahala- local term generally used to refer to several species of grouper (*Epinephelus* spp.)

Yavusa- the largest kinship and social division of Fijian society

Mataqali- the primary social division in Fijian society, larger than an *itokatoka* (family), and smaller than a *Yavusa* (tribe)

Tikina- a district, otherwise known as *Tikina Couvata*. This may further be divided into subdistricts, or *Tikina Vou*

Vanua- similar to a *Tikina* in that it can contain in it a number of *Yavusa* and villages, it will have traditional head

Yasana- a province

Tanoa- a large wooden bowl used for preparing kava

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