Scaling and Sustaining Locally Managed Marine Areas

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ABSTRACT

In many parts of the tropics, coastal communities are increasingly assuming responsibility for nearshore resources under arrangements known as "locally managed marine areas" or LMMAs. Broadly similar to marine protected areas, LMMAs are managed for sustainable, long-term use rather than biodiversity conservation itself, and typically employ a range of management techniques, including periodic closures, gear restrictions, secure access rights, species-specific reserves and permanently closed, fully protected areas (no-take zones) to achieve this aim. Evidence suggests that, when effective, LMMAs can encourage responsible fishing, strengthen compliance and may help to safeguard food security and increase resource abundance. However, the recently established and informal nature of many initiatives means that several questions remain unanswered. In particular, little is known about the effectiveness of LMMAs in achieving long-term ecological goals, with initiatives often lacking the necessary financial resources to sustain both effective management and the collection and evaluation of robust socio-ecological evidence on which such management often depends. A second unresolved question concerns the status and scale of LMMAs outside the Western and Central Pacific, the region where research interest has concentrated to date. This PhD combines social and ecological research to investigate how to address these key research gaps and improve the effectiveness of locally managed marine areas.

The work falls into four interdisciplinary data chapters, with a core focus of examining the role of research design in improving assessments of LMMAs and other spatially explicit tools like marine protected areas, as well as the potential role of tourism as a source of funding and support for LMMAs. The opening two data chapters take an interdisciplinary approach to the assessment of a long-established network of LMMAs in Rarotonga, Cook Islands, beginning with an examination of tourist satisfaction with the network, using respondent completion questionnaires. The results found widespread disappointment, with around half of respondents not satisfied with coral cover or diversity and around 40% dissatisfied with fish abundance, size or diversity.

The second data chapter swaps socio-economics for ecology, evaluating the effectiveness of Rarotonga's LMMAs using three contrasting research approaches. A network-level analysis, which assessed *overall* responses to protection, suggested no LMMA effects, with abundance and biomass of species targeted by fishers not

significantly higher within LMMAs compared to control sites. In contrast, a site-level analysis, which explored the differences between each of the *individual* LMMAs and their paired controls, revealed significant differences in targeted abundance (2.5-3 times higher), biomass (4.3-5.4 times higher) and community structure between LMMAs and controls at the two sites where hotels were acting as co-management partners. A third analysis, which used an asymmetric approach to examine the performance of each LMMA against multiple control sites, accorded with the second, finding that both targeted biomass and abundance were significantly higher at the hotel-supported sites than at multiple controls.

The third data chapter continues to examine the role of research design in improving assessments of area-based management initiatives. A novel double-blind randomised controlled trial using underwater video fish surveys demonstrated that observers who were told that the transects they were assessing came from a no-take marine reserve significantly and erroneously overestimated the effectiveness of the reserve vs. those who were not told, inflating their fish counts by approximately 28% (95% CI 18.5% to 40.5%, $p>0.0001$).

The final chapter explores new horizons for LMMAs, presenting the first assessment of the status and extent of LMMAs in the Western Indian Ocean (WIO) and supporting legal frameworks. Using the most comprehensive dataset of the region's LMMAs and marine protected areas yet developed, this analysis revealed that more than 11,000km^2 of marine resource in the WIO was managed in LMMAs. The analysis also found that many initiatives were hampered by underdeveloped legal and enforcement mechanisms and argued for a regional network of LMMA practitioners to share best practice on financing and evaluation and to encourage the development of further LMMAs. Taken together, the different chapters expand our knowledge of LMMAs in the Western and Central Pacific and WIO and provide important baselines against which to evaluate future management strategies in the Cook Islands and WIO. They highlight the importance of capturing visitor perceptions of marine resource management and protection efforts in tourism-dependent island states, and provide some evidence for the role coastal hotels can play in supporting and sustaining LMMAs in tropical developing countries. Finally, they demonstrate that both a common analytical flaw and a previously unresearched bias may confound results in evaluations of the ecological effectiveness of protected areas, underscoring the need to incorporate blind assessment and to more carefully consider potentially confounding variables in future research designs.

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Declaration

I declare that this thesis is my own work and has not been submitted for any other degree or award. It was supervised by Dr Julie Hawkins and latterly by Professor Dave Raffaelli and funded through a joint studentship from the Natural Environment Research Council and the Economic and Social Research Council. The four core data chapters (2-5) are presented as standalone manuscripts. The contributions of co-authors to each of these chapters are detailed below.

Chapter 2: SR formulated the idea, conducted the research, analysed the data and wrote the manuscript. LGA, GC and EW assisted in the gathering of primary field data under my direction and supervision. All data were collected in accordance with protocols I developed, and were collated and reviewed by me. LGA and JPH contributed to writing the manuscript.

Chapter 3: SR formulated the idea, conducted the research, analysed the data and wrote the manuscript. LGA (Lucy Anderson), GC (Georgia Coward) and EW (Emily Williams) assisted in the gathering of primary field data under my direction and supervision. All data were collected in accordance with protocols I developed, and were collated and reviewed by me. LGA and JPH contributed to writing the manuscript.

Chapter 4: SR designed the study and hypothesis, conducted the experiments, analysed the data and drafted the manuscript. LGA helped to conduct the experiment and acted as confederate to ensure blinding was maintained. DAB (Dominic Andradi-Brown) filmed the video transects. DAE (Dan Exton) contributed to writing the manuscript.

Chapter 5 is based on a jointly authored publication

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SR formulated the idea, conducted the research, analysed the data and wrote the manuscript. SP and MS assisted with data collection and contributed to writing the manuscript. JPH contributed to writing the manuscript.

Finally, parts of **Chapter 1 and 6** have been reworked into a jointly authored publication

Rocliffe, S., Anderson, L.G., Glass, L., and Harris, A., 2016 (in review). Incentives for community-based tropical marine conservation: panacea or pipe dream? *Aquatic Conservation*

Stephen Rocliffe

1 General Introduction

*"the concentric cones, the hammerheads, the donaces, real bounding shells, the broques, the red helmets, the angel-winged strombes, the aphysies, and many other products of the **inexhaustible** ocean"*
(Verne, 1869).

*"I believe that the cod fishery, the herring fishery, the pilchard fishery, the mackerel fishery, and probably all the great sea fisheries, are **inexhaustible**; that is to say, that nothing we do seriously affects the number of the fish. And any attempt to regulate these fisheries seems consequently, from the nature of the case, to be useless"*
(Huxley, 1883)

1.1 Benefits from the oceans – ecosystem goods and services

Our oceans are vital to us. They provide an array of good and services that underpin human livelihoods, health, wellbeing and survival (Costanza et al., 2014, 1997; Millennium Ecosystem Assessment, 2005; TEEB, 2010). They support 80% of the world's biodiversity, supply almost half the oxygen we breathe, and absorb 30% of anthropogenic carbon dioxide emissions (DEFRA, 2010; IPCC, 2014). They also deliver the primary source of protein for more than a billion people worldwide and generate income for hundreds of millions of coastal dwellers: almost 120 million workers are directly dependent on commercial capture fisheries and associated industries for their livelihoods (The World Bank, 2012; UN FAO, 2014).

The direct and indirect contributions of marine and coastal ecosystems to human wellbeing are known as marine ecosystem services (ES, Duffy, 2006; TEEB, 2010). They are usually categorised into one of four main service "bundles": provisioning, regulating, cultural and supporting (also known as habitat) (Table 1.1)

TABLE 1.1 ECOSYSTEM SERVICES PROVIDED BY OR DERIVED FROM MARINE AND COASTAL ECOSYSTEMS*

Services	Examples
Provisioning	
Food	Production of fish, algae, and invertebrates
Raw materials	Production of timber, fuelwood, peat and aggregates
Medicinal resources	Extraction of materials from biota; genes for resistance to plant pathogens, ornamental species
Regulating	
Climate regulation	Regulation of greenhouse gases, temperature, precipitation, and other climatic processes; chemical composition of the atmosphere
Biological regulation	Resistance of species invasions; regulating interactions between different trophic levels; preserving functional diversity and interactions
Pollution control and detoxification	Retention, recovery, and removal of excess nutrients and other pollutants
Erosion regulation	Retention of soils and sediments
Extreme event regulation	Flood control, storm protection
Cultural	
Recreational	Opportunities for tourism and recreational activities
Aesthetic appreciation	Appreciation of natural features; inspiration for culture, art and design
Spiritual and inspirational	Personal feelings of wellbeing
Educational	Opportunities for formal and informal education and training
Supporting/habitat	
Biodiversity	Habitats for resident or transient species; maintenance of genetic diversity
Soil formation	Sediment retention and accumulation of organic matter
Nutrient cycling	Storage, recycling, processing, and acquisition of nutrients

* Includes open ocean and coastal seas; estuaries are excluded. Sources: Böhnke-Henrichs et al., 2013; Duffy, 2006; Millennium Ecosystem Assessment, 2005; TEEB, 2010

In a seminal paper almost two decades ago, Costanza et al. (1997) estimated the value of global ecosystem services at US$ 33 trillion per year (in 1995 $US), attributing

US$21 trillion to marine ecosystems and US$12 trillion to their terrestrial counterparts. This figure, which was considerably larger than global gross domestic product at the time, helped to build awareness amongst decision makers of the need to properly account for the value of natural assets in bottom-line calculations (Braat and de Groot, 2012; Costanza et al., 2014). Almost a decade later in 2006, ecosystem services became more firmly entrenched in the lexicon of policy makers with the publication of the Millennium Ecosystem Assessment, a four year study undertaken by the United Nations and involving more than 1,300 scientists (Costanza et al., 2014; Millennium Ecosystem Assessment, 2005). This was followed a year later by a second international initiative: The Economics of Ecosystems and Biodiversity (TEEB, 2010). TEEB's outputs were more broadly disseminated among the global news media than earlier assessments, leading to a surge in interest in ecosystem services from the general public and from business, and attracting both supporters seeing ES as a means to reframe the relationship between people and nature (Costanza et al., 2014; Helm, 2015), as well as critics concerned about the "commodification" of nature that such a reframing might imply (Costanza, 2006; McCauley, 2006; Monbiot, 2014). In 2014, Constanza et al. published revised figures estimating the value of marine ecosystem services at US$ 50 trillion per year (in 2007 $US), an increase of 72% over 1997 figures after conversion to 2007 $US (Costanza et al., 2014).

1.2 Threats to marine and coastal ecosystems and resources

Despite this enormous value, marine and coastal ecosystems worldwide are threatened by a range of anthropogenic pressures, including pollution, habitat loss, climate change, rising coastal populations and overfishing (Halpern et al., 2008; Jackson, 2008; Jackson et al., 2001).

Jules Verne may have given the world prophetic descriptions of submarines and SCUBA diving in his classic work *20,000 leagues under the sea*, but from the quote that prefaces this chapter, it appears that he couldn't have got it more wrong when it came to human exploitation of the sea (Verne, 1869). Human harvesting of marine animals began more than 40,000 thousand years ago and was already impacting fauna at the local level by the time *20,000 leagues* was published (Lotze et al., 2006; O'Connor et al., 2011).

A century and a half on and the oceans are in trouble. Although overall extinction rates of marine fauna remain relatively low when compared to land, global analyses suggest that more than 90% of previously important marine species have been depleted (Lotze et al., 2006), that 29% of fish stocks are being fished at biologically unsustainable levels (UN FAO, 2014), that large predatory fish biomass is only about 10% of pre-industrial levels (Myers and Worm, 2003) and that 83% of fished reefs are missing half their expected biomass (MacNeil et al., 2015).

Anthropogenic pressures have also caused the very different environments of mangroves and coral reefs to both recede by 1-2% per year (Bruno and Selig, 2007; Duke et al., 2007) and have substantially increased biological invasions, with ocean going vessels inadvertently transporting over 3,000 potential invaders on any given day (Cariton and Geller, 1993; Torchin et al., 2002). Overall, no part of the ocean is entirely unaffected by humans, and 41% is strongly affected by multiple threats (Halpern et al., 2008).

Global climate change is also threatening the oceans. Rising carbon emissions from human activities have caused the average sea surface temperature to increase by 0.31°C to 0.65°C over the past half century (Hoegh-Guldberg et al., 2014). This warming of the ocean has had several impacts, including: i) reduced mixing of the water column, which has lowered oxygen concentration in deep areas of the ocean and negatively affected surface nutrient concentrations; ii) sea level rise, which is threatening low lying nations like the Maldives and Kiribati; iii) increasing frequency and severity of storm systems, which is intensifying stress on coastal ecosystems; iv) coral bleaching, the ejection of symbiotic dinoflagellate algae zooxanthellae from coral tissues, which is increasing die off; and v) ocean acidification, which has raised the average pH of the ocean by 0.1 units since pre-industrial times, lowering coral skeleton density and reducing resistance to breakage and bioerosion, as well as reducing the growth rates of shelled molluscs like oysters and clams, undermining the industries that depend on upon them (Buddemeier et al., 2004; Ekstrom et al., 2015; Hoegh-Guldberg et al., 2015, 2007; Hughes et al., 2003; Maynard et al., 2009; Veron et al., 2009).

Cumulatively, these impacts are causing the ocean to change faster than at any other point in tens of millions of years, putting marine ecosystems and organisms under a strain that is unprecedented in modern history (Hönisch et al., 2012; Pörtner et al., 2014). The consequences for human well-being have been far reaching and numerous, leading to greater social conflict, undermining storm protection and decreasing flows of other ecosystem services (Brashares et al., 2014; McCauley et al., 2015; Worm et al., 2006). With over a billion people worldwide, many of them small-scale fishers, depending on seafood as their primary protein source, declines are also a major source of concern for future food security (Brashares et al., 2014; Gutierrez et al., 2011).

1.3 The failure of traditional fisheries management

Effective management of the broad, multifaceted and increasingly severe stressors facing marine and coastal ecosystems worldwide has historically been compromised by the tendency of the scientific, policy-making and resource management communities to treat individual threats in isolation (Fraschetti et al., 2011). Traditional fisheries management, for example, typically seeks to maintain the sustainability and productivity of fisheries through regulation and assessment of stocks on a species-by-species basis. Several management measures may be used to achieve this aim, including spatial and temporal restrictions on catch and effort, as well as gear controls and size limits (Holland, 2003).

When combined with rights-based management and correctly aligned incentives, this approach can yield successful and sustainable fisheries, for example in Iceland, New Zealand and Madagascar (Costello et al., 2008; Grafton et al., 2006; Hilborn, 2007a; Oliver et al., 2015). In general, however, single-species fisheries management by definition ignores the impacts of the fishery on the broader ecosystem – for instance through disregarding the effects of destructive gears such as dredges, or by failing the account for the benefits of non-extractive uses including tourism – and has been unable to prevent huge levels of overfishing globally (Costello et al., 2008; Fraschetti et al., 2011; Jackson et al., 2001; Worm et al., 2006).

In response to these failures, there has been growing recognition of the importance of an integrated ecosystem-based management (EBM) approach in the marine realm (Foley et al., 2010). Unlike traditional fisheries management, where the focus is on a single ecosystem service like food production, the aim of EBM is to sustain the capacity of marine ecosystems to deliver a suite of services over the long term (Halpern et al., 2010; Tallis et al., 2010). To achieve this, EBM gives explicit consideration to how humans interact with and use natural resources, as well as to the inherent trade offs necessary to meet multiple, often conflicting, management goals (Gallagher et al., 2004; Halpern et al., 2010).

1.4 The promise of Marine Protected Areas

1.4.1 MPAs: a brief history

Marine protected areas are a widely advocated tool for marine conservation and fisheries management and are considered to be a potentially effective means to achieve EBM goals. Protected areas in the marine realm were a more recent innovation than in terrestrial ecosystems, principally because human-driven changes to the sea are much harder to perceive than those to the land. It is for this reason that more than a decade after the establishment of the terrestrial Yellowstone Park in the USA, and at a time when the American buffalo (*Bison bison*) and the Passenger pigeon (*Ectopistes migratorius*) were nearing extinction, the prevailing view in both the arts (Verne, 1869) and the sciences (Huxley, 1883) was that the ocean's goods were inexhaustible, as the quotes at the start of this chapter attest (Fraschetti et al., 2011). It was not until almost a century later, in the 1960s and 1970s, that marine protected areas finally began to gain traction, spurred on by recovery of stocks in the North Sea after the cessation of fishing activities there during the Second World War (Roberts, 2010). Today, there are more than 17,000 MPAs located in offshore and coastal areas worldwide (Thomas et al., 2014).

1.4.2 MPAs: definition, management categories and governance types

An MPA is defined by the International Union for Conservation of Nature (IUCN) as:

> *"A clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values."*
> (Day et al., 2012).

In recognition of the wide variety of uses and objectives potentially embodied by such a definition, the IUCN has developed guidance that categorises an MPA by six management types and four governance types, summarised in Table 1.2 (Day et al., 2012; Dudley, 2008).

TABLE 1.2 MARINE PROTECTED AREA MANAGEMENT CATEGORIES, GOVERNANCE TYPES AND EXAMPLES

IUCN Category	Explanation	Example
Management Category		
Ia Strict nature reserve	Strictly protected for biodiversity and also possibly geological/ geomorphological features, where human visitation, use and impacts are controlled and limited to ensure protection of conservation values	Cousin Island, Seychelles (Chapter 5)
Ib Wilderness area	Usually large unmodified or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, protected and managed to preserve their natural condition	The Chassahowitza Wilderness, USA.
II National park	Large natural or near-natural areas protecting large-scale ecological processes with characteristic species and ecosystems, which also have environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities	Wakatobi Marine National Park (Chapter 4)
III Natural monument or feature	Areas set aside to protect a specific natural monument, which can be a landform, sea mount, marine cavern, geological feature such as a cave, or a living feature such as an ancient grove	The Blue Hole Natural Monument, Belize
IV Habitat/species management area	Areas to protect particular species or habitats, where management reflects this priority. Many need active and regular interventions to meet the needs of particular habitats or species, but this is not a prerequisite	South Water Caye Marine Reserve, Belize
V Protected landscape or seascape	Where the interaction of people and nature over time has produced a distinct character with significant ecological, biological, cultural and scenic value, and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values	Apo Island, Philippines (Chapter 3)
VI Protected areas with sustainable use of natural resources	Areas that conserve ecosystems, together with associated cultural values and traditional natural resource management systems. Generally large, mainly in a natural condition, with a proportion under sustainable natural resource management and where low-level non-industrial natural resource use compatible with nature conservation is seen as one of the main aims	Velondriake, Madagascar (Chapter 5)
Governance types		
Governance by government	Federal or national ministry/agency in charge; sub-national ministry/agency in charge; government-delegated management (e.g. to NGO)	Great Barrier Reef, Australia (Chapter 1)
Shared governance	Collaborative management (various degrees of influence); joint management (pluralist management board); transboundary management (various levels across international borders)	Cook Islands Marine Park (Chapters 2 and 3)
Private governance	By individual owner; by non-profit organisations (NGOs, universities, cooperatives); by for-profit organisations (individuals or corporate)	Chumbe Island Coral Park, Tanzania (Chapters 3 and 5)
Governance by indigenous peoples and local communities	Indigenous peoples' conserved areas and territories; community conserved areas – declared and run by local communities	Velondriake, Madagascar (Chapter 5)

Sources: (Day et al., 2012; Dudley, 2008)

In addition to the multiple management categories, governance types and uses of MPAs, a wide variety of terms are used to describe them. Such terms vary between – and often within – countries and include: marine sanctuary, marine park, marine reserve, marine and coastal protected area, reference area, biosphere reserve, marine conservation zone, marine conservation area, nature reserve, marine management area, no-take zone, and national park, as well as a slew of acronyms including SSSI, SAC, HMPR, MCPA, MCZ, SPA, ASC and NTZ. To further muddy the waters, different nations do not always agree on the same meaning for a single term, even within the same region. In Kenya, marine parks prohibit all fishing and resource extraction, while in neighbouring Tanzania, they permit some fishing. Conversely, marine reserves in Kenya allow low-level fishing, but their namesakes in Tanzania prohibit all extraction.

For clarity, in this thesis, I restrict myself as much as possible to two terms: i) marine protected areas, places off limits to exploitation or excessive use; and ii) marine reserves, permanent MPAs where all extraction is prohibited.

1.4.3 MPAs: benefits to nature and people

Meta-analyses have shown that MPAs, especially permanent marine reserves, can increase the average biomass, density, size and diversity of species within their boundaries (Halpern, 2003; Lester et al., 2009, Table 1.3). Evidence also indicates that spillover of adults (Gell and Roberts, 2002; Russ et al., 2004; Russ and Alcala, 2010) and export of larvae beyond protected boundaries can benefit surrounding fisheries (Harrison et al., 2012; Pelc et al., 2010; Yamazaki et al., 2014), particularly when multiple areas are established together as an interconnected network (Gaines et al., 2010). As a source of large, fecund individuals, well-designed and enforced MPAs can additionally help to increase the resilience to anthropogenic stressors of targeted populations, and also protect many other ecosystem services, including vital tourism and recreation opportunities (Hastings et al., 2012; Micheli et al., 2012). Recent research suggests that further expanding the coverage of MPAs to around 30% of global territorial waters could create between 150,000 and 180,000 jobs and generate economic benefits of between US$490 billion and US$930 billion (Brander et al., 2015).

TABLE 1.3 THE ECOLOGICAL EFFECTS OF MARINE RESERVES. A COMPARISON OF META-ANALYSES THAT HAVE ASSESSED MEAN PERCENTAGE CHANGE IN KEY BIOLOGICAL VARIABLES.

Study	N	Biomass	Density	Size	Richness
Mosquera et al. (2000)	12	-	272%	-	-
Côté et al. (2001)	19	-	25%*	-	11%
Halpern (2003)	89	352%	151%	29%	25%
Claudet et al. (2008)	19	-	146%	-	8%*
Lester et al. (2009)	124	446%	166%	28%	21%
Molloy et al. (2009)	33	-	66%	-	-
Stewart et al. (2009)	30	59%	54%	-	114%

* Not statistically significant; – no data available; N = number of reserves studied

1.4.4 MPAs: progress towards international biodiversity targets

Given the evidence for the potential benefits of MPAs, It is perhaps unsurprising that establishing greater numbers of MPAs has been at the centre of global marine conservation policy efforts since at least 2006, when the parties to the Convention on Biological Diversity (CBD) committed to effectively conserving 10% of the world's oceans by 2012 (CBD, 2006). Initial progress was slow – causing the parties to move the deadline to 2020 (CBD, 2010; Wood et al., 2008b) – but the pace of establishment has since accelerated (Spalding et al., 2013; Thomas et al., 2014). MPAs now cover 12.3 million km^2, 3.4% of world's ocean, 10.9% of territorial waters (0-12nm) and 8% of Exclusive Economic Zones (12-200nm) (Thomas et al., 2014). So pronounced is this increase that the 10% goal, known as Aichi target 11, is on track to be achieved, perhaps even before 2020 (Spalding et al., 2013).

However, a few very large MPAs are largely responsible for this upsurge, a trend that looks set to continue as huge new areas are established in the waters of the Cook Islands, New Zealand, Chile, Pitcairn Island, Palau, and the Pacific Remote Islands (BBC News, 2015, 2014; Devillers et al., 2014; Milman, 2015; Mooney and Eilperin, 2015; Nature, 2015; Pala, 2013). Further, most of these MPAs are located in uninhabited or low-population-density areas (Spalding et al., 2013) or in places that are unpromising for extractive activities (Devillers et al., 2014)

1.5 The shortcomings of Marine Protected Areas

1.5.1 A lack of effective management

The focus on more, large MPAs has come at a cost: far less attention has been paid to ensuring they are effective at achieving their aims (but see Pelletier *et al.*, 2005, 2008; Pomeroy *et al.*, 2005; Beliaeff and Pelletier, 2011; Burke *et al.*, 2011; Fox *et al.*, 2014). Evidence suggests that, despite their widespread use and increasing popularity, MPAs are often 'paper parks' with little conservation value (Halpern, 2014; Mora et al., 2006). Though negative assessments of sites are rare (but see Caveen et al., 2014), global (Balmford et al., 2004; Burke et al., 2011) and regional (e.g. Samoilys and Obura, 2011) evaluations suggest that many MPAs lack adequate management, enforcement and financing.

Even some of the most iconic MPAs have not been immune from criticism. For example, a recent study of the 12,000km^2 of the Great Barrier Reef (GBR) that was closed to fishing following the park's 2004 rezoning found no evidence of recovery in catch levels within the broader GBR in the following 9 years (Fletcher et al., 2014).

This conclusion contrasts markedly with the calculations used to justify the rezoning, which predicted that recovery would become apparent within three years (Fletcher et al., 2014). Between 1985 and 2012, the reef lost half its coral cover (De'ath et al., 2012), a decline that is predicted to continue in coming years (Cooper et al., 2012), with climate change, poor water quality, coastal development and illegal fishing all posing major threats to the health of the reef (Great Barrier Reef Marine Park Authority, 2014). So great are these impacts that the reef narrowly avoided being added to the World Heritage "in danger" list in May 2015 and given an unwelcome new name: the Not-So-Great Barrier Reef (Brodie and Waterhouse, 2012; Slezak, 2015).

1.5.2 A lack of focus on social dimensions

Historically, the biophysical dimensions of MPAs have received the greatest attention from policy makers, NGOs, resource managers and researchers. However, in many cases, it is the social factors – the social, cultural, political and economic variables that shape individual and collective behaviour – which have a greater influence on the design and performance of MPAs, particularly when they are established in areas of high human population density (Mascia, 2004; Pomeroy et al., 2007). For example, researchers in southwest Madagascar spent two years synthesising biophysical data to develop a zoning plan for the Velondriake MPA, only for affected coastal communities to request a wholescale boundary revision prior to implementation (Harris, 2007). The changes were made, enhancing the perceived legitimacy of the area amongst local stakeholders, and Velondriake has gone on to be widely regarded as a socially successful MPA (Cripps and Harris, 2009; Harris, 2007; Oliver et al., 2015).

In addition to being the *product* of social processes, MPAs also have social *consequences* (Chaigneau and Brown, 2016; Mascia, 2004). As with other forms of resource management, MPAs apportion rights to access and use marine resources to individuals and groups, and in so doing, indirectly and directly affect society (Mascia, 2004). Some of the most potentially disruptive societal impacts have occurred where the establishment of an MPA has lead to a transition away from a fishing economy in favour of one that is more tourism-dominated. The establishment of the El Nido-Taytay Managed Resource Protected Area in the Philippines, for example, helped to engender a rapid growth in tourism arrivals, with annual visitor numbers increasing from very few to almost 18,000 a decade later (Hodgson and Dixon, 2000). Attracted by the greater economic opportunities on offer, many residents ceased fishing and switched to working as boat captains and tour guides in the new marine reserve, where they were joined by migrants from other areas of the Philippines, altering community dynamics. (pers. comm. Irma Rose Marcelo).

Researchers have often overlooked the complexities of the social consequences of MPA implementation. For example, Roberts et al. (2001) tout the Soufrière marine protected area in St. Lucia as an example of win-win conservation, delivering both increased social and biological benefits. Others, however, are critical of this interpretation. Trist (1999) for example, notes that "Soufrière's marine space is a site of heated political struggles" and decries the misleading "tropical tranquillity implicit in tourism marketing", while Christie (2003, p. 2) contends that

> *the conclusion that the Soufrière MPA is widely accepted is based on poorly described survey methods that are not up to standards of rigorous social science and is offered without consideration of equally pertinent narrative accounts of conflict and local resistance.*

This disconnect between historical tendencies to consider MPAs through a simple biophysical lens and the reality that MPAs are complex, linked social-ecological systems (Fraschetti et al., 2011; Pollnac et al., 2010) is likely to underpin the lack of effectiveness of many initiatives, or else the creation of MPAs that are ecologically successful but socially damaging, creating a false sense of security among managers that conservation is being achieved whilst causing economic hardship and social displacement among small-scale fishing communities (Christie, 2004; Claudet and Guidetti, 2010). This failure to adequately engage communities in supporting marine conservation and resource management efforts can be considered as a defining paradox of the sector. Since the economies of coastal communities in many countries invariably depend on goods and services produced by marine ecosystems, these communities theoretically stand to gain the most from marine protection, providing local interests and resource rights are factored appropriately into protected area design (Mascia, 2004; Pomeroy et al., 2007).

1.6 Overcoming MPA shortcomings

It is clear from the preceding section that if we want to reduce the high number of paper parks and improve the conservation value of MPAs, it is sensible to focus both on the potential of community-centred approaches to deliver marine resource management and protection, as well on interdisciplinary approaches to evaluate this potential.

This dual focus is particularly necessary when considering conservation and management along populated coastlines in the tropics, since it is in these contexts, where resource pressure and reef degradation are often most acute, and where food security is of paramount concern for hundreds of millions of resource users, that the need for effective MPAs is arguably greatest (Gutierrez et al., 2011; Veron et al., 2009).

Paradoxically, it is in precisely these contexts where it is most challenging to create successful and durable MPAs. Against a backdrop of rising coastal populations and communities inadequately compensated for loss of access to fishing grounds, the five key features that underpin the most effective MPAs – fully protected, well enforced, established for more than 10 years, larger than $100km^2$ and in isolated locations (Edgar et al., 2014) – tend to be either absent or unachievable. As such, figures for coastal MPA coverage lag far behind those for offshore areas (Maire et al., 2016; Thomas et al., 2014). Furthermore, even where large areas do persist along coastlines, they are increasingly threatened by efforts to downgrade, downsize, or degazette them, prompting increasing interest in other, more inclusive approaches to coastal resource management and conservation (Mascia and Pailler, 2011).

1.6.1 The need for alternative modes of governance: towards co-management of coastal resources with LMMAs

Co-management, which occurs when local communities share responsibility for marine resource management with governments or other partners, has emerged in recent decades as an alternative to the dominant paradigms in marine conservation that government-mandated protected areas must be the cornerstone of marine protection efforts, and in fisheries management that privatisation is essential to prevent the tragedy of the commons (Cinner et al., 2012c; Gutierrez et al., 2011; Hardin, 1968). Using examples from fisheries and other common-pool resource systems, several authors have argued that co-management can create sustainable fisheries (e.g. Beddington et al., 2007; Berkes, 2007; Cinner et al., 2012c; Costanza et al., 1998; Dietz et al., 2003; Gutierrez et al., 2011; Ostrom, 1990; Pretty, 2003; Wamukota et al., 2012). For examples, a recent review of 130 initiatives in 44 countries found that co-management could contribute to the successful management of aquatic resources and concluded that it was the only practical solution for most of the world's fisheries (Gutierrez et al., 2011).

1.6.1.1 LMMAs: history and definition

Across the Indo-Pacific, areas where marine resources are at least in part under community control are usually termed "locally managed marine areas" or LMMAs (Govan, 2009). Broadly similar to marine protected areas but managed for sustainable, long-term use rather than biodiversity conservation itself, LMMAs have occurred largely as a result of disillusionment with top-down, centralised government interventions (Cohen and Foale, 2013; Govan et al., 2006; Wamukota et al., 2012).

The LMMA approach has its origins in informal systems of community-based marine management that were often used in many parts of the Pacific to replenish stocks ahead of feasts, to protect sacred sites, or to mark the death of prominent community members (Christie and White, 1997; Cohen and Foale, 2013; Johannes, 1978). Motivations for closures were thus driven by socio-cultural traditions rather than sustainable use, though in some situations there may also have been fisheries management benefits (Cohen and Foale, 2013; Foale et al., 2011).

In 2000, at meetings in the Philippines and Fiji, regional marine conservation practitioners and community leaders coined the phrase "locally managed marine area". The attendees defined an LMMA as:

An area of nearshore waters and coastal resources that is largely or wholly managed at a local level by the coastal communities, land-owning groups, partner organizations, and/or collaborative government representatives who reside or are based in the immediate area
(Govan and Tawake, 2009; Parks and Salafsky, 2001)

LMMAs are referred to by several other names in the Indo-Pacific (Table 1.4).

TABLE 1.4 NAMES USED TO DESCRIBE LMMAS IN THE INDO-PACIFIC

Country	Name
Cook Islands	Ra'ui site
Fiji	Tabu area; Traditional reserve; Community-protected area
French Polynesia	Rahui
Hawaii	Kapu zone; Traditional MPA; Cultural marine conservation district
Kenya	Community Conservation Area; tengefu; hifadhi za kijamii; vilindo vya wenyeji
Indonesia	Sasizen; Community-based marine protected area; Sasi
Malaysia	Community-based marine protected area
Marshall Islands	Mo
New Zealand	Rahui
Palau	Bau zone
Papua New Guinea	Tabu area; Customary area
Philippines	Community-based marine protected area
Samoa	Sa
Solomon Islands	Tambu zone; Community-managed reserve; Community conservation area;
Tanzania	Collaborative Fisheries Management Area
Tokelau	Lafu
Tuvalu	Tapu
Vanuatu	Tabu

Sources: Rocliffe and Peabody (2012), Samoilys and Obura (2011), Parks and Salafsky (2001), Govan and Tawake (2009)

1.6.1.2 LMMAs: contemporary use

LMMAs often blend local and scientific knowledge as well as customary and contemporary management systems, with many employing a range of techniques including permanent no-take marine protected areas, periodic closures, gear restrictions, species-specific reserves, secure tenure rights and alternative livelihood strategies (Jupiter et al., 2014; Mills et al., 2011).

Although the level of community involvement and the overall management model is context-specific to an extent, a key aspect is local control. Technical support may be provided by government agencies, private sector stakeholders or non-governmental organisations, but it is the resource users themselves who make most of the management decisions, including the location of any protected areas (Evans et al., 2011; Gutierrez et al., 2011; Tawake, 2007). No global inventories exist, but evidence from the Pacific alone suggests that more than 500 communities in 15 countries manage 12,000km^2 of coastal resources, 1,000km^2 of which is within area-based closures (Govan, 2009). There are also likely to be thousands more communities managing resources informally who do not appear on official lists (Cohen et al., 2014; Govan, 2009).

LMMAs have also been established in temperate regions, where they are more typically referred to as territorial use rights in fisheries, or TURFs (e.g. Aburto et al., 2013; Castilla and Defeo, 2001; Gelcich et al., 2012). TURFs that include a no-fishing area are known as TURF-reserves (Afflerbach et al., 2014). Although there is a subtle distinction between LMMAs and TURFs– most LMMAs assign rights to fish in a given area, while all TURFs do – in practice the term is used interchangeably (Afflerbach et al., 2014; Auriemma et al., 2014; Barner et al., 2015).

The central objective of many LMMAs is to improve the long-term sustainability of fisheries, though they are frequently initiated to meet other goals such as enhancing livelihoods, strengthening customs or empowering the community (Cohen et al., 2014; Jupiter et al., 2014). When effective, LMMAs can improve compliance with regulations by resource users, encourage responsible fishing and enhance adaptive capacity (Gutierrez et al., 2011; Léopold et al., 2013; Levine and Richmond, 2014). They have also been associated with short-term increases in resource abundance (Bartlett et al., 2009; Cinner et al., 2012c, 2005; Oliver et al., 2015; Pollnac et al., 2001) and have shown some promise as a means to safeguard food security and to address coastal poverty (Oliver et al., 2015; Weiant and Aswani, 2006)

However, evidence that LMMAs can achieve long-term ecological and social goals is scarce and inconclusive (Cohen et al., 2014; Léopold et al., 2013). For example, in an examination of 42 marine co-management systems in Kenya, Tanzania, Madagascar, Indonesia and Papua New Guinea, Cinner et al. (2012c) found that although 88% of resource users surveyed reported high levels of compliance and 54% perceived a benefit to their livelihoods, co-management could also create a degree of social inequality by favouring wealthier users. They additionally found that fish biomass in co-managed areas was generally greater than in areas without local management, though substantially lower than in no-take marine reserves in the same countries (Cinner et al., 2012c).

1.6.1.3 LMMAs as a type of MPA

For an area to be regarded as an MPA under the definition and framework in Section 1.4.2, its primary objective must be to achieve nature conservation outcomes (Day et al., 2012). As such many LMMAs, which are often managed for sustainable extraction of fish or invertebrates (Chapters 3 and 5), might not historically have been considered MPAs under the definition, irrespective of whether they deliver conservation benefits. The same is true for marine and coastal areas where the explicit focus is on tourism. In both cases, the "associated ecosystem services" that are delivered (food production in the first example and recreation and tourism services in the second) could be regarded as conflicting with the aim of nature conservation.

More recent guidance has provided greater clarity on this issue. *The Promise of Sydney*, a blueprint for MPA practice that emerged from the 2014 IUCN World Parks Congress, makes the following primary recommendation

> *"To urgently increase the ocean area that is effectively and equitably managed in ecologically representative and well-connected systems of MPAs or other effective conservation measures. This network should target protection of both biodiversity and ecosystem services and should include at least 30% of each marine habitat. The ultimate aim is to create a fully sustainable ocean, at least 30% of which has no extractive activities."*
> (IUCN, 2014a).

Given that less than 1% of the ocean (Thomas et al., 2014) is currently protected by marine reserves, a goal of 30% no-take coverage by 2030 is an ambitious one. Nonetheless, by recommending that the expanded global network should aim to protect both biodiversity and ecosystem services, the *Promise of Sydney* opens the door to increasing recognition and use of management tools where the primary objective is not necessarily biodiversity conservation.

1.6.2 The need for interdisciplinary research approaches

Over the past decade, there has been increasing recognition of the linked social, economic and ecological dynamics inherent in MPAs and consequently, of the role and importance of a broad spectrum of disciplines in improving protected area design and performance (Christie and White, 2007; Cinner et al., 2012c; Claudet and Guidetti, 2010).

Contemporary MPA research encompasses not only ecological studies centred on how fish and invertebrate assemblages respond to protection (Chapter 3), but also statistical investigations into improving impact assessment (Chapters 3 and 4), social and economic analyses exploring how MPAs benefit different stakeholders (Chapter 2), legal examinations of MPA governance (Chapter 5), mathematical modelling to design areas and predict protection effects, genetic and otolith analyses to study connectivity among MPAs, and fisheries science and economic assessments of how MPAs can achieve fisheries management goals (Fraschetti et al., 2011).

From a social-ecological perspective, many authors have used the work of Elinor Ostrom (Ostrom, 2009, 2007, 1990) to guide their analyses. Early work (Pomeroy et al., 2001; Siar et al., 1992) drew primarily on Ostrom's institutional design principles, a set of eight rules for developing robust and successful co-management arrangements, distilled from an examination of largely terrestrial case studies (Ostrom, 1990).

The principles comprise

i. Clearly defined boundaries which delineate both who has resource use rights, as well as the extent of the resource
ii. Rules adapted to local social and ecological conditions
iii. Collective choice arrangements to allow stakeholders to participate in the decision making process
iv. Accountability mechanisms to promote effective monitoring
v. Graduated sanctions for rule violators
vi. Conflict resolution mechanisms that are both low cost and accessible
vii. Recognition by higher-level authorities of the rights of the community to self organise
viii. Organisation in the form of multiple levels of nested enterprises (Ostrom, 1990)

The mechanisms and models of human behaviour that underpin the principles are complex and multifaceted, but are centred on notions of "fallible, norm-adopting individuals who pursue contingent strategies in complex and uncertain environments" (Ostrom, 1990, p. 185). As such, they accord with North's (1990) characterisation of institutions as structures for lowering uncertainty in complex and changeable environments (Cox et al., 2010), suggesting that by doing so, trust can be built and sustained, and can provide a foundation for collective action.

Several researchers have concerned themselves with the utility and validity of the principles (See Cox et al., 2010 for a review). Support remains strong, though some authors have criticised their theoretical underpinnings, or argued that they are too prescriptive for the broad range of contexts in which they might be applied (Agrawal, 2001; Cox et al., 2010; Stevenson and Tissot, 2014).

In response, Ostrom contextualised and further refined the principles into a multilevel, nested Social-Ecological Systems (SES) framework for evaluating SES systems (Ostrom, 2009).

As Figure 1.1 shows, the framework emphasises that complex SESes have four core and relatively discrete subsystems, which nonetheless interact to create outcomes at the SES level. These outcomes in turn feed back to alter the subsystems as well as other SESes of differing size and complexity (Ostrom, 2009). Each of the four subsystems – i) resource systems (e.g. a tropical coastal fishery); ii) resource units (fish, invertebrates); iii) governance systems (organisations and rules that govern resource use); and iv) users (coastal fishers and other stakeholders) – is composed of a number of second-level variables (e.g. number of users, socio-economic attributes of users, history of use, social capital), which are in turn made up of deeper-level variables (Ostrom, 2009; Stevenson and Tissot, 2014).

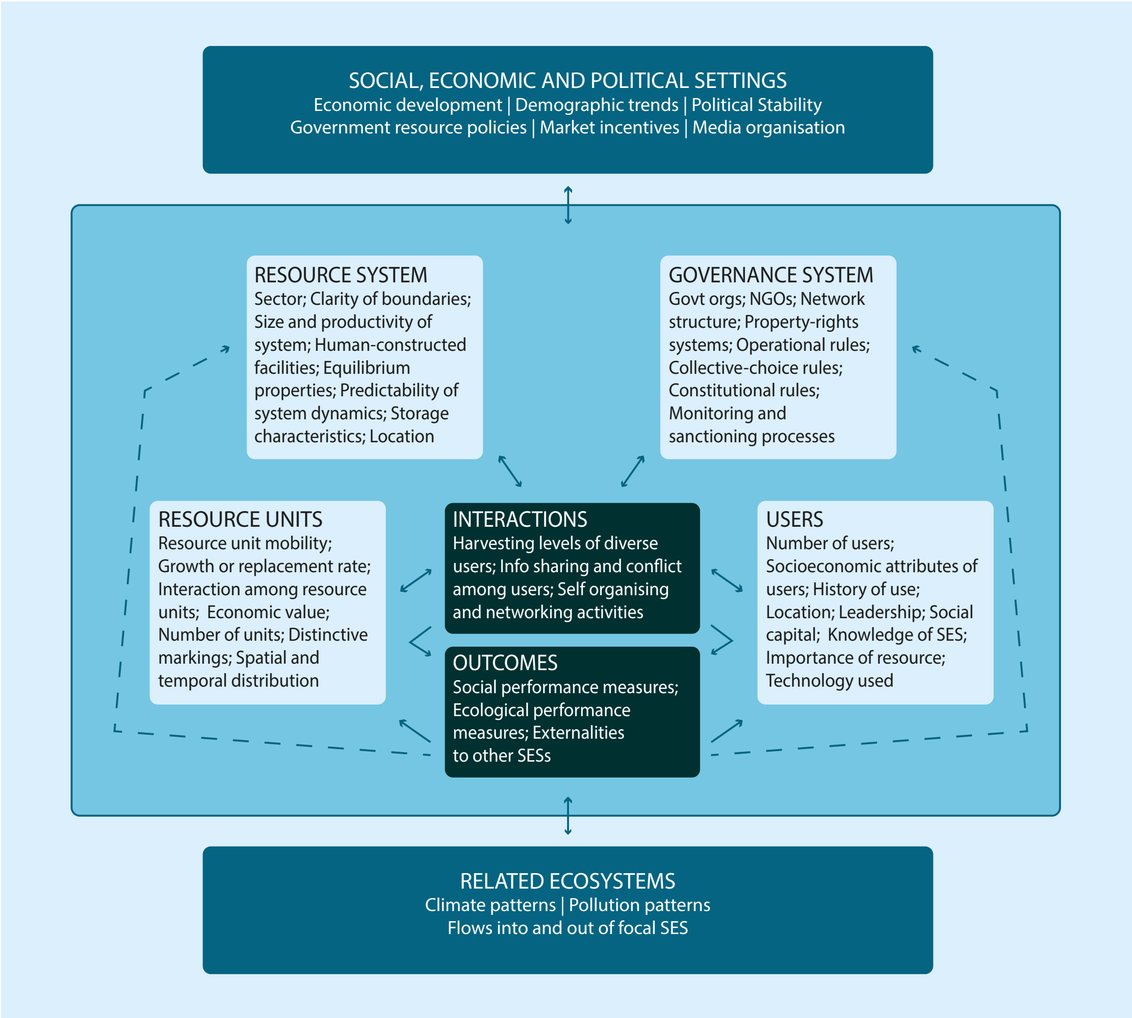

FIGURE 1.1 THE CORE SUBSYSTEMS AND SECOND-LEVEL VARIABLES IN A FRAMEWORK FOR ANALYSING SOCIAL-ECOLOGICAL SYSTEMS (OSTROM, 2009)

A growing body of both original (Cinner et al., 2012c; MacNeil and Cinner, 2013) and meta-analytical research (Evans et al., 2011; Gutierrez et al., 2011) has made use of the framework to evaluate the effectiveness of marine co-management arrangements. This work has been complemented by several broader research and policy initiatives with an explicit social focus. Once such initiative presently underway is *Solving the Mystery of Marine Protected Area Performance*, an interdisciplinary research programme involving more than twenty institutions seeking to integrate and analyse biophysical and social data from MPAs across the globe, and to better understand the relationship between MPA governance and ecosystem structure, function, and services (SESYNC, 2013). Another is the development of a new *Social Compact* at the IUCN World Parks Congress (IUCN, 2014b). The Compact, which forms part of the *Promise of Sydney* discussed in the preceding section seeks to promote a conservation ethic that "supports diverse knowledge systems and values, [and] delivers rights-based and equitable conservation for improved governance of natural resources and tangible benefits for livelihoods." The *Promise of Sydney* further sets out a commitment to "enhance diversity, quality and vitality in governance and management, including the appropriate recognition and support of areas conserved by Indigenous Peoples [and] local communities" (IUCN, 2014c). Such a commitment is explicit acknowledgement of the trend away from government-run conservation fortress that exclude people and towards sustainable areas that create trade-offs between conservation and human use (Fraschetti et al., 2011).

1.7 LMMAs: key interdisciplinary research gaps

The recently established and informal nature of many LMMA initiatives means that several interdisciplinary questions remain unanswered. In particular, little is known about the long-term effectiveness of LMMAs at achieving their objectives, with initiatives often lacking the necessary financial resources to sustain both effective management and the collection and evaluation of robust socio-ecological evidence on which such management often depends (Frid and Crowe, 2015; Gutierrez et al., 2011; Jupiter et al., 2014; Léopold et al., 2013; Levine and Richmond, 2014). Moreover, almost no research has considered the status and scale of LMMAs outside the Western and Central Pacific region. Each of these gaps, and the contribution of this thesis towards addressing them, is discussed below and in the sections that follow.

1.7.1 Role of tourism in improving effectiveness of LMMAs

The short-term costs of marine protection efforts – which fall largely on coastal communities – are often perceived to outweigh future benefits, which may be uncertain (Buxton et al., 2014), diffuse (Hanna, 2004), slow to accrue (Russ and Alcala, 2010), and shared with others who may not have invested in protection efforts (Pomeroy et al., 2007). This challenge is particularly great among small-scale fishing communities throughout the tropics, where a high degree of coastal poverty results in strong dependence on fishing as a primary or sole source of food and income (Walmsley et al., 2006), and where high discount rates (fishers' time preference for immediate versus delayed reward) can hamper long term planning (Oliver et al., 2015; Pomeroy et al., 2001).

With the benefits of marine conservation thus less than assured in these contexts, recent years have seen a surge in interest among conservation and development practitioners to identify management models that can make protected areas more economically acceptable to coastal communities (Niesten et al., 2013). Since healthy coasts generate billions of dollars of value for the tourism and recreation industry every year, tourism has proven to be a popular means of supporting conservation and development and has a broad base of support (Kiss, 2004; Millennium Ecosystem Assessment, 2005).

For example, despite the criticism levelled at the GBR (Section 1.5.1), almost 70,000 people are employed as a result of the estimated US$5.7 billion of economic activity it generates each year (Deloitte Access Economics, 2013; Hoegh-Guldberg et al., 2004). By comparison, the fishing industries associated with the reef are worth around US$120 million per year (Hoegh-Guldberg et al., 2015).

Overall, coral reefs are estimated to contribute nearly US$30 billion to the global economy each year, a third of which (US$9.7 billion) is generated by nature-based tourism, including snorkelling and diving (Cesar et al., 2003; Millennium Ecosystem Assessment, 2005). Coastal tourism is among the fastest-growing sectors of the global tourism industry, and forms a major part of the economies of many Small Island Developing States (SIDS), a group of 52 developing countries that share common development challenges, arising from their small size, exposure to global environmental challenges, remoteness, limited resource base, and vulnerability to external shocks (Hall, 2010; Lewis and Roberts, 2010; World Tourism Organization, 2012). Marine protection efforts in SIDS are typically funded by NGOs, so tourism-centred approaches are especially attractive in these contexts, since they may provide a financial mechanism for sustaining conservation programming over the longer term by decoupling it from time-bound donor support (Kiss, 2004).

To date, many SIDS have established LMMAs (Govan and Tawake, 2009) and the explicit recognition in the Sustainable Development Goals of tourism as a means to increase the economic benefits to SIDS from the sustainable use of marine resources suggests that more will follow (UN, 2015). Although it is hard to determine accurate numbers given the often informal nature of locally managed initiatives, evidence suggests that there are established initiatives in more than 20 tropical countries (Rocliffe and Peabody, 2012). Taken together, these nations welcomed more than 10.8 million international arrivals in 2012 and are expected to steadily increase their respective market shares through to 2030 (World Tourism Organization, 2014a, 2011).

There are two clear research opportunities to better understand the opportunities that tourism can offer to LMMAs as well as some of the challenges that may arise.

First, coastal tourism in SIDS, as elsewhere, is underpinned by the appeal of the natural environment, with research indicating that visitation is often contingent on healthy reefs, beaches and other ecosystems (Fujita et al., 2013; Hampton and Jeyacheya, 2013; Hay, 2013; Lutchman et al., 2005; Uyarra et al., 2005; Vianna et al., 2012; World Tourism Organization, 2012). Given both the centricity of tourism to the economy of many SIDS as well as tourist preference for attractive natural features, visitor perceptions of marine protection and resource management efforts should be reflected in the development and prioritisation of management actions (Sayan and Karagüzel, 2010; Torres-Sovero et al., 2012). Moreover, since SIDS are particularly vulnerable to climate change impacts, heeding and acting on such perceptions may enhance destination competitiveness, leading to greater receipts from tourism, and a potential increase in funding and capacity for mitigation initiatives. Yet despite these theoretical benefits, little research attention has been directed towards visitor perceptions in tourism-dependent island states.

Second, since many SIDS have both high levels of tourist visitation and increasing numbers of LMMAs, one approach for sustaining locally-centred initiatives over the long-term could be through the involvement of coastal hotels as co-management partners. However, although the numbers of both protected areas and tourists are rising globally, hotel engagement remains limited (Spalding et al., 2013; Stolton et al., 2014; World Tourism Organization, 2014a). Initiatives to date, typically termed entrepreneurial MPAs (Bottema and Bush, 2012; Colwell, 1998; de Groot and Bush, 2010) or hotel managed marine reserves (HMMRs, Svensson et al., 2009), have tended to be costly, complex and imposed with limited consultation, and have thus met with only partial success.

Examples of more informal and collaborative arrangements between hotels and local communities do exist, for example in the Cook Islands (Miller, 2009), but the effectiveness of this model of LMMA support and its potential to scale to other tropical developing countries has yet to be examined.

1.7.2 Role of research design in improving assessments of LMMAs

Judging the effectiveness of conservation and management interventions is of central importance to conservation science and is an essential component of adaptive management and effective decision making (Ferraro and Pattanayak, 2006; Osenberg et al., 2011). The fundamental question that underpins any evaluation is easily articulated but challenging to measure: what would have happened if there had been no intervention (Ferraro and Pattanayak, 2006; Osenberg et al., 2011). In ecological assessments of area-based management initiatives such as MPAs and LMMAs, this question is usually tackled with Control-Impact (CI) designs that typically use belt transects or other underwater survey methods to sample from multiple locations within a single managed and a single control site (Osenberg et al., 2011). On each transect, indicators like fish abundance or diversity are estimated and compared at the site level. Where a statistically significant difference in an indicator is found between two sites, this is considered evidence of an effect of the managed site (Osenberg et al., 2006).

Because CI designs are easy to analyse and have low resource requirements, they have proven to be attractive for researchers and managers alike. Indeed, 70% of the 89 studies included in the most highly cited synthesis of effects of no-take marine reserves used CI designs (Halpern, 2003). However, such designs can have substantial drawbacks (Claudet and Guidetti, 2010; Frid and Crowe, 2015; Underwood, 1992, 1991). A single-point-in-time comparison that detects a significant difference between a single control site and a single LMMA, and ascribes this difference to the effects of the LMMA will be pseudoreplicated (Hurlbert, 1984, 2009). While the two sites being compared may be alike, they will not be identical, and it is very probable that there would have been statistically significant differences between them prior to the LMMA being established (Osenberg et al., 2006). Such a design therefore lacks the replication necessary to logically disentangle the effects of the LMMA from other sources of spatial variation, and risks the effects of this management tool being overestimated (Claudet and Guidetti, 2010; Osenberg et al., 2011).

Given the enduring popularity of MPAs, and the increasing attention LMMAs are receiving from international agencies, NGOs and the donor community, it is timely to reconsider whether control impact assessments can provide a sufficiently robust evidence base to enable management actions to be targeted effectively.

Regardless of the design adopted, there are several biases with the potential to undermine the accuracy of comparative transect surveys (Elphick, 2008; Thompson and Mapstone, 1997). Usually, such biases will have a roughly equal influence on estimates from both control and managed sites, and so rarely confound results substantially (Edgar et al., 2004). Further, they can generally be minimised though comprehensive training and calibration of those who conduct the assessments (Thompson and Mapstone, 1997). However, not all biases act so consistently. For example, expectation bias may consciously or unconsciously lead a researcher to favour information supportive of his or her hypothesis, resulting in a more positive evaluation of an experimental intervention than of a control (Burghardt et al., 2012; Hrobjartsson et al., 2013; Tuyttens et al., 2014).

This has important yet currently unaddressed implications for evaluations of MPAs and LMMAs. Because researchers know which sites have received management (i.e. treatment) and which are controls, there is a danger that they may unconsciously overestimate the effects of management, or even detect a change where none has occurred. There is therefore an urgent need to examine the potential for expectation bias is conservation research and to determine methods for minimising it in future research designs.

1.7.3 Status and geographical extent of LMMAs

To date, the development and establishment of LMMAs has occurred overwhelmingly in the Western and Central Pacific (see, for example Cohen et al., 2014; Govan, 2011, 2009; Govan et al., 2012, 2011, 2008, 2006; Govan and Tawake, 2009; Hastings et al., 2012; Jupiter et al., 2014; Tawake, 2007). According to Govan and Tawake (2009), there are active LMMA initiatives in more than a dozen nations throughout this region. In eight countries and territories – Fiji, Indonesia, Palau, Papua New Guinea, Philippines, Pohnpei, Solomon Islands and Vanuatu – there are country-wide networks designed to facilitate information sharing and peer-to-peer learning amongst coastal communities (Govan and Tawake, 2009; Rocliffe and Peabody, 2012). These networks in turn form part of a broader regional practitioner LMMA network.

However, LMMAs are being established in other parts of the tropics, especially in the Western Indian Ocean (WIO), where little is known about the status or extent of LMMAs, or the legal frameworks that underpin them (but see Katikiro et al. (2015) for a discussion of the barriers to LMMA adoption in Tanzania and Cinner et al. (2009) for an overview of frameworks supporting co-management in Kenya and Madagascar). In addition, because LMMAs in the WIO have tended to operate in relative isolation, with little communication or coordination between support organisations or implementing communities, there is also a critical need to explore the feasibility of a region-wide network of practitioners to share best practice on management topics such as financing and evaluation and to encourage the development of further LMMAs.

1.8 Research approach, aims and objective

This thesis is presented as a series of chapters written for peer-reviewed publication but reformatted to accord with institutional requirements. The four data chapters integrate a variety of novel research approaches, borrowing not only from ecology, conservation science and socio-economics, but also from disciplines as diverse as medicine, psychology and business management. Given the self-contained nature of each data chapter, there is some unavoidable repetition between the general introduction and discussion and the data chapters themselves.

For source material, this thesis relies on primary cross-disciplinary research in the Cook Islands (Chapters 2 and 3) and the UK (Chapter 4), workshops held in Kenya, Madagascar, Fiji and Korea (Chapter 5), and is informed throughout by field visits to the Cook Islands, Fiji, Zanzibar, Madagascar, Mayotte and Comoros.

As such, the work presented here is largely global in perspective and interdisciplinary in method. While rooted firmly within the EBM approach to marine resource conservation and management, and fully cognisant of the need to consider MPAs as linked-social ecological systems, the selection of study sites and subject matter for the individual chapters for this thesis was driven more by practical realities than theoretical considerations. To this end, it reflects the research priorities of NGO practitioners, government officials and resource managers working with coastal communities in the Cook Islands and Western Indian Ocean. It is therefore guided less by explicit frameworks like Ostrom's for evaluating SES systems and more by the maxim "Ostrom's law" that arguably underpins them: that a resource arrangement that works in practice can work in theory (Fennell, 2011).

Initially, the geographical extent of this thesis was constrained to the Western Indian Ocean, a region that has received little attention from LMMA researchers to date (Chapter 5). Chapter 5, which concludes my data chapters, was thus in fact the first to be developed. However, since a study site with a long-established network of tourism-supported initiatives was crucial to addressing the key interdisciplinary research gaps identified in Section 1.7, and because the results of Chapter 5 revealed: i) that the region's LMMAs were considerably younger than those in the Western and Central Pacific; and ii) that incentive-based approaches like tourism that build community interest and help to sustain LMMAs over the long term were underway in WIO, but were largely at an earlier stage than in the Pacific, the remaining data chapters shift geographical focus to the Western and Central Pacific.

The Cook Islands – specifically, the main island of Rarotonga – was selected as the primary field research site because of its long-established network of LMMAs, the tourism-driven nature of its economy and its model of hotel participation in marine protection efforts, which appears to be less resource intensive and more scalable that the efforts that have so far emerged in the WIO, at places like Chumbe Island in Zanzibar and Vamizi Island in Mozambique (Chapters, 2, 3, 5).

To add a further level of geographic complexity, the video transects used in Chapter 4 come from Indonesia and not the Cook Islands. However, since the purpose of this chapter is to explore a bias with the potential to undermine the accuracy of biophysical assessments of marine protected area effectiveness, rather than to specifically evaluate such efforts in the Cook Islands, the location from where the video transects were taken was unimportant. Because video-based methods have yet to be implemented in the Cook Islands, the transects were sourced instead from a well-established video-based monitoring programme in Indonesia.

The overarching objective of this thesis is to better understand how to scale and sustain locally managed marine areas. In achieving this objective, I focus primarily on addressing three specific gaps:

1. The role of research design in improving assessments of LMMAs and other area-based management tools
2. The potential role of tourism as a source of funding and support for LMMAs
3. The status and extent of LMMAs outside the Western and Central Pacific region where the overwhelming majority of the research effort to date has been concentrated

The specific aims are fourfold:

1. To explore how different research designs may lead to different management responses at a long-established network of LMMAs in the Cook Islands, and to discuss the potential of coastal hotels as a scalable mechanism for LMMA funding and support in tropical developing countries

2. To quantitatively evaluate perceptions of marine management efforts in the Cook Islands among the country's largest economic stakeholder: its overseas visitors.
3. To investigate and quantify expectation bias in conservation research and to explore methods for minimising it in future research designs, for example in comparative assessments of LMMAs
4. To trace the evolution and expansion of community management in the WIO and present the first ever inventory and assessment of the region's LMMAs

While the focus of this thesis is LMMAs, it is hoped that the findings, particularly those where the focus is on improving assessments (Chapters 3 and 4), will be of interest to researchers and practitioners working on other spatially explicit management approaches like MPAs.

In the first data chapter, *Chapter 2*, I tackle a key social aspect of tourism-centred LMMAs, using respondent-completion questionnaires to examine perceptions and knowledge of the marine environment amongst visitors to Rarotonga, the Cook Islands' principal visitor hub. Specifically, I i) characterise recreational users of the beaches and the island's lagoon; ii) examine awareness of the *ra'ui*, the island's network of no-take LMMAs; iii) explore support for the *ra'ui* and other marine conservation initiatives; and iv) investigate perceptions of and satisfaction with Rarotonga's marine environment.

Building on the findings of the previous chapter, in *Chapter 3,* I switch from socio-economics to ecology, using the network of LMMAs in Rarotonga as a model to explore how different research designs may lead to different management responses, and to discuss the potential of hotels as a scalable mechanism for LMMA funding and support in tropical developing countries. I conduct three analyses to assess differences in the biomass and abundance of fish species targeted by fishers in LMMA sites and at paired controls. In the first, I use a spatially replicated design to examine the *overall* differences across all of the sites. In the second, I use an essentially unreplicated design to examine the differences between each of the *individual* LMMAs and their controls. In the third, I use an asymmetric design to examine the differences between each LMMA and multiple control sites. I discuss the relative merits and challenges of the two analyses, and identify priority actions to improve marine resource management in the Cook Islands.

I continue to examine the role of research design in improving assessments of area-based management initiatives in *Chapter 4*, with a novel investigation into blind assessment, which plays a vital role in modern medicine to reduce the risk of human expectation affecting findings, but has yet to be adopted in conservation science. Blinding usually refers to keeping study participants, and those involved in assessment, management, or data collection unaware of the allocated treatment or true hypotheses, in order to avoid influence caused by that knowledge.

For this study, postgraduate students in ecology and environmental science unknowingly participated in a double-blind randomised controlled trial during a workshop to measure the effectiveness of MPAs. I tested two hypotheses: i) that non-blinded observers with knowledge of which site is a control and which is a marine reserve would record larger reserve effects in terms of fish abundance than blinded observers; and ii) that this effect would vary with fish abundance, with greater effects registered on transects where fish abundance was higher.

In *Chapter 5*, I explore new horizons for LMMAs, switching geographical focus to the other side of the Indian Ocean from the Western and Central Pacific, presenting the first inventory of LMMAs in the WIO and assessing them in terms of geography, numbers, size and governance structures. I compare the key attributes of these areas to those under government stewardship and evaluate potential contributions to international biodiversity commitments. To determine prospects for future LMMA expansion, I also explore the legal frameworks that underpin locally managed marine initiatives in Madagascar, Kenya, Mozambique and Tanzania. Finally, I make recommendations for improving local marine management, including the establishment of a regional network of practitioners to share best practice on financing and evaluation and to encourage the development of further LMMAs.

The discussion (*Chapter 6*) summarises findings of the preceding data chapters and makes recommendations both for further research and for successful LMMAs. The limitations and implications of the research are discussed and conclusions drawn.

2 The importance of tourist perceptions of marine resource management in Small Island Developing States (SIDS)

2.1 Preface

Tourism in many SIDS is largely dependent on an attractive natural environment of healthy reefs, beaches and other ecosystems, yet visitor perceptions of marine protection and resource management efforts are rarely assessed and incorporated into destination management strategies. In this chapter, I address this paradox from a social science standpoint, using respondent-completion questionnaires to quantitatively evaluate perceptions of marine management efforts in the Cook Islands among the country's principal economic stakeholder: its overseas visitors.

2.2 Abstract

Small Island Developing States (SIDS) are increasingly confronted with the need to balance economic development from tourism with effective management of the marine and coastal resources on which much visitation depends. Visitor perceptions of marine resource management and protection efforts are therefore critical components in maintaining this balance. The Cook Islands is one of the most tourism-dependent economies on earth, relying on its marine life and beaches to sustain a sector with an estimated total contribution to GDP of between 75 and 90%. In this paper, I surveyed 468 visitors to the Cook Islands with respondent completion questionnaires to examine satisfaction with the marine environment and to investigate awareness of and support for marine conservation initiatives in the Cook Islands. Our findings indicate widespread visitor disappointment with the marine environment on the main island of Rarotonga, with around half of respondents not satisfied with coral cover or diversity and around 40% dissatisfied with fish abundance, size, or diversity. Less than half of respondents (45.9%) were aware of the network of no-take marine reserves (the *ra'ui*) in the Cook Islands, and none were able to correctly identify all sites. Support for marine protection efforts was high, with 93.6% rating reserves as important or very important and 85.5% wanting to see at least 30% of Cook Islands waters afforded full no-take protection. Our results provide a baseline against which to evaluate the effectiveness of future management strategies. Based on my findings, I identify priority management actions to address negative perceptions. I also call for a greater role for visitor perception research to help guide sustainable tourism development efforts, both in the Cook Islands and in other tourism-dependent island states.

2.3 Introduction

Tourism is one of the world's largest and fastest growing economic sectors (Fujita et al., 2013). According to the United Nations World Tourism Organisation (UNWTO), international tourist arrivals have shown almost uninterrupted growth over the last six decades, rising from 25 million in 1950 to more than a billion in 2013 (World Tourism Organization, 2014b). Under current projections, this number is expected to increase by 3.3% per year, reaching 1.8 billion by 2030 (World Tourism Organization, 2011). At present, tourism accounts for over 9% of the world's GDP, 1 in 11 jobs and is a major contributor to socio-economic development in many countries (World Tourism Organization, 2014b, 2011).

Tourism is of vital importance to many Small Island Developing States (SIDS), a group of 52 developing countries that share common sustainable development challenges, arising from their small size, remoteness, limited resource base, exposure to global environmental challenges and vulnerability to external shocks (Hall, 2010; Lewis and Roberts, 2010; World Tourism Organization, 2012). In recognition of these challenges, the international community has made special efforts to support SIDS, most notably the 1994 Barbados Plan of Action (BPoA), the 2002 World Summit on Sustainable Development, the 2005 Mauritius strategy for the further implementation of the BPoA, the 2012 Rio 20+ Summit, the 2014 Nassau declaration, and the International Year of Small Island Developing States (also 2014), all of which have called for specific measures to support sustainable development, particularly tourism-centric ones (Hay, 2013; UNEP, 2011; World Tourism Organization, 2014c). The Nassau declaration, for example, states that "The key position of tourism in the economy of SIDS, its growth potential, and relevance to the natural and cultural assets of islands, underline the need to prioritise tourism in global and local strategies for their sustainable development" (World Tourism Organization, 2014c).

Between 2001 and 2011, the number of international tourists visiting SIDS rose by more than 12 million to 41 million, which corresponds to 4% of all global arrivals (World Tourism Organization, 2012). Today, the sector is both a primary contributor to employment and the largest source of foreign exchange for more than half of all SIDS, with tourism receipts accounting for almost a third of their total exports against a global average of just over 5% (UNEP et al., 2012; World Tourism Organization, 2012). Of the top twenty most tourism-dependent countries (measured as tourism receipts as a percentage of GDP), eighteen are SIDS, with 3 in the Pacific, 11 in the Caribbean and one each from the Atlantic, Indian Ocean, Mediterranean and South China

Tourism in SIDS is highly dependent on the quality of the marine and coastal environment, with research suggesting that visitation is principally driven by the appeal of beaches, coral reefs and other unique ecosystems (Fujita et al., 2013; Hampton and Jeyacheya, 2013; Hay, 2013; Lutchman et al., 2005; Uyarra et al., 2005; Vianna et al., 2012; World Tourism Organization, 2012). Given the importance of tourism to the economy of many SIDS and the co-dependence between the quality of marine ecosystems and visitation, visitor perceptions of marine resource management and protection efforts are critical components in the development and prioritisation of effective management polices (Sayan and Karagüzel, 2010; Torres-Sovero et al., 2012). However, few studies have attempted to quantitatively assess such perceptions in tourism-dependent island states (but see Pereira et al., 2003; Leujak and Ormond, 2007 for developing countries, Pendleton et al., 2001; Shivlani et al., 2003; Parnell et al., 2005; Tonge and Moore, 2007; Koutrakis et al., 2011 for developed countries).

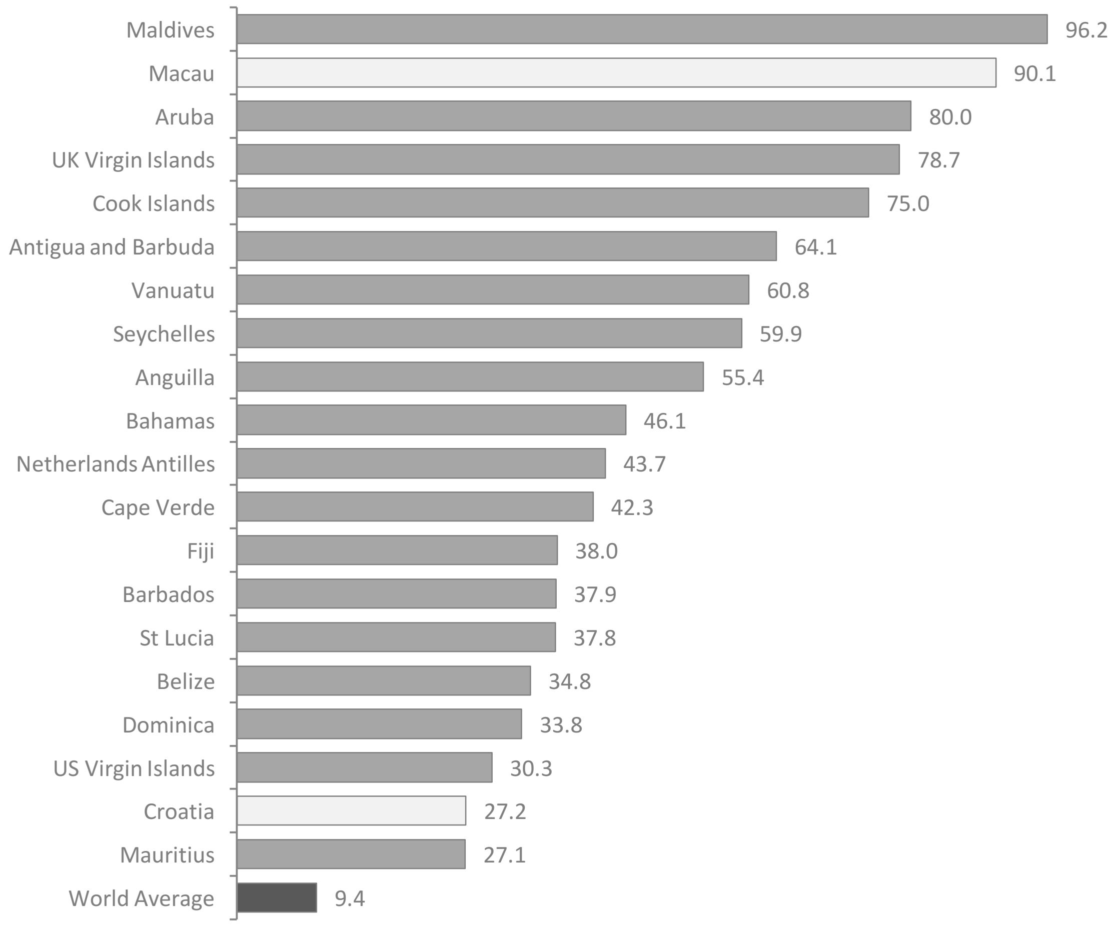

FIGURE 2.1 TOP-TWENTY MOST TOURISM DEPENDENT COUNTRIES AND WORLD AVERAGE (TOURISM RECEIPTS AS A PERCENTAGE OF GDP). 2011-2013 AVERAGE

Bars in mid grey are Small Island Developing States (SIDS).

Source: developed from data from (World Travel & Tourism Council, 2014).

Data are a composite of 3 indicator suites: direct contribution, indirect contribution and induced contribution. Changes to the methodology in 2011 mean that it isn't possible to compare previous years.

Cook Islands figures were obtained from the Cook Islands Director of destination development Metua Vaiimene by personal communication

The Cook Islands is heavily reliant on tourism, and the industry is both a major employer and the main source of government revenue (World Travel & Tourism Council, 2014). In recent decades, marine protection efforts have centred on the use of locally managed no-take marine reserves known as *ra'ui*[1], but this is now set to change following the announcement of a 1.1 million km^2 marine park across the southern half of Cook Islands waters (Koteka, 2012). With stakeholder consultations for the park underway, the time is particularly appropriate to evaluate perceptions of current management efforts from the perspective of the largest economic stakeholder: the visitors themselves.

In this study, I used respondent-completion questionnaires to examine perceptions and knowledge of the marine environment amongst visitors to Rarotonga, the Cook Islands' principal visitor hub. Specifically, I aimed to: i) characterise recreational users of the beaches and the island's lagoon; ii) examine awareness of the *ra'ui*, including permitted and prohibited activities; iii) explore support for the *ra'ui* and other marine conservation initiatives; and iv) investigate perceptions of and satisfaction with Rarotonga's marine environment. Based on my findings, I identify priority management actions to address negative perceptions. I also call for a greater role for visitor perception research to help guide sustainable tourism development efforts, both in the Cook Islands and in other tourism-dependent island states.

[1] *Ra'ui* is both singular and plural. For clarity, I use "a *ra'ui*" to mean a single reserve and "*the ra'ui*" when referring to the network of six reserves on Rarotonga.

2.4 Methods

2.4.1 Study site

The Cook Islands is a group of 15 small islands in the South Pacific that extend eastwards from Samoa towards French Polynesia and lie between 9°S and 23°S and 167°W and 156°W (Figure 2.2). Tourism is by far the most important economic activity and is estimated to contribute 75-90% to GDP (pers. comm. Metua Vaiimene), making the Cook Islands one of the world's most tourism-dependent economies (Tourism in SIDS is highly dependent on the quality of the marine and coastal environment, with research suggesting that visitation is principally driven by the appeal of beaches, coral reefs and other unique ecosystems (Fujita et al., 2013; Hampton and Jeyacheya, 2013; Hay, 2013; Lutchman et al., 2005; Uyarra et al., 2005; Vianna et al., 2012; World Tourism Organization, 2012). Given the importance of tourism to the economy of many SIDS and the co-dependence between the quality of marine ecosystems and visitation, visitor perceptions of marine resource management and protection efforts are critical components in the development and prioritisation of effective management polices (Sayan and Karagüzel, 2010; Torres-Sovero et al., 2012). However, few studies have attempted to quantitatively assess such perceptions in tourism-dependent island states (but see Pereira et al., 2003; Leujak and Ormond, 2007 for developing countries, Pendleton et al., 2001; Shivlani et al., 2003; Parnell et al., 2005; Tonge and Moore, 2007; Koutrakis et al., 2011 for developed countries).). The industry is overwhelmingly centred on Rarotonga, which is the only island with an international airport (Milne et al., 2014; Phillips et al., 2006).

Over the past two decades, international visitor arrivals have more than doubled from 53,569 people in 1993 to an estimated 121,158 in 2013 (Statistics Cook Islands, 2014). Most international arrivals were from New Zealand (65%) and to a lesser extent Australia (19%) in 2013, a trend which is long standing (Statistics Cook Islands, 2014). At 9 international visitors for every local resident, the Cook Islands' carrying capacity – defined as the ratio of visitor numbers to population size (World Tourism Organization, 2012) – was among the world's highest in 2013.

In a 2014 survey of visitors to the Cook Islands by Milne et al. (2014), 54% of respondents indicated that the most appealing element of their visit was the natural environment, which included the pristine waters and unspoilt nature of the islands. This interplay between tourism and environment is recognised by the Cook Islands' tourism master plan, which sets out a vision "to develop tourism that sustains and enhances the well-being of resident Cook Islanders and their environment, society, economy and culture" (Phillips et al., 2006). A particular obstacle to achieving this goal is the state of Rarotonga's marine environment, which has long been cause for concern (Dahl, 1980; Van Pel, 1955). The shallow lagoon and fringing reef that encircle the island are seriously degraded, with contributing factors including subsistence fishing, uncontrolled inflow of land-based pollutants and fertilisers, poorly planned coastal development, crown-of-thorns starfish outbreaks, bleaching and cyclone damage (Cook Islands Ministry Of Marine Resources, 2000; Government of the Cook Islands, 2002; Spalding, 2001; Vieux et al., 2008). These problems have caused a phase shift from a coral- to an algal-dominated reef (Spalding, 2001; Vieux et al., 2008). This change has been reflected in fish community assemblages, with a general decrease in the abundance of corallivores and planktivores, and an increase in herbivores and omnivores between 1999 and 2006 (Vieux et al., 2008).

In the Cook Islands, as in many other parts of the Pacific, ownership of marine resources was traditionally vested in local communities and inherited through patrilineal descent (Evans, 2006). A feature inherent in this was the *ra'ui* system, which banned removal of natural resources in a manner broadly synonymous with contemporary marine protected areas (Tiraa, 2006; Vierros et al., 2010). The practice was phased out in Rarotonga after 1915, but reintroduced in the late 1990s to help improve the marine environment (Lutchman et al., 2005; Miller et al., 2009). At present, there are six *ra'ui* designated in the Rarotonga lagoon and closed to the harvesting of invertebrates and fish. Taken together, the six cover 255.2ha, which equates to approximately 25% of the lagoon flat.

2.4.2 Questionnaire survey

The 21-question survey consisted principally of closed questions and 5-point Likert items that typically ranged from 5 = extremely important/satisfied to 1= not at all important/satisfied. Questions were grouped into five sections.

1) *Socio-demographics*. This section included questions about gender, age, education level, nationality and usual country of residence, and also contained a 5-point attitude rating item to enable visitors to self-assess their knowledge of coral reefs.

2) *Visit attributes.* Five questions assessing the frequency of beach and lagoon use, activities undertaken whilst visiting, frequency of SCUBA diving and snorkeling, competence level for SCUBA and snorkeling, and visitor knowledge of corals.

3) *Visitor awareness of ra'ui system*. Four questions to determine whether and how respondents had heard of the *ra'ui*, could correctly identify which areas of the lagoon were protected and knew which activities were prohibited and which were permitted in these areas.

4) *Visitor attitudes towards marine conservation.* One question to assess respondent perceptions of four aspects of marine conservation via a 5-point Likert item and two questions to determine what respondents felt were appropriate levels of protection for Cook Islands waters.

5) *Visitor perceptions of and satisfaction with natural resource management.* Three questions using 5-point Likert items to assess the level of importance respondents associated with 12 attributes including litter, crowding, coral and fish quality, and their level of satisfaction with each attribute.

Appendices B and C show the full question and answer format used in the questionnaire, together with key attributes of each variable.

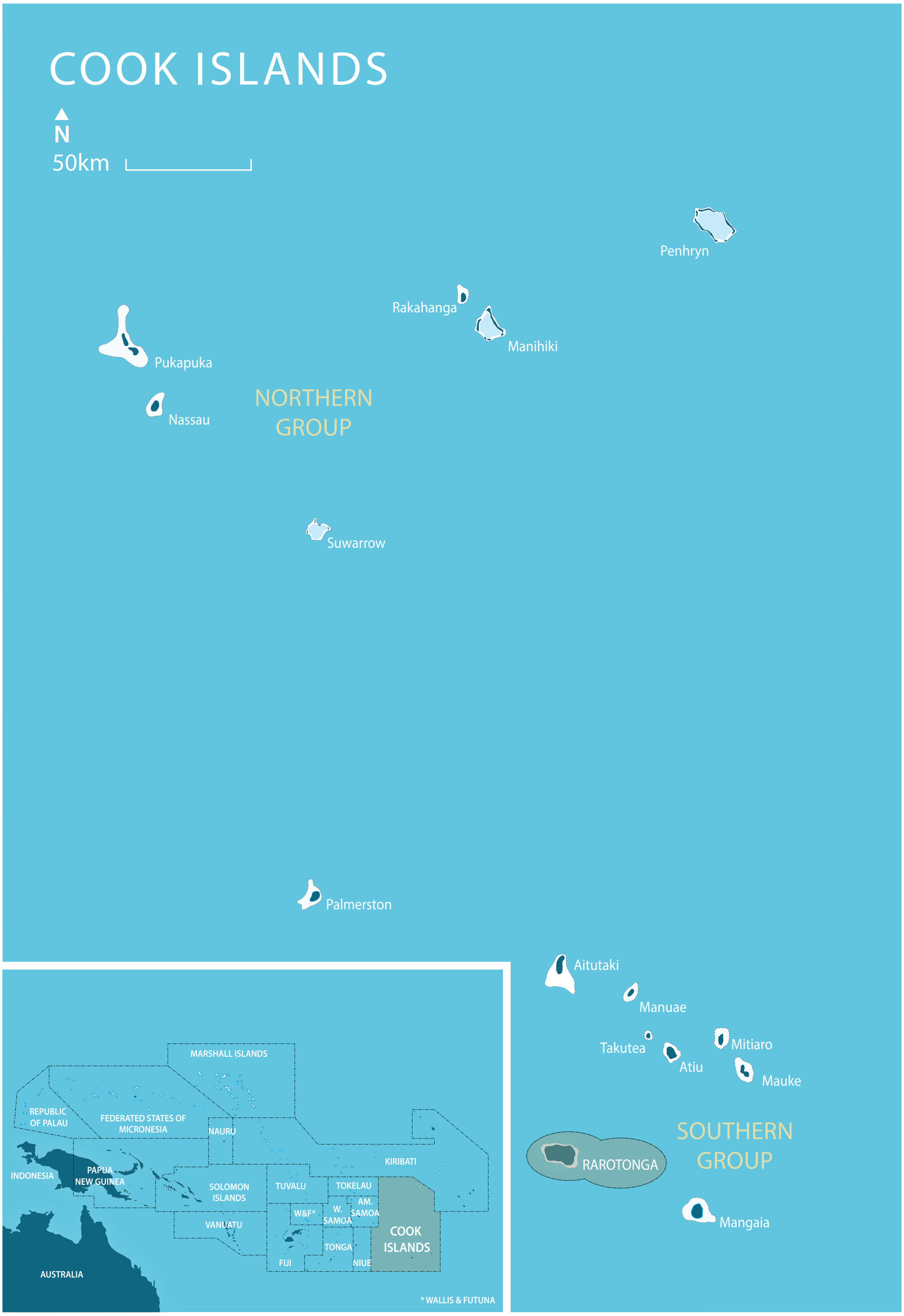

FIGURE 2.2 STUDY SITE LOCATION: COOK ISLANDS, SOUTH PACIFIC

2.4.3 Sampling strategy

The questionnaire was developed in June and July 2013 and refined using a three-step pre-testing and piloting procedure following best practice advice of Dillman et al. (2008). First, an expert panel of UK academics and Cook Islands tourism and resource management professionals (n=7) completed questionnaires and provided feedback. Second, I conducted think aloud cognitive interviews (Tourangeau, 1984) with potential respondents (n=10) in Rarotonga. In both cases, feedback was used to refine fixed response choices, wording, question order, visual design and any navigational issues. Finally, the questionnaire was piloted under field conditions to 30 respondents.

Actual data collection took place in August and September 2013 at Rarotonga International Airport. Intercept surveys with a skip interval were used to select potential respondents: every fifth person who passed the survey station was asked by an interviewer to complete a questionnaire. Our target population was visitors to Rarotonga who had used the beaches or lagoon during their stay (99.4% of all those who were stopped). I distributed 493 surveys in total of which 468 were returned completed, a response rate of 94.9%.

2.4.4 Ethics and representativeness

The questionnaire and associated protocols satisfied The University of York Environment Department Ethics Committee guidelines on ethical conduct.

To assess whether respondents were representative of all visitors, I compared two socio-demographic attributes (age and usual country of residence) of the sample population against actual visitor arrivals from disembarkation cards in 2013, which had been completed by all visitors throughout the year. Information on other attributes such as the gender or education level of Cook Islands visitors was not available. In each case, no significant difference was detected between the two groups (Chi squared goodness of fit test, $X^2 = 7.49$, $p>0.05$ for country and $X^2 = 9.04$, $p>0.05$ age) suggesting that the sample is representative of the visiting population with respect to these characteristics and suffers from negligible seasonal bias.

2.4.5 Data analysis

I used univariate Mann-Whitney U and paired X^2 tests to compare data between groups and generalised linear models (GLMs) to assess multivariate relationships between three response variables (proportion of respondents with high affinity for marine environment, proportion with high satisfaction with marine environment and proportion aware of the *ra'ui*) and multiple predictor variables, using binomial errors and logit link functions. Predictor variables were checked for inter-correlation before being entered into a model. I reduced the models using combined backward- forward stepwise selection, removing or adding variables according to the Akaike Information Criterion (AIC). Finally, I assessed the suitability of the reduced models using residual diagnostic plots and goodness-of-fit metrics via the dispersion parameter. All data analyses were performed in R (version 3.1.1 R Development Core Team, 2014).

Two of the three response variables were formed from the combination of individual Likert items. Marine conservation affinity was calculated by summing individual scores on five importance attributes and dividing them by the total number of statements. The five attributes were "amount of coral", "different types of coral", "number of fish in lagoon", "number of big fish in lagoon" and "different types of big fish in lagoon". Marine conservation satisfaction was calculated in the same way, but for five corresponding satisfaction attributes. I assessed the consistency of these two scales using the McDonald's omega (McDonald, 1999). In both cases, all five statements showed a very high degree of internal consistency ($\Omega = 0.94$ for affinity, $\Omega = 0.96$ for satisfaction). I also assessed the consistency of the awareness scale (a combination of responses to 4 statements about marine conservation (see footnote Table 2.3) using the same method. Here too, internal consistency was excellent ($\Omega = 0.93$). To explore the relationships between affinity and satisfaction and the predictor variables, I collapsed both scores into low and high, dividing the sample into two groups after D'Antonio et al (2012).

To determine priority actions for management, I performed importance-satisfaction analysis and service-quality gap analysis after Tonge & Moore (2007). For the importance-satisfaction analysis, I used the individual means of the 12 importance and satisfaction attributes to generate coordinates for placement in a two dimensional matrix (Figure 2.3). I partitioned the plot into four sections after Aballo et al. (2007), each with different potential management implications.

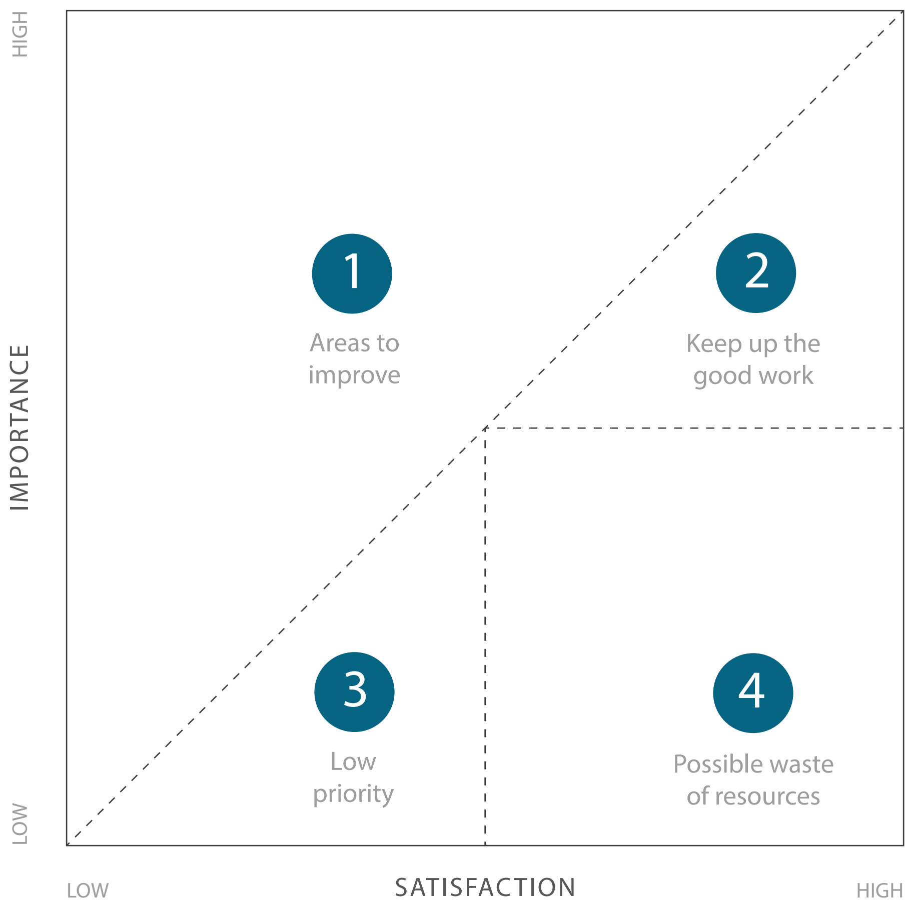

FIGURE 2.3 IMPORTANCE SATISFACTION MATRIX

Source: developed from (Abalo et al., 2007). The boundaries of the partitions were derived from the intersection of the grand means for importance (horizontal dashed line) and satisfaction (vertical dashed line) and a line connecting the points where performance and satisfaction match (diagonal dashed line) (Chen, 2014; Rial et al., 2008; Ryan and Cessford, 2003).

For the service-quality gap analysis, I obtained gap values by subtracting the means of the same 12 importance attributes from the means of the corresponding satisfaction attributes and testing for a statistically significant difference with paired T-tests (Ryan and Sterling, 2001; Tonge and Moore, 2007). Negative gap values indicate that the mean level of satisfaction is lower than the mean level of importance for a given variable and suggest that management intervention may be required. Conversely, positive gap values are due to higher mean satisfaction vs. performance and suggest that no additional management action is needed (Chen, 2014; Tonge and Moore, 2007).

2.5 Results

2.5.1 Sociodemographic characteristics

Of the 468 visitors who completed the questionnaire, 52.1% (244) were female and 47.9% (224) male. Respondents were split evenly across age classes with roughly a fifth in each of the four central categories (26-35, 36-45, 46-55 and 56-65) and the remaining 20% divided between the youngest and oldest categories (Table 2.1). More than 91% of respondents lived in New Zealand and Australia, though 17 different countries were represented in the sample, including the UK (2.4%), Germany (1.3%) and the USA (1.1%). All but 0.4% of the sample had completed secondary schooling and more than half (56.9%) had a university education.

TABLE 2.1 KEY SOCIODEMOGRAPHIC CHARACTERISTICS OF RESPONDENTS (N=468).

Variable	Response category	Total (n)	Total(%)
Gender	Male	224	47.9
	Female	244	52.1
Age	18-25	45	9.6
	26-35	88	18.8
	36-45	96	20.5
	46-55	93	19.9
	56-65	88	18.8
	65+	58	12.4
Education level	Primary school	2	0.4
	Secondary school	200	42.7
	Undergraduate degree	181	38.7
	Postgraduate degree	85	18.2
Usual place of residence	NZ	359	76.7
	Australia	68	14.5
	Europe	30	6.4
	North America	5	1.1
	Other	6	1.3

2.5.2 Profile of visit

Almost three quarters of respondents (72.4%) spent between 4 and 9 days on the beach, with nearly a further fifth (19%) spending more than 9 days there. Just 8.3% visited the beach for less than 3 days of their trip, suggesting that use of the beach and lagoon is a major component of most respondents' stays. Among the sample population, the most popular activities were associated with the shore or with relaxation, rather than with nature. Most respondents visited the lagoon to walk along the shoreline (90.2%), sunbathe (84.6%) or swim (84.0%). The top nature-based activity was snorkelling, which was undertaken by more than three quarters of respondents (75.9%). Other frequently cited activities included photography (62.2%), canoeing (49.1%) and taking a lagoon cruise (30.1%). SCUBA diving was one of the least popular activities among visitors, with less than 1 in 14 (7.1%) taking part.

Although only a small proportion of respondents went SCUBA diving during their stay in Rarotonga, almost a third (32.7%) were qualified to dive. Around half of these divers (18.7% of all respondents) went at least once a year, suggesting that some of the divers who visit Rarotonga may not see it as a diving destination.

2.5.3 Visitor knowledge and awareness of the r*a'ui*

Respondents were asked whether they had heard of the *ra'ui* system of community-run marine reserves and if so, through which communications channels they had heard about them. Just under half of the sample (45.9%) said that they knew about the *ra'ui*. Older visitors were significantly more likely to be aware of the *ra'ui* than younger ones, with 58.6% of over 65s saying that they had heard of the *ra'ui* vs. 44.1% for younger age classes (X^2 = 3.72, DF= 1, p = 0.05).

Ra'ui awareness also differed significantly in the case of beach use, with respondents who spent more days on the beach significantly more likely to express awareness than those who stayed for less time (X^2 = 5.16, DF= 1, p = <0.05, 57.3% for those who spent 9+ days as opposed to 43.3% for shorter stays). These results are supported by the GLMs, which suggest that time spent at or in the lagoon, age and whether a respondent had experience of snorkelling are all significant predictors of overall awareness of the *ra'ui* (Table 2.7).

Of the 215 respondents who were aware of the *ra'ui*, a third (33.2%) had obtained information from brochures and leaflets, and a quarter (23.8%) from signage. Other common information sources were word of mouth (20.6%), guidebooks and hotels (both 12.1%).

Respondents were also asked to identify which beaches were within the boundaries of a *ra'ui* and which activities were permissible within a *ra'ui*. These responses are summarised in Table 2.2. I adopted a similar approach for both questions, inviting respondents to pick from predefined lists of locations and activities and awarding one point per correct answer. Thus, a respondent who knew that Edgewater, Fruits of

Rarotonga, Muri and Rarotongan Hotel were *ra'ui* sites but that Black Rock, Club Raro and Asunto were not, would score a maximum of seven points.

TABLE 2.2 VISITOR AWARENESS OF (A) *RA'UI* BOUNDARIES AND (B) PERMITTED ACTIVITIES IN A *RA'UI* (N=215)

(A) Location	Is a *ra'ui* site?	Correct (%)
Black Rock	No	88.8
Edgewater Hotel	Yes	11.2
Fruits of Rarotonga/Tikioki	Yes	25.6
Muri	Yes	38.1
Rarotongan Hotel/Aroa	Yes	30.2
Club Raro	No	96.3
Asunto	No	98.6
(B) Activity	Is allowed in a *ra'ui*?	Correct (%)
Commercial fishing	No	99.1
Fish feeding	Yes	38.1
Recreational fishing	No	98.1
SCUBA Diving	Yes	40.9
Shell collecting	No	99.1
Snorkelling	Yes	74.4
Spearfishing	No	99.5
Swimming	Yes	72.1
Walking on the reef	Yes	14.0

Although the majority of respondents knew that Club Raro (96.3%), Asunto (98.6%) and Black Rock (88.8%) were not *ra'ui* sites, they were less certain about the sites that were. The most recognised *ra'ui* was at Muri (38.1%) and the least recognised was at the Edgewater hotel (11.2%). Overall, no respondent correctly identified all of the *ra'ui* and just 4.2% managed 6 out of 7 (mean score 3.89, SD=0.84).

Respondents were overwhelmingly aware that extractive activities were prohibited at *ra'ui* sites, correctly stating that commercial fishing (99.1%), recreational fishing (98.1%), shell collecting (99.1%) and spearfishing (99.5%) were not allowed. Most of the sample additionally and erroneously believed that potentially damaging and disruptive activities like fish feeding (61.9%) and walking on the reef (86%) were prohibited, and many thought there was a ban on SCUBA diving (59.1%) though this is also not the case.

2.5.4 Attitudes toward marine reserves and conservation

In this section of the questionnaire, respondents gave their views on 4 statements about marine conservation (Table 2.3), using five-point Likert items that ranged from not at all important to extremely important. They were also asked to state what percentage of Cook Islands waters should be fully protected from fishing and to estimate current levels of no-take protection. Overall, there was overwhelming agreement with each of the four statements with 93.6% of respondents rating marine reserves as important or very important, and 91.2%, 91.9% and 92.3% rating local community involvement, policing and education as important or very important, respectively. There was little variation in the mean (4.41-4.58) or standard deviation (0.91-1.00) for each statement.

TABLE 2.3 MEANS OF RESPONSES FOR VISITOR ATTITUDES TOWARDS MARINE RESERVES (N=468), RATED ON A FIVE-POINT LIKERT ITEM WHERE 1 = NOT AT ALL IMPORTANT AND 5 = EXTREMELY IMPORTANT.

How important do you consider...to be?	Mean	Standard Deviation
Areas of the sea that are protected from fishing (marine reserves)	4.58	0.92
Local community involvement in marine reserves	4.41	1.00
Policing of marine reserves	4.45	0.94
Education about marine reserves	4.49	0.91
Composite attitudinal score [1]	4.48	0.87

1: The composite attitudinal score was calculated by summing individual scores on each statement and dividing them by the total number of statements

For the composite measure, which was calculated by summing individual scores on each statement and dividing them by the total number of statements, there was a statistical difference in scores between people who were aware of the *ra'ui* and those who were not, with those who indicated that they had heard of the *ra'ui* being more likely to agree more strongly with the statements (Mann–Whitney U, W = 22592, p = 0.01). However, there was no significant difference between scores for respondents with diving (M–W U, W = 20474, p = 0.37) or snorkelling experience (M–W U, W = 11652, p = 0.73) and those without.

When asked what percentage of Cook Islands waters was within fully protected marine reserves, 80.1% of respondents did not know, 6% gave the correct answer ("less than 10%") and 14% answered incorrectly. Respondents were also asked what percentage of Cook Islands waters they thought should be fully protected from fishing.

Of those who expressed an opinion (n=400), most (56.3%) believed that more than 50% should be protected, 16.2% said 40-49%, 13.0% said 30-39% and 9.4% said 20-29%. Just 4.7% of respondents said 10-19%, and only 0.4% (2 respondents) believed that less than 10% should be protected.

2.5.5 Importance and satisfaction

In terms of the aspects of the beach and lagoon that they felt were most important, respondents scored water quality ($\bar{x}$=4.64) and litter levels on the beach ($\bar{x}$=4.69) and in the lagoon ($\bar{x}$=4.71) most highly, with a respective 67.5%, 76.2% and 78.6% rating them as extremely important (Table 2.4). Other highly ranked attributes included fish abundance ($\bar{x}$=4.2, 78.8% ranking as very important or extremely important), fish diversity ($\bar{x}$=4.1, 75.1%), coral abundance ($\bar{x}$=3.98, 66.3%) and coral diversity ($\bar{x}$=3.88, 63.6%). The attributes that respondents felt were the least important were the range of activities available ($\bar{x}$=2.92, 38.7% marking not very important or not at all important) and crowding of the beach ($\bar{x}$=3.33, 41.3% rating as somewhat important) and lagoon ($\bar{x}$=3.32, 42.9%).

Whether a respondent had experience of snorkelling and the amount of time they had spent on the beach were significant predictors of marine conservation affinity, a composite of the five importance variables "amount of coral", "different types of coral", "number of fish in lagoon", "number of big fish in lagoon" and "different types of big fish in lagoon" (Table 2.7). However, none of the demographic variables (gender, age and education) had a statistically significant influence.

The aspects of the beach and lagoon that respondents were most satisfied with were the water quality ($\bar{x}$=4.18, 86.4% extremely or very satisfied) and levels of crowding ($\bar{x}$=3.98 for lagoon, 3.97 for beach) and litter ($\bar{x}$=3.99 for lagoon, 3.80 for beach). Respondents were least satisfied with coral diversity ($\bar{x}$=2.53, 57.0% not very or not at all satisfied) and abundance ($\bar{x}$=2.66, 49.8%) as well as fish size ($\bar{x}$=2.56, 55.5%), diversity ($\bar{x}$=2.81, 44.6%) and abundance ($\bar{x}$=2.83, 41.1%).

As with importance, whether a respondent had experience of snorkelling was a significant predictor of overall marine conservation satisfaction, a composite of the five satisfaction variables "amount of coral", "different types of coral", "number of fish in lagoon", "number of big fish in lagoon" and "different types of big fish in lagoon (Table 2.7). As before, none of the demographic factors (gender, age and education) had a statistically significant influence.

TABLE 2.4 FREQUENCIES AND MEANS OF RESPONSES FOR VISITOR PERCEPTIONS OF IMPORTANCE OF BEACH/LAGOON ATTRIBUTES, RATED ON A FIVE-POINT LIKERT ITEM RANGING FROM 1 = NOT AT ALL IMPORTANT TO 5 = EXTREMELY IMPORTANT. $\bar{x}$ DENOTES MEAN, SD DENOTES STANDARD DEVIATION, N DENOTES NUMBER OF RESPONDENTS WHO ANSWERED THE QUESTION. MOST FREQUENT RESPONSES IN BOLD.

Attribute	Important (%)	Neutral (%)	Not important (%)	$\bar{x}$	SD	N
Access to toilet facilities	**50.11**	32.55	17.34	3.57	1.01	467
Availability of a range of activities	25.38	35.89	**38.73**	2.92	1.03	457
Water quality in lagoon	**96.57**	2.36	1.07	4.64	0.58	466
Amount of litter in lagoon	**94.42**	3.43	2.15	4.71	0.67	466
Amount of litter on beach	**94.00**	3.43	2.57	4.69	0.7	467
Number of people on the beach	40.17	**41.25**	18.57	3.33	0.99	463
Number of people in the lagoon	38.74	**42.86**	18.40	3.32	1	462
Amount of coral	**66.30**	26.52	7.17	3.98	0.94	460
Different types of coral	**63.58**	23.71	12.72	3.88	1.05	464
Number of fish in lagoon	**78.76**	15.02	6.22	4.2	0.84	466
Number of big fish in lagoon	**54.11**	31.17	14.72	3.68	1.04	462
Different types of fish in lagoon	**75.05**	17.79	7.16	4.14	0.87	461
Marine conservation affinity [1]	-	-	-	3.98	0.83	466

1: The marine conservation affinity variable was calculated by summing individual scores on five features and dividing them by the total number of statements. The five features were “amount of coral”, “different types of coral”, “number of fish in lagoon”, “number of big fish in lagoon” and “different types of big fish in lagoon”.

TABLE 2.5 FREQUENCIES AND MEANS OF RESPONSES FOR VISITOR PERCEPTIONS OF SATISFACTION WITH BEACH/LAGOON ATTRIBUTES, RATED ON A FIVE-POINT LIKERT ITEM RANGING FROM 1 = NOT AT ALL SATISFIED TO 5 = EXTREMELY SATISFIED. $\bar{x}$ DENOTES MEAN, SD DENOTES STANDARD DEVIATION, N DENOTES NUMBER OF RESPONDENTS WHO ANSWERED THE QUESTION. MOST FREQUENT RESPONSES IN BOLD

Attribute	Satisfied (%)	Neutral (%)	Not satisfied (%)	$\bar{x}$	SD	N [2]
Access to toilet facilities	36.89	**46.64**	16.47	3.24	0.9	431
Availability of a range of activities	**65.65**	30.82	3.53	3.73	0.71	425
Water quality in lagoon	**86.36**	11.47	2.16	4.18	0.72	462
Amount of litter in lagoon	**76.25**	18.74	5.01	3.99	0.87	459
Amount of litter on beach	**67.39**	22.61	10.00	3.8	0.97	460
Number of people on the beach	**78.31**	19.96	1.74	3.97	0.7	461
Number of people in the lagoon	**77.66**	20.39	1.95	3.98	0.72	461
Amount of coral	23.24	27.00	**49.77**	2.66	1.11	426
Different types of coral	19.48	23.47	**57.04**	2.53	1.06	426
Number of fish in lagoon	24.30	34.58	**41.12**	2.83	1.01	428
Number of big fish in lagoon	19.67	24.82	**55.50**	2.56	1.06	427
Different types of fish in lagoon	24.88	30.52	**44.60**	2.81	1.07	426
Marine conservation satisfaction [1]	-	-	-	2.68	0.96	428

1: The marine conservation satisfaction variable was calculated by summing individual scores on five features and dividing them by the total number of statements. The five features were “amount of coral”, “different types of coral”, “number of fish in lagoon”, “number of big fish in lagoon” and “different types of big fish in lagoon”

2: N is lower than for importance scores (Table 2.4) because not all respondents had experienced all of the beach/lagoon features

2.5.6 Importance-satisfaction analysis

The mean importance and satisfaction scores that respondents assigned to the beach/lagoon attributes were used in conjunction with importance-satisfaction analysis and service-quality gap analysis to determine which of the 12 attributes most urgently required management attention.

For the service-quality gap analysis, the mean importance score (I) of each attribute was subtracted from the corresponding satisfaction score (S) to ascertain the gap (G) between the two (Table 2.6). Thus gap values are negative when the mean level of satisfaction is lower than the mean level of importance for a given attribute and positive when mean satisfaction is higher than mean performance. Negative gap values indicate that the attributes in question may require management attention, whereas positive scores suggest that no additional management action is needed (Chen, 2014; Tonge and Moore, 2007). The analysis revealed that 3 of the 12 attributes had positive gaps and 9 had negative gaps (Table 2.6). In particular, the five variables associated with marine fauna (Table 2.6, numbers 8-12) all showed large and significantly negative gaps ($p<0.00001$ in each case, gap = -1.12 to -1.37), suggesting that the primary focus should be on improving these attributes.

TABLE 2.6 IMPORTANCE-SATISFACTION ANALYSIS FOR BEACH/LAGOON ATTRIBUTES. S=SATISFACTION, I=IMPORTANCE, G=GAP

Attribute	S mean	I mean	G (S-I)	Quadrat	Priority
1. Access to toilet facilities	3.24	3.57	-0.33*	3	9
2. Availability of a range of activities	3.73	2.92	0.81*	4	
3. Water quality in lagoon	4.18	4.64	-0.46*	2	8
4. Amount of litter in lagoon	3.99	4.71	-0.72*	2	7
5. Amount of litter on beach	3.8	4.69	-0.89*	2	6
6. Number of people on the beach	3.97	3.33	0.64*	4	
7. Number of people in the lagoon	3.98	3.32	0.66*	4	
8. Amount of coral	2.66	3.98	-1.32*	1	4
9. Different types of coral	2.53	3.88	-1.35*	1	2
10. Number of fish in lagoon	2.83	4.2	-1.37*	1	1
11. Number of big fish in lagoon	2.56	3.68	-1.12*	1	5
12. Different types of fish in lagoon	2.81	4.14	-1.33*	1	3

* Denotes statistical significance (Paired t-test)

This finding is consistent with the results of the importance-satisfaction analysis (Figure 2.4), a plot of the mean importance score of each attribute against the corresponding satisfaction score. Attributes that require priority management attention are those that cluster to the top left of the plot, away from the dashed diagonal line that indicates a perfect agreement between importance and satisfaction. In the case of Figure 2.4, the attributes that most need addressing are the same as in the gap analysis: “number of fish in the lagoon”, “different types of coral”, “different types of fish in the lagoon”, “amount of coral” and “number of big fish in the lagoon”.

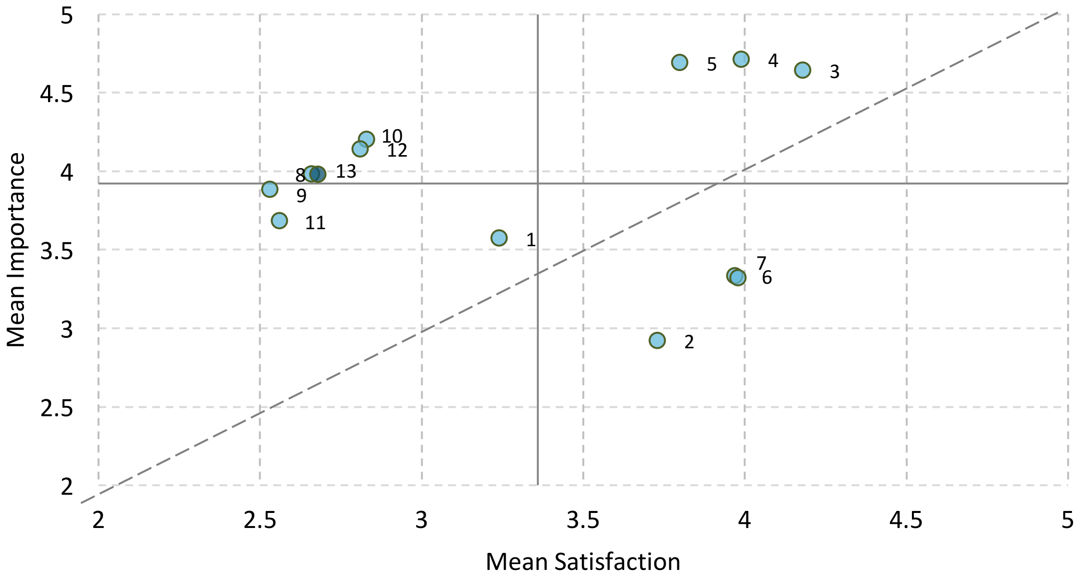

FIGURE 2.4 IMPORTANCE-SATISFACTION PLOT FOR BEACH/LAGOON ATTRIBUTES

Coordinates for dot placement are derived on from the mean scores of each of the 12 importance and satisfaction attributes. Crosshairs derived from grand means of all 12 variables. Data points 1-12 correspond to those in Table 2.6. Point 13 is the grand mean of the five marine fauna attributes (points 8-12). High priority actions (8-12) are those clustered in the top left, distant from the dashed diagonal line that indicates a perfect agreement between importance and satisfaction.

Table 2.7 Results of Generalised Linear Models with binomial errors showing significant predictors of (A) marine conservation affinity, (B) marine conservation satisfaction and (C) *ra'ui* awareness.

Response variable	Model R^2	Significant predictor variable(s)	Estimate	Z value	P value
A) Marine conservation affinity[1]	88.2	Snorkelling	1.32 ± 0.44 SE	2.99	0.003
		Days spent on beach	0.54 ± 0.23 SE	2.36	0.02
B) Marine conservation satisfaction[2]	96.0	Snorkelling	1.41 ± 0.33 SE	4.30	<0.0001
C) *Ra'ui* awareness	95.7	Age	0.24 ± 0.07 SE	3.60	0.0003
		Days spent on beach	0.30 ± 0.11 SE	2.64	0.008
		Snorkelling	0.59 ± 0.29 SE	1.99	0.05

1. Marine conservation affinity is a composite of the five importance variables “amount of coral”, “different types of coral”, “number of fish in lagoon”, “number of big fish in lagoon” and “different types of big fish in lagoon

2. Marine conservation satisfaction is a composite of the five satisfaction variables “amount of coral”, “different types of coral”, “number of fish in lagoon”, “number of big fish in lagoon” and “different types of big fish in lagoon

R^2: percent deviance explained (100x[residual deviance/null deviance])

SE: standard error

Variable set A: marine conservation affinity (response variable), days spent on beach, whether dived, whether snorkelled, reef knowledge, age, education, gender

Variable set B: marine conservation satisfaction (response variable), days spent on beach, whether dived, whether snorkelled, reef knowledge, age, education, gender, affinity

Variable set C: *ra'ui* awareness (response variable), days spent on beach, whether dived, whether snorkelled, reef knowledge, age, education, gender, affinity

2.6 Discussion

The results of this study provide important insights into visitor perceptions of and satisfaction with the marine environment in Rarotonga, as well as into awareness of and support for marine conservation initiatives in the Cook Islands and more broadly. Here, I contextualise these insights and discuss each of them in more detail.

2.6.1 Knowledge of and attitudes towards marine conservation initiatives

Visitor awareness of and support for conservation initiatives are fundamental components of effective environmental management (Leujak and Ormond, 2007; Parnell et al., 2005), particularly in SIDS with a high dependency on tourism. In the survey, there was overwhelming support for the concept of marine reserves, with more than 93% of respondents rating reserves as important or very important, and just 0.4% saying they were unimportant. More than 95% of respondents believed that at least 20% of Cook Islands waters should be covered by no-take marine reserves and the majority (56.3%) felt that more than 50% of the country's waters should be fully protected. This suggests that there is broad support from visitors to increase marine protection in the Cook Islands to a level that would far exceed both current coverage and the levels suggested for the forthcoming Cook Islands Marine Park. Awareness of how much of Cook Islands waters are actually protected was low amongst respondents, though most of the 20% who ventured an answer selected the correct category offered to them: less than 10%. Likewise, visitor awareness of the *ra'ui* system of locally managed marine reserves was low, with less than half of respondents having heard of the *ra'ui*. The age of respondents, whether they had experience of snorkelling and the amount of time they had spent at or in the lagoon were significant predictors of overall awareness of the *ra'ui*, with older respondents who had spent more time at the lagoon displaying greater awareness than their younger counterparts who had spent less time at the lagoon. None of the respondents who said they had heard of the *ra'ui* could correctly identify all of the areas of the lagoon with *ra'ui* designation, and only 20% scored more than 4 out of 7.

When the *ra'ui* were reintroduced in the late 1990s in Rarotonga, they enjoyed broad support from the local community, in part due to an active education programme (Hoffmann, 2002; Miller, 2009). However, awareness-raising activities have declined in recent years such that little public information about the *ra'ui* is currently available and there are no schools activities, beyond the annual two-day Lagoon Day event (pers. comm. Jackie Evans, Ben Ponia). This has led to confusion about the status of the *ra'ui* amongst locals and to an overall drop in support for them (Tiraa, 2006). Because effective enforcement of the *ra'ui* depends on public knowledge and support, it would be prudent to reintroduce an education and awareness programme. Since annual international arrivals to the Cook Islands outnumber residents by a factor of 9 to 1, and given that this study has found evidence of low visitor awareness of the *ra'ui*, I recommend that this programme should target visitors as well as local communities. To this end, the remit of the recently opened Marine Park information hub, which is designed to increase public awareness of and participation in the Cook Islands Marine Park, could be extended to cover the *ra'ui*. This could be complemented by a programme of educational talks in local hotels and schools, as well as information boards at all *ra'ui* sites.

Our research suggests that it would also be sensible to improve the signage that indicates where the *ra'ui* are located. Respondents indicated that signs were their most common source of information about the *ra'ui* after brochures and leaflets, yet several of the *ra'ui* signs around Rarotonga refer to former *ra'ui* areas that are no longer protected (personal observation). This is likely to have caused confusion about *ra'ui* locations amongst questionnaire respondents. Indications are that the annual cost of an awareness programme would be between US$17,000 and US$19,000 (pers. comm. Anna Tiraa, (Govan and Tawake, 2009)). Since the tourism industry benefits from the *ra'ui* (Tiraa, 2006), the costs for such a programme could be met by the sector directly, if government funding is not available.

Although awareness of the existence and correct boundaries of the *ra'ui* was low, respondents were very knowledgeable about the activities that are permitted and prohibited within the boundaries. For example, more than 99% of those who answered the question correctly identified that extractive activities like commercial and recreational fishing, shell collecting and spearfishing were not allowed. However, evidence suggests that visitor awareness of a correct practice may not lead to a non-destructive environmental interaction, and in turn that future educational and interpretive efforts targeting visitors should focus more on the fragility of the lagoon environment and the need to protect it, rather than on regulations governing use (D'Antonio et al., 2012). Further, increasing visitor knowledge of the *ra'ui* may decrease overall satisfaction since higher knowledge may equip them with a greater awareness of the level of degradation in the lagoon (Leujak and Ormond, 2007).

2.6.2 Importance and satisfaction

In terms of their perceptions of the beaches and lagoon, respondents attributed the most importance to water quality and litter levels, and also rated fish and coral abundance and diversity highly. These findings are broadly consistent with other studies. For example, Uyarra et al (2005) reported that clear water was among the most important environmental features determining destination choice in the Caribbean, whilst several authors have found a lack of litter to be similarly influential, in both terrestrial and marine settings (e.g. Watson et al., 1992; Morin et al., 1997; Tonge and Moore, 2007). Similarly, numerous studies have shown that fish abundance and diversity, and coral cover and diversity are highly important to visitors (e.g. Andersson, 1997; Shafer et al., 1998; Williams and Polunin, 2000; Barker, 2003; Edney, 2012; Kirkbride-Smith et al., 2013), especially for those who dive or snorkel, where fish abundance is valued particularly highly (Kirkbride-Smith et al., 2013; Musa et al., 2006; Uyarra et al., 2009; Williams and Polunin, 2000).

Our results show that, overall, 95% of respondents were very satisfied with Rarotonga's beaches. This finding is broadly consistent with the results of an on-going visitor survey conducted by the New Zealand Tourism Research Institute (Milne et al., 2014). The most recent publically available results, which cover the period between 01 January 2014 and 31 March 2014, report satisfaction levels of 96%, with 74% of respondents indicating that they were very satisfied.

Whilst overall satisfaction with the beaches was high in my study, the majority of respondents expressed concern about fish size, diversity and abundance, as well as coral cover and abundance. Generalised linear models revealed that whether a respondent had experience of snorkelling was a significant predictor of satisfaction with these five attributes. However, none of the three socio-demographic variables (gender, age and education) added explanatory power to the model, suggesting that dissatisfaction with these aspects of the visitor experience was widespread rather than restricted to a particular group. The results of the Milne visitor survey highlighted similar concerns, with 9% of respondents considering the natural environment to be the least appealing aspect of their Cook Islands experience (Milne et al., 2014). Comments focussed on excessive litter, lack of environmental stewardship and the poor condition of the lagoon, reefs and marine life (Milne et al., 2014).

However, the proportion of respondents expressing dissatisfaction with these elements is much smaller in the Milne survey than in my study (9% vs. around 50%). There are two potential reasons for this discrepancy. First, this study was concerned with the beaches and lagoon only, so respondents were only asked to indicate the levels of importance and satisfaction they attributed to beach-related aspects. Although my findings suggest that visitors felt coral and fish abundance and diversity were among the most importance components of their beach and lagoon experience, they may well have considered other aspects of their visit such as the quality and cost of public services, accommodation and food and drink to be more important. Second, whilst the Milne survey had a larger sample size than this research (1079 respondents vs. 468), the response rate is considerably lower for the former (23.6% vs. 94.7% for this study). As such there is a significant risk of non-response error: that the 76.4% who didn't respond will differ from those who did (Dillman et al., 2008). In addition, when compared to official visitor statistics from disembarkation cards, the Milne survey is not representative in terms of country of residence (Chi squared goodness of fit test, $X^2 = 20.8$, $p < 0.001$). There are also indications that shorter staying visitors (average stay 9 days vs. 11 from official statistics) and women (60% of respondents) are overrepresented in the Milne sample (Milne et al., 2014; Statistics Cook Islands, 2014).

2.6.3 Priority environmental management actions for the Cook Islands

The results of the importance-satisfaction matrix and gap analysis strongly suggest that management action to enhance visitor experiences of Rarotonga's beaches and lagoons should concentrate on improving the lagoon environment, particularly fish and coral abundance and diversity, and to a lesser extent the abundance of larger fish

Some authors (e.g. Uyarra et al., 2009; D'Antonio et al., 2012) have urged caution in the use of self-rated importance and satisfaction measures as a sole basis for effective environmental management, suggesting that such perceptions should be validated where possible with expert opinion or empirical measurement of conditions. In this study, however, respondent perceptions reflect the scientific reality: the lagoon is seriously degraded (Egerton, 2005; Miller, 2009; Spalding, 2001; Vieux et al., 2008). Several initiatives have been launched to address this issue, most recently a US$14m project to upgrade sanitation systems in 1,000 coastal homes and reduce the pollution load discharged into the lagoon (pers. comm. Jamie Short). To help to address visitor perceptions, this project should be complemented with an education and awareness programme targeting *ra'ui* users (section 4.1), as well as the passing of legislation to strengthen *ra'ui* enforcement capacity. At present, the *ra'ui* have no legal basis, and rely instead on respect for traditional chiefly authority, or *mana* (Miller, 2009; Miller et al., 2011). However, this belief system is waning in influence, particularly in Rarotonga, and no longer delivers effective protection of marine resources (pers. comm. B Ponia). Legislation to give legal recognition to the *ra'ui* has been drafted (Government of the Cook Islands, Draft), but has faced opposition from some community representatives and is to be finalised. This legislation should additionally consider reducing the number of activities permissible in a *ra'ui*. For example, reef walking by visitors is widespread in the Aroa *ra'ui* and even encouraged by the neighbouring hotel, despite evidence that the practice can be highly damaging (Hawkins and Roberts, 1993).

Whilst this study underscores the current need to prioritise environmental management actions that target the lagoon, the future need for such actions is likely to be even greater. In common with many other SIDS, the Cook Islands is developing its tourism product beyond its traditional rest and recreation focus to one that leverages the Islands' natural assets to target eco-aware travellers (Cook islands Government, 2013, 2011; Newport, 2011; Vaiimene, 2012). However, since there is evidence that environmentally conscious visitors place greater emphasis than traditional tourists on the preservation of natural ecosystems (Lewis and Roberts, 2010; Müller and Job, 2009), this strategy may be compromised should current environmental problems with the lagoon fail to be addressed.

2.7 Conclusions

In SIDS, there is a high level of co-dependence between tourism and the natural environment. Without a healthy environment, tourism cannot prosper and may even fail (Mellor, 2003; Newport, 2011). Understanding visitor perceptions of marine resource management and protection efforts is therefore vital to the development of effective environmental management polices and the definition of priorities (Sayan and Karagüzel, 2010; Torres-Sovero et al., 2012). Because many SIDS are particularly vulnerable to impacts that result from global climate change (Hay, 2013), paying greater attention to visitor perceptions of the natural environment can not only help destinations to stay competitive but may even lead to greater receipts from tourism, potentially increasing the available funding and capacity for mitigation initiatives.

The case study presented here reveals widespread visitor disappointment with marine resource health in Rarotonga and low awareness of, yet high support for, marine conservation efforts. To address these issues, I recommend that the Cook Islands strengthens and passes legislation to improve *ra'ui* enforcement and initiates an education and awareness programme to target residents and visitors alike. As in many SIDS, tourism is the engine of the Cook Islands economy and has delivered sustained growth. Careful planning and management will be required to continue this growth into the future, together with a greater role for visitor perception research to help set priorities. As a priority, the current research programme should be expanded to track satisfaction with the marine environment, and focus on increasing the response rate (for example by adopting Dillman's Tailored Design Method (Dillman et al., 2008)) and on the use of modelling to explore how perceptions vary between different visitor segments. For other tourism-dependent SIDS that are less developed than the Cook Islands, emerging methods using social media in place of empirical surveys (Wood et al., 2013) may also offer useful insights into visitor perceptions, but with greater cost-effectiveness.

3 IS THERE A ROLE FOR HOTELS IN SUPPORTING LOCALLY MANAGED MARINE AREAS? A TALE OF THREE ANALYSES

3.1 Preface

Chapter 2 explored an important social dimension of tourism-centred LMMAs, using respondent-completion questionnaires to examine perceptions and knowledge of the marine environment amongst visitors to Rarotonga, the Cook Islands' principal visitor hub. In this chapter, I build on these findings, switching largely from socio-economics to ecology, and using the network of LMMAs in Rarotonga as a model to critically examine the role of research design in improving assessments of LMMAs, as well as the role of tourism as a potential mechanism to support and fund LMMAs.

3.2 Abstract

Co-management, the sharing of responsibility between local communities and governments or other partners, is increasingly recognised as a key strategy for small-scale fisheries management. Across the Indo-Pacific, such approaches are usually termed "locally managed marine areas" (LMMAs). When effective, LMMAs can encourage responsible fishing, strengthen compliance and improve adaptive capacity,

and may help to safeguard food security, address coastal poverty and increase resource abundance. However, evidence that LMMAs can achieve long-term ecological goals is limited. Initiatives often lack the financial support necessary to build capacity and sustain effective management, and the recently established and informal nature of many areas has largely precluded the collection of comprehensive, long-term datasets that can convincingly separate claimed effects of LMMAs from other confounding variables. Here, I use a long-established network of no-take LMMAS in Rarotonga, Cook Islands as a model to explore these financial and evidential gaps. I use data from these sites to conduct three analyses: a network-level analysis to assess *overall* responses to protection across all of the sites, using sites and paired control areas as replicates; a site-level analysis to explore the differences between each of the *individual* LMMAs and their paired controls; and an asymmetric analysis to examine the performance of each LMMA against multiple control sites.

Results from the first analysis suggested no effect of protection at the network level: the abundance and biomass of species targeted by fishers was not significantly higher within LMMAs compared to control sites, and there were no significant differences in overall fish community structure. By contrast, the second analysis revealed significant differences in targeted abundance (2.5-3 times higher), biomass (4.3-5.4 times higher) and community structure between LMMAs and controls at two of the sites (Aroa and Edgewater), although there was no true replication. The third analysis accorded with the second, finding that both targeted biomass and abundance were significantly higher at the same two LMMAs than at multiple controls, thereby increasing the likelihood that the results from the second analysis were evidence of a reserve effect rather than simply a consequence of spatial confounding.

Given the crucial importance of good evidence in effective marine resource management, but also the potential cost of inaction when such evidence is not available, I focus the discussion on the relative merits and challenges of the three analyses, and identify priority actions to improve marine resource management in the Cook Islands. Interestingly, the two LMMAs where significant differences were observed in the second and third analyses had engaged coastal hotels as co-management partners, so I also discuss the potential of hotels as a scalable mechanism for LMMA funding and support in tropical developing countries.

3.3 Introduction

Marine protected areas (MPAs), places which are off-limits to exploitation or excessive use, are a widely advocated tool for marine conservation and fisheries management. Meta-analyses have shown that MPAs, especially permanent no-take reserves (MPAs where all extraction is prohibited) can increase the average biomass, density, size and diversity of species (Halpern, 2003; Lester et al., 2009). Mounting evidence also indicates that spillover of adults (Gell and Roberts, 2002; Russ and Alcala, 2010) and export of larvae beyond protected boundaries can benefit surrounding fisheries (Harrison et al., 2012; Pelc et al., 2010).

Despite their widespread use, MPAs are often of little conservation value, existing only as 'paper parks' without active management or enforcement (Halpern, 2014; Mora et al., 2006). Though negative assessments of individual sites are rare in the literature, global (Balmford et al., 2004; Burke et al., 2011) and regional (e.g. Samoilys and Obura, 2011) evaluations suggest that many MPAs lack adequate management or financing. The most effective MPAs tend to be fully protected, well enforced, established for more than 10 years, larger than 100km^2 and in isolated locations (Edgar et al., 2014). However, creation of large, isolated areas where fishing is prohibited is unlikely to be achievable along populated coastlines, where resource pressure and degradation are often most acute, and where over a billion people depend on seafood as their primary protein source (Gutierrez et al., 2011; Veron et al., 2009). In these contexts, co-managed initiatives may be a preferable approach.

Co-management, where local communities share responsibility for marine resource management with governments or other partners, is receiving increasing attention worldwide (Cinner et al., 2012c). A recent review of 130 initiatives in 44 countries found that co-management could contribute to the successful management of aquatic resources and concluded that it was the only practical solution for most of the world's fisheries (Gutierrez et al., 2011).

Across the Indo-Pacific, areas where marine resources are at least in part under community control are usually termed "locally managed marine areas" (LMMAs) (Govan, 2009; Rocliffe et al., 2014). LMMAs often blend local and scientific knowledge as well as customary and contemporary management systems, with many employing a range of techniques including permanent no-take marine protected areas, periodic closures, gear restrictions, species-specific reserves, access rights and alternative livelihood strategies (Jupiter et al., 2014; Mills et al., 2011).

When effective, LMMAs can improve compliance with regulations by resource users, encourage responsible fishing and enhance adaptive capacity (Gutierrez et al., 2011; Léopold et al., 2013; Levine and Richmond, 2014). They have also been associated with short-term increases in resource abundance (Bartlett et al., 2009; Cinner et al., 2012c, 2005; Pollnac et al., 2001) and have shown some promise as a means to safeguard food security and address coastal poverty (Oliver et al., 2015; Weiant and Aswani, 2006) However, evidence that LMMAs can achieve long-term ecological goals remains scarce and inconclusive (Cohen et al., 2014; Léopold et al., 2013).

There are two principal reasons for this. First, the recently established and informal nature of many initiatives (Chapter 5) has for the most part precluded the collection of comprehensive, long-term datasets that can convincingly separate the effects of LMMAs from other confounding variables. To date, evidence of ecological effects from spatially explicit tools like LMMAs and MPAs has largely been accrued using Control-Impact (CI) designs (Osenberg et al., 2011) in which multiple areas within a single managed site and a single control site are sampled, typically with visual survey techniques like belt transects. Indicators such as fish biomass or density are derived from data collected on each transect and compared at the site level. A statistically significant difference in an indicator between the two sites is taken as evidence of an effect of the managed site (Osenberg et al., 2006).

The potential pitfalls of CI designs are well known (Claudet and Guidetti, 2010; Frid and Crowe, 2015; Underwood, 1992, 1991). A single-point-in-time comparison that finds a significant difference between a single LMMA and a single control site and attributes this difference to the effects of the LMMA will be pseudoreplicated (Hurlbert, 2009, 1984). Though the control and LMMA sites being compared may be similar, they will not be identical, and it is entirely possible that there would have been statistically significant differences between them before the LMMA was established, the so-called area effect (Osenberg et al., 2006). It follows that, without replication across several sites, the effects of the LMMA treatment cannot be logically separated from other sources of spatial variation, leading to potential overestimates of the effects of this management tool (Claudet and Guidetti, 2010; Osenberg et al., 2011).

Given the crucial importance of a defendable evidence base in effective marine resource management, considerable research effort has been devoted to improving assessments of MPAs – and by extension LMMAs – and designs that include temporal replication before and after establishment of a managed area and/or spatial replication across multiple control and managed areas are becoming increasingly common (Claudet and Guidetti, 2010; Heffner et al., 1996).

However, in spite of their potential drawbacks, CI assessments remain appealing for researchers and managers, and can provide sufficient evidence to trigger management action (Frid and Crowe, 2015; Gell and Roberts, 2003). Closely related to classical experiments, CI designs can be analysed with standard techniques like analysis of variance with which most researchers are familiar (Osenberg et al., 2011). Further, because they constitute a single point-in-time evaluation, data can be collected and analysed within a single field season or donor funding cycle, reducing field time and associated costs, as well as the need to apply for multiple rounds of research funding. Studies that use CI designs are common in the early marine protected area literature (e.g. Roberts and Hawkins, 1997) and continue to be published today (e.g. Mumby et al., 2011), albeit with less regularity (Claudet and Guidetti, 2010). Such studies comprise more than 70% of the 89 studies included in the most highly cited synthesis of effects of no-take marine reserves (Halpern, 2003) and also form the ecological underpinnings of a forthcoming interdisciplinary analysis of global MPA performance (Gill and Fox, 2015).

A second reason for the paucity of evidence that LMMAs can achieve long-term ecological goals is that, upon implementation, LMMAs often face the challenges of multiple and potentially conflicting objectives (e.g. enhancing fisheries-dependent livelihoods whilst conserving biodiversity) and frequently lack the leadership, technical knowledge and financial support necessary for effective management and monitoring (Gutierrez et al., 2011; Jupiter et al., 2014; Léopold et al., 2013; Levine and Richmond, 2014).

Since many LMMAs exist in tropical developing countries with high tourist visitation, one potential mechanism for enhancing long-term funding and support might be to engage tourist-sector partners such as coastal hotels. However, despite rising tourist visitation (World Tourism Organization, 2014a) and increasing numbers of protected areas globally (Spalding et al., 2013), hotel involvement in marine resource management remains limited (Stolton et al., 2014, Table 3.5). Efforts to date, usually termed hotel managed marine reserves (HMMRs: Svensson et al., 2009) or entrepreneurial MPAs (Bottema and Bush, 2012; Colwell, 1998; de Groot and Bush,

2010) have tended to be top-down rather than collaborative in nature, as well as costly, complex and time consuming to implement, factors which are likely to have discouraged wider adoption. For example, establishment of the reserve at Chumbe Island necessitated 3 years of negotiations with the government of Zanzibar and cost US$1.2 million, while the development of the Sugud Islands Marine Conservation Area in Malaysia was a four-year endeavour (Riedmiller, 2008a; Teh et al., 2008).

In contrast to these more structured arrangements, hotels in Rarotonga, Cook Islands have assumed responsibility for the management of two of the island's six no-take LMMAs (known locally as *ra'ui*) under informal, rapidly implemented arrangements with no financial outlay, an approach that is yet to receive significant research attention. In the present study, I use this long-established network of LMMAs as a model to explore how different research designs may lead to different management responses, and to discuss the potential of hotels as a scalable mechanism for LMMA funding and support in tropical developing countries. I conduct three biophysical analyses to assess differences in the biomass and abundance of fish species targeted by fishers in LMMA sites and at paired controls. In the first, I use a spatially replicated design to examine the *overall* differences across all of the sites. In the second, I use an essentially unreplicated design to examine the differences between each of the *individual* LMMAs and their controls. In the third, I use an asymmetric design to examine the differences between each LMMA and multiple control sites.

I focus the discussion on the relative merits and challenges of the three analyses, and identify priority actions to improve marine resource management in the Cook Islands.

3.4 Methods

3.4.1 Study site

Rarotonga is the capital and largest of the Cook Islands, a group of 15 small islands that extend eastwards from Samoa towards French Polynesia and lie between 9°S and 23°S and 167°W and 156°W (Figure 3.1). Rarotonga is encircled by a fringing reef enclosing a shallow lagoon with a maximum water depth of 3m and a width from shore to crest that varies from 30m on the north coast to 900m on the southeast coast (Crossland, 1928; Gauss, 1982; Stoddart, 1972). The state of the lagoon has long been cause for concern (Van Pel, 1955) with badly planned coastal development, subsistence fishing, bleaching, cyclone damage, crown-of-thorns starfish outbreaks and an uncontrolled inflow of land-based pollutants and fertilisers all contributing to the degradation (Government of the Cook Islands, 2002; Vieux et al., 2008). As a result, the reef has transitioned to a more algal-dominated state, a shift that has been reflected in fish community assemblages, with a general increase in herbivores and omnivores observed between 1999 and 2006 (Vieux et al., 2008).

In the Cook Islands, as in many other parts of the Pacific, ownership of marine resources was traditionally vested in local communities and inherited through patrilineal descent (Evans, 2006). One of the features of this system was the *ra'ui,* a traditional area set aside for conservation (Vierros et al., 2010). The *ra'ui*, which were established by high chiefs or heads of landowning lineages, banned the harvesting of marine resources to allow stocks to increase (Tiraa, 2006). After being lifted for a harvest period, a *ra'ui* could be established in another area or re-imposed on the same area at a later date (Tiraa, 2006). On Rarotonga, the *ra'ui* fell out of use in the 1960s due to weak enforcement and a lack of community support, but were reintroduced in 1998 to safeguard against declines in fisheries resources and help counteract environmental degradation (Sims, 1990). The revived system enjoyed broad support from the local community, and the five areas that were initially delineated increased to twelve (Tiraa, 2006).

At present, there are six *ra'ui* designated in the Rarotonga lagoon, covering a combined 255.1ha, approximately 25% of the lagoon flat. For this study I focus on five *ra'ui*, since the sixth had a mean water depth of less than 30cm at high tide, making it too shallow to be surveyed for fish populations (Table 3.1). The *ra'ui* are generally closed to the harvesting of fish and invertebrates, but have been opened on occasion, usually for feasts.

FIGURE 3.1 MAP OF THE SOUTHERN COOK ISLANDS AND RAROTONGA, SHOWING LOCATIONS OF FIVE LOCALLY MANAGED MARINE RESERVES (*RA'UI*) AND CONTROL SITES SURVEYED IN THIS STUDY

3.4.2 Research design

In total, I studied 10 sites: five *ra'ui* and five paired controls. To ensure that control sites were as similar to the *ra'ui* areas as possible and to in an attempt to control for habitat variability, I used high-resolution benthic habitat maps (Khaled bin Sultan Living Oceans Foundation, 2013). Sites of the same overall spatial extent with the most comparable broad habitat types (overall percentages of coral bommies, coral framework, pavement, algae ridge, and areas dominated by rubble and sediment) were selected, under the constraint that each control was at least 200m from its paired reserve (Walmsley and White, 2003). I also applied a bootstrapping procedure developed by Bros and Cowell (1987) to fish abundance data from 20 pilot transects to determine an optimum sample size of 16 transects per site (160 transects in total).

TABLE 3.1 CHARACTERISTICS, ENFORCEMENT HISTORY AND MANAGEMENT OF FIVE *RA'UI* IN RAROTONGA. SOURCES: (MILLER ET AL., 2011; KEY INFORMANT INTERVIEWS; RAUMEA AND PONIA, 2010)

Site name	Akapuao	Aroa	Aroko	Edgewater	Tikioki
Area (ha)	101.1	32.5	71.1	5.4	40.2
Date established	May 2000	May 2000	Feb 1998	2008	Feb 1998
Managed by hotel	No	Yes	No	Yes	No
Level of enforcement	Medium	High	Low	High	Low
Guards	No, but staff at neighbouring hotel have seized fisher nets in the past	Yes: hotel staff confront fishers and ask them to leave	No	Yes: hotel staff confront fishers and ask them to leave	No, but frequent visitation by tourist boats makes poaching difficult during daylight hours
Protection history	Opened in 2011, but for unknown duration	Opened for two weeks in 2001 for sea cucumber and trochus harvest and for 3 months in 2003 for trochus. Opened in 2010 for 1 week	Opened 16 Feb 2000 to 2nd March 2000. Three species are allowed to be harvested at all times	Never opened	Opened 1 Feb 2000 for 24 hours. Part of site made a permanent marine sanctuary (*ra'ui* Motukore) in 2000
Issues	Poaching	Fish feeding and reef trampling by tourists	Poaching	Fish feeding and reef trampling by tourists	Fish feeding

3.4.3 Data collection

I sampled each site using haphazardly located 50 x 5 m belt transects and underwater visual census methods to quantify fish and benthic assemblages. After tying off the tape but before starting a count, I waited for five minutes for the community to 'recover' from the disturbance (Dickens et al., 2011). On the first of two passes, I identified and counted all non-cryptic, diurnally active fish to the lowest possible taxon and estimated total length to nearest centimetre. On the second pass, one surveyor made semi-quantitative estimates of structural complexity on a scale of 1–5 where 1 equates to no holes or irregularities in substrate and 5 represents a complex surface with many spaces, nooks, crannies, under-hangs and caves (Pinca et al., 2009). In a previous study, Hawkins et al. (2006) compared this approach to more laborious techniques of estimating cover and found that levels of accuracy were within approximately 5%. A second surveyor recorded the cover of live coral and algae at every 0.5m point intercepted by the 50m-long transect (100 points transect^{-1}) (Stockwell et al., 2009). To avoid the diurnal-nocturnal fish changeover period and to minimise changes in fish activity due to tide, all surveying was carried out between 08:00-16.30 and within 2.5 hours either side of high tide (Becker et al., 2012; English et al., 1997; Miller et al., 2011). At sites where supplementary fish feeding was known to occur (Aroa, Tikioki, Edgewater), I waited until at least 30 minutes after feeding before surveying. Personal observations had shown this to be sufficient to allow fish aggregations caused by the feeding to dissipate.

Before starting data collection, all surveyors were trained to achieve consistency with each other in estimates of fish abundance and length, coral and algae cover and structural complexity using techniques outlined by Bell et al. (1985) and Samoilys and Carlos (2000). During this period, I also conducted 16 semi-structured interviews of key informants with resource users, community leaders, fisheries officers, government officials and tourism operators to establish the protection history and current enforcement practices for each *ra'ui* and to determine which species were targeted by fishers. These interviews were only used to provide background information and are not one of the three analyses referred in the chapter title. Together with their associated protocols, the interviews were approved by the University of York Environment Department Ethics Committee. All research was conducted between 15 June and 06 September 2013, under Cook Islands research permit EX10585.

3.4.4 Data preparation

I converted fish lengths from the underwater surveys to biomass using the allometric length-weight conversion $W = a\ TL^b$, where W is weight in grams, TL is total length in millimetres and a and b are species-specific constants. I obtained the length-weight fitting parameters a and b for each species or closest congener from Fishbase (Froese and Pauly, 2000) and Kulbicki et al. (2005). I also used Fishbase to assign species to one of three broad functional groups (carnivore, omnivore and herbivore), restricting ourselves to these categories because the diets of some species change ontogenetically and with the environment (Aburto-Oropeza et al., 2011). To facilitate comparison with other studies, I converted biomass estimates to kilograms per hectare (kg ha^{-1}) and abundance to density per 250 square metres (num $250m^{-2}$).

3.4.5 Data analysis

Because the focus of the study was on how different analytical approaches may lead to different and potentially conflicting management responses, I conducted three main analyses, one using a spatially replicated design to examine the differences between *ra'ui* and control sites at the network level (herein referred to as the "network-level analysis"), a second using an unreplicated design to assess differences between each individual *ra'ui*-control pairing (the "site-level analysis"), and a third using an asymmetric design to explore the differences between each *ra'ui* and multiple control sites (the "asymmetric analysis").

For the first two investigations, I examined whether there were statistically significant differences between *ra'ui* and controls in regard to: i) community structure; ii) biomass and abundance of targeted species; and iii) trophic level of the entire reef fish assemblage. I first used non-metric multidimensional scaling (nMDS) to investigate differences in fish community structure between each *ra'ui*-control pairing. The nMDS was based on family-level abundance data and calculated using Bray-Curtis dissimilarity measures and multiple random starts to ensure that the global solution was reached (Bray and Curtis, 1957; McCune and Grace, 2002). Because a few families were especially abundant, I performed Wisconsin double standardisations or square root transformations as required, so as to improve detection of differences driven by families with lower abundance (Cohen and Alexander, 2013). I also used permutational multivariate analysis of variance (PERMANOVA) to test whether fish communities were significantly different between *ra'ui* sites and their paired controls (permutations = 999).

Next, I used one-way analysis of variance (ANOVA) to test whether mean density and biomass of targeted species were significantly different between the *ra'ui* and their controls. Prior to testing, I assessed data for normality via inspection of quantile-quantile plots and histograms, and for homogeneity of variance using Fligner-Kileen tests. Since these data revealed non-normality, I applied natural logarithmic ($\log_n$) transformations to the variables biomass and abundance to satisfy assumptions for parametric analysis.

Finally, to explore differences between *ra'ui* and control sites for different trophic groups, I calculated the natural logarithm of the Response Ratio (ln *RR*) for carnivorous, omnivorous and herbivorous species. The response ratio, a common measure of difference in marine protected area studies (e.g. Côté et al., 2001; Halpern, 2003; Samoilys et al., 2007; Lester et al., 2009; Hamilton et al., 2010; Miller et al., 2011) is defined here as the ratio of mean fish biomass in *ra'ui* and control areas. Ratios greater than 0 indicate higher fish biomass within *ra'ui* than in controls; values less than 0 indicate the reverse – i.e. a negative response to protection. Ratios were regarded as significantly different from zero when the 95% confidence interval of the grand mean did not overlap zero. To aid interpretation, I back transformed and converted ratios to a percentage increase or decrease. All analyses were performed in R (version 3.1.1 R Development Core Team, 2014) with the packages vegan and metafoR.

For the third analysis, I examined differences in fish assemblages at four of the *ra'ui* and four unprotected control locations. Given that the habitat matching assessment (Figure 3.2) indicated that the Aroko *ra'ui* and its paired control were significantly less structurally complex and more sediment dominated than the other four locations, it was excluded from this analysis. I compared the abundance and biomass of targeted species at each of the four remaining *ra'ui* (Aroa, Akapuao, Edgewater and Tikioki) to the average of the four remaining controls. Since the intention was therefore to determine whether each single *ra'ui* behaved differently to multiple controls, the design was asymmetric in nature. An in-depth treatment of the potential of such designs to help avoid problems of spatial confounding and to robustly detect differences between impact and control sites is provided by Glasby (1997) and Underwood (1994). Effects of protection on targeted biomass and abundance were tested using a two-factor asymmetric PERMANOVA with the terms LMMA vs. Controls (fixed, orthogonal) and Location (random, nested in LMMA vs. Controls). As such, variation was partitioned into two components: the differences between LMMAs and controls, and the variability between the controls. The analysis was based on Euclidean distances after Curley et al. (2013), with Type III partial sums of squares calculated using 9999 random permutations under a reduced model. It was performed in PRIMER v7 (Clarke and Gorley, 2015), with the PERMANOVA+ package (Anderson et al., 2008).

3.5 Results

3.5.1 Protection history and enforcement practices

The key informant interviews revealed that the Aroa and Edgewater *ra'ui*, which are managed by neighbouring hotels, were the only two sites where regulations were well enforced. Compliance at the other three sites – Akapuao, Aroko and Tikioki – was weak to non-existent, with frequent poaching occurring. Protection history also varied considerably among the five sites. At the time the research took place in 2013, two sites had been established for 15 years, a further two for 13 and one for 5. However, four of the five *ra'ui* had been temporarily opened to fishing since establishment, two of them – Aroa and Akapuao – in the three-year period before this research. Only the Edgewater *ra'ui* had been closed permanently since establishment.

3.5.2 Habitat matching

Using data derived from the benthic mapping, I found no evidence of broad-scale substrate differences between *ra'ui* sites and their controls (Figure 3.2a, X^2 test of association, $p>0.05$ for each *ra'ui*-control pairing). Estimates of structural complexity, which were derived from transect-level observations by the research team, were largely consistent with this finding. I detected no significant differences between *ra'ui* and controls at 3 sites (Aroa, Edgewater and Akapuao, $p>0.05$ for each), although I did find differences at Tikioki ($p=0.007$) and Aroko ($p=0.002$) (Figure 3.2b). Among study locations, both structural complexity and broad-scale substrate composition varied considerably (Figure 3.2a, b). On average, the most structurally complex habitats occurred within the Akapuao *ra'ui* and the Tikioki control (mean of 2.3 out 5 for each), whilst the least complex were observed at the Aroko control site (mean of 1.2).

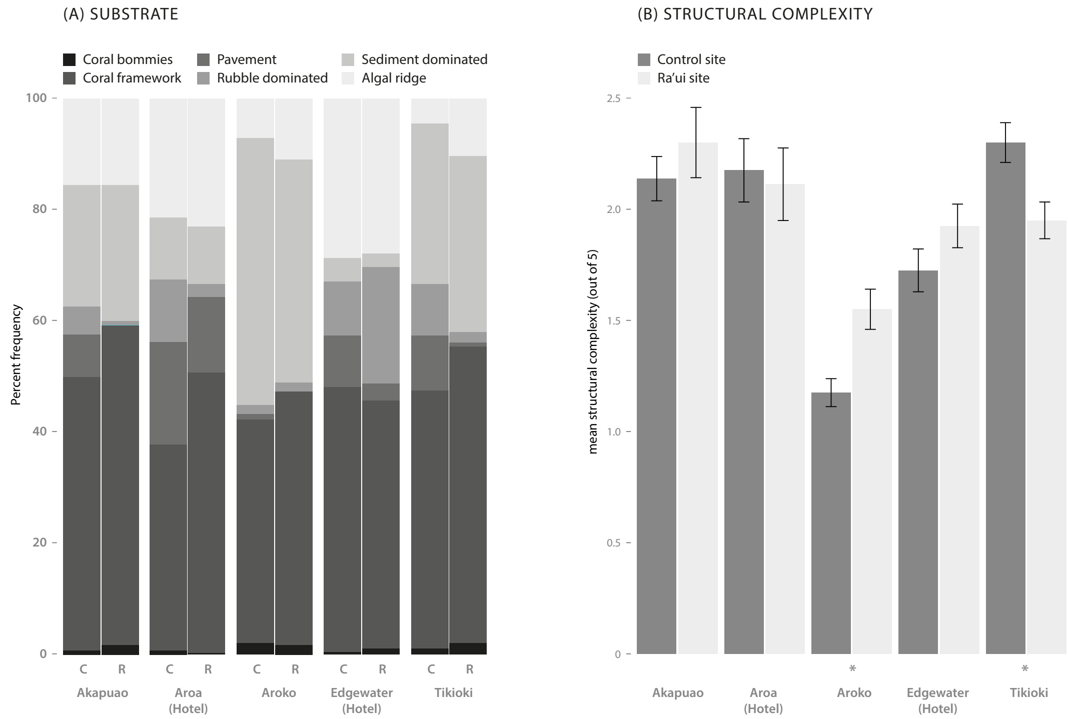

FIGURE 3.2 SIMILARITIES IN HABITAT BETWEEN RESERVE AND CONTROL SITES FOR FIVE *RA'UI* IN RAROTONGA, COOK ISLANDS. (A) PERCENT FREQUENCY OF DIFFERENT CATEGORIES OF SUBSTRATE. (B) MEAN STRUCTURAL COMPLEXITY. * DENOTES A SIGNIFICANT DIFFERENCE BETWEEN A *RA'UI* AND CONTROL SITE. ERROR BARS ARE ± SE.

3.5.3 Network-level analysis: fish community patterns

During this study, I counted a total of 12,903 fishes belonging to 107 species and 22 families. Average richness and abundance differed greatly among the five locations, varying from a total of 66 species and 171.9 fish per 50x5m transect in the Aroa *ra'ui* to 37 species overall and 22.3 fish per transect in the Aroko control site. Across all sites, the three most commonly observed families were Acanthuridae (surgeonfish), Pomacentridae (Damselfish) and Labridae (Wrasse), which together accounted for 80% of total fish abundance.

A nonmetric multidimensional scaling (nMDS) analysis showed no evidence of separation in community structure between *ra'ui* sites and their controls at the family level (Figure 3.3). Two-dimensional stress was low to moderate (0.22), suggesting high goodness-of-fit. PERMANOVA results (F=0.65, p=0.76) confirmed this pattern, indicating that there was no significant difference between *ra'ui* and control communities.

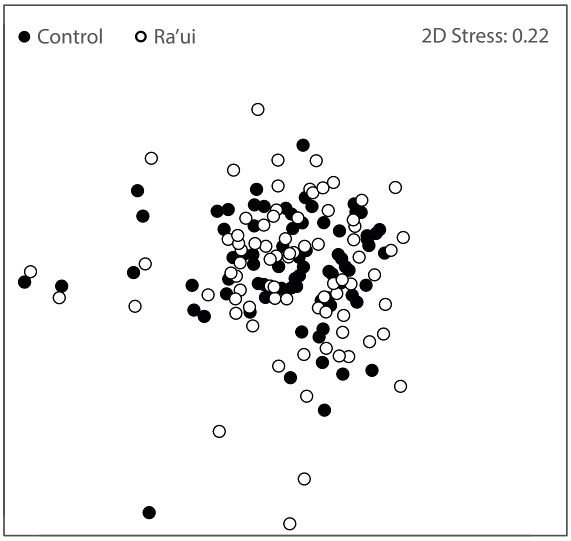

FIGURE 3.3 NONMETRIC MULTIDIMENSIONAL SCALING ANALYSIS OF FISH COMMUNITIES IN FIVE *RA'UI* (WHITE DOTS) AND PAIRED CONTROL AREAS (BLACK DOTS) IN RAROTONGA, COOK ISLANDS.

3.5.4 Network-level analysis: abundance and biomass

Overall, mean abundance of targeted species fluctuated substantially from *ra'ui* to *ra'ui*, from 61.56 (± 20 se) individuals per 250m^2 at Aroa to 6.44 (± 2.92 se) at Aroko (Figure 3.5a). Biomass (a product of density and size) was similarly variable, ranging from 371 (± 130.02) kilograms per hectare at Aroa to 27.58 (± 10.53 se) at Tikioki (Figure 3.5b). Biological responses to *ra'ui* establishment only partly accorded with expected effects. Across all five sites, abundance and biomass of species targeted by fishers was approximately 1.9x higher and 3.1x higher, respectively, within *ra'ui* sites compared to controls. However, the heterogeneity between and within sites was such that the differences were not statistically significant in either case (ANOVA, F=0.38, p=0.56 for abundance, F=0.69, p=0.43 for biomass).

3.5.5 Network-level analysis: differences by trophic group

I used log response ratios (ln *RR*) to explore differences in levels of biomass by trophic group (Carnivore, Omnivore and Herbivore) for the entire reef species assemblage (i.e. not just targeted species). At the network level, differences between *ra'ui* sites and controls were significant only for carnivores, with overall biomass for the network 47.7% higher (Calculated from the exponential of [response ratio-1] x 100; 95% CI 4.1% to 109.6%) than controls. I also found small differences between *ra'ui* sites and controls for omnivores (effect size of 0.17, 95% CI -0.11 to 0.45) and herbivores (-0.08, 95% CI -0.34 to 0.18), but I regarded these as non-significant because the 95% confidence intervals included zero.

3.5.6 Site-level analysis: fish community patterns

The nMDS analysis suggested some differentiation between community structure at *ra'ui* sites and their paired controls at the family level (Figure 3.4). Two-dimensional stress was low to moderate (from 0.14 for Aroko to 0.25 for Aroa) for all sites, suggesting high goodness-of-fit. Differences were greatest at Edgewater and Aroa, the two sites with the strongest levels of enforcement. There was also some evidence of separation between *ra'ui* and control communities at Akapuao, a site with occasional enforcement, while families were less divergent at Tikioki and Aroko. PERMANOVA results confirmed these patterns, with pairwise comparisons showing that *ra'ui* and control communities were significantly different from each other at Edgewater and Aroa only (Table 3.2).

TABLE 3.2 RESULTS FROM PERMUTATIONAL MULTIVARIATE ANALYSIS OF VARIANCE (PERMANOVA) ANALYSIS FOR FISH COMMUNITIES FROM FIVE LOCALLY MANAGED MARINE AREAS (*RA'UI*) VS. THEIR PAIRED CONTROLS

Site name	Pseudo-F statistic	Significance
Aroa (Hotel managed)	2.043	**0.046**
Edgewater (Hotel managed)	4.987	**0.001**
Akapuao	1.818	0.096
Tikioki	1.090	0.344
Aroko	1.286	0.228

Significant p values are shown in bold

3.5.7 Site-level analysis: abundance and biomass

As with the replicated design, *ra'ui* establishment only partly accorded with expected effects. Species targeted by fishers were neither significantly more abundant, nor had significantly greater biomass, within reserve boundaries at Akapuao, Aroko and Tikioki. Only the two hotel-managed sites showed clear differences, with targeted species at Edgewater and Aroa approximately 3 times (by ANOVA, F=26.7, p=<0.001) and 2.5 times (by ANOVA, F=6.4, p=0.018) more abundant within *ra'ui* sites than at paired controls, respectively (Table 3.3). Biomass differences were even larger, with mass (kg) per hectare within *ra'ui* sites exceeding control site figures by approximately 441.8% at Aroa (by ANOVA, F=10.3, p=0.003) and 333.5% (by ANOVA, F=19.2, p=<0.001) at Edgewater.

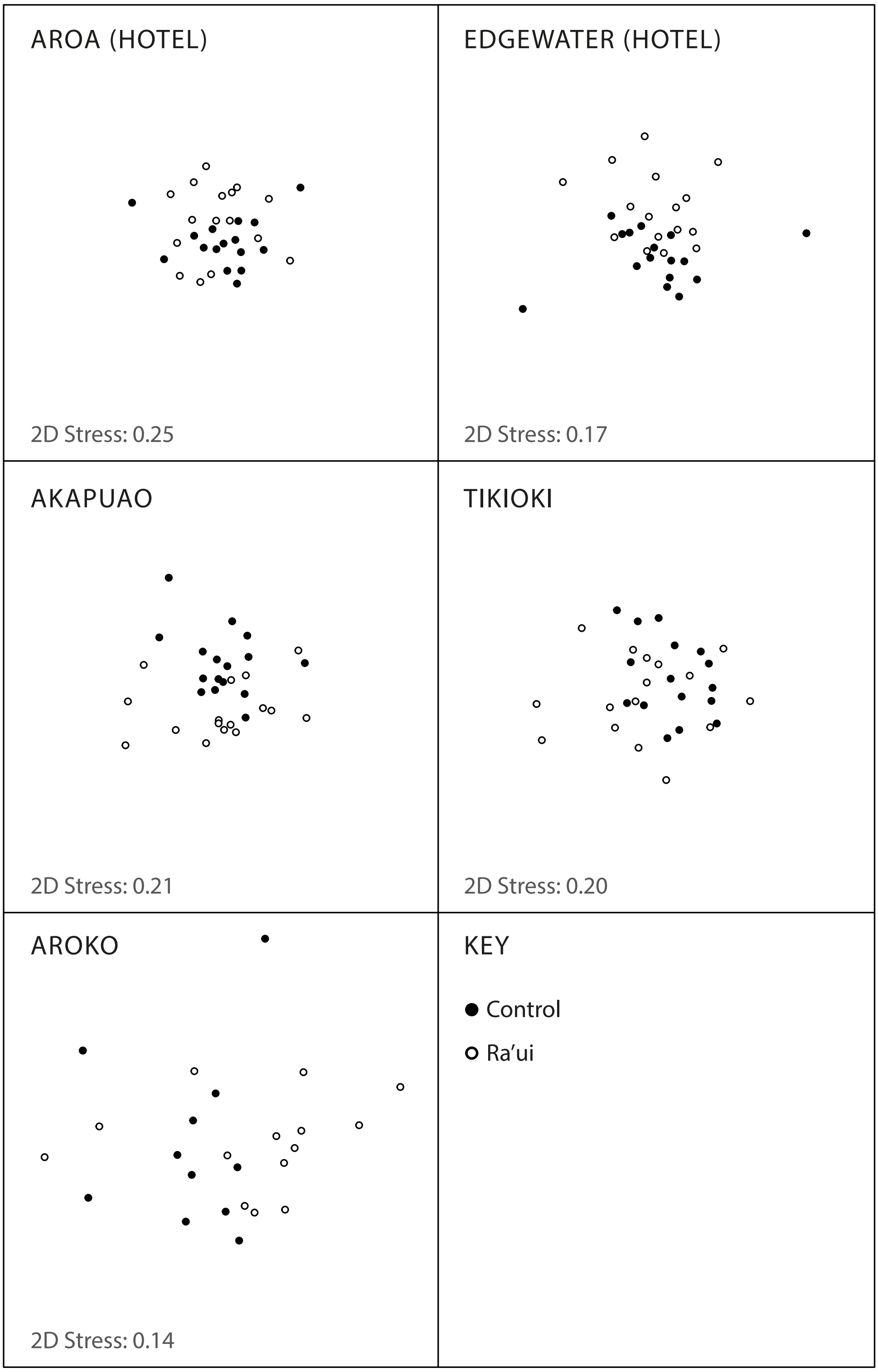

FIGURE 3.4 NONMETRIC MULTIDIMENSIONAL SCALING ANALYSIS OF FISH COMMUNITIES IN *RA'UI* (WHITE DOTS) AND CONTROLS (BLACK DOTS) IN RAROTONGA, COOK ISLANDS.

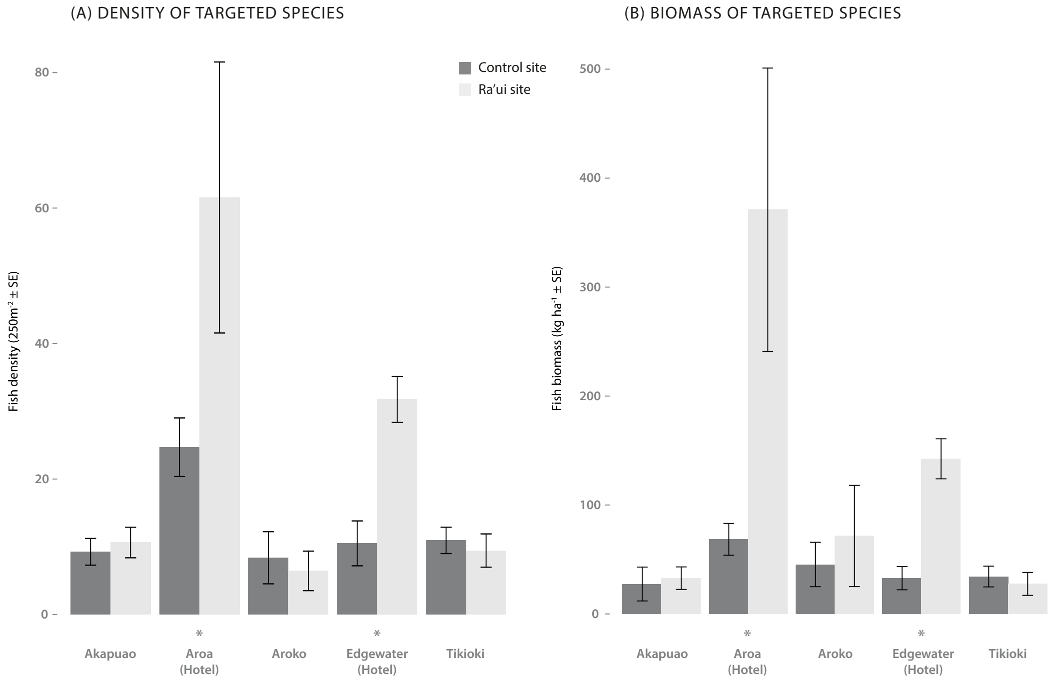

FIGURE 3.5 DIFFERENCES BETWEEN FIVE *RA'UI* (LIGHT GREY BARS) AND THEIR ASSOCIATED CONTROLS (DARK GREY BARS) IN RAROTONGA, COOK ISLANDS. UNTRANSFORMED DATA. (A) DENSITY OF TARGETED FISHES. (B) BIOMASS OF TARGETED FISHES. * DENOTES A SIGNIFICANT DIFFERENCE BETWEEN A *RA'UI* AND CONTROL SITE. ERROR BARS ARE ± SE.

TABLE 3.3 RESULTS OF ONE WAY ANOVAS EXAMINING DIFFERENCES IN DENSITY AND BIOMASS OF TARGETED FISH SPECIES BETWEEN FIVE *RA'UI* AND THEIR PAIRED CONTROLS IN RAROTONGA, COOK ISLANDS

Site name	F statistic	Significance
(A) ABUNDANCE		
Aroa (Hotel managed)	6.286	**0.018**
Edgewater (Hotel managed)	26.660	**<0.001**
Akapuao	0.147	0.704
Aroko	0.383	0.543
Tikioki	0.602	0.444
(B) BIOMASS		
Aroa (Hotel managed)	10.286	**0.003**
Edgewater (Hotel managed)	19.153	**<0.001**
Akapuao	2.550	0.123
Aroko	0.179	0.677
Tikioki	0.725	0.402

Significant p values are shown in bold

3.5.8 Site-level analysis: differences by trophic group

For the trophic level analysis, I found that log response ratios for biomass varied considerably between sites and across trophic levels (Figure 3.6). Of the five study sites, only the two hotel-managed *ra'ui* demonstrated clear differences for carnivores, with biomass for the Edgewater reserve 61.6% higher (95% CI 8.3% to 141.1%) and for Aroa 334.9% higher (95% CI 134.0% to 716.6%) than their respective controls. For omnivores, only the Edgewater was significantly different from its paired control (effect size of 1.36, 95% CI 0.86 to 1.86). I recorded a moderately positive but non-significant difference at Akapuao and negative (i.e. larger biomass in the control than in the *ra'ui*), non-significant differences at Aroko, Tikioki and Aroa. Herbivores showed no significant difference in biomass inside and outside of all sites, except at Aroko, though the large difference I detected there (biomass in the *ra'ui* 357.2% higher than in control, 95% CI 49.2% to 1301.3%) is likely to be an artefact of low fish density.

3.5.9 Asymmetric analysis: abundance and biomass

The results of the asymmetric analysis correspond with those uncovered by the site-level analysis. Abundance and biomass of species targeted by fishers were significantly higher within *ra'ui* boundaries than at multiple controls at both Aroa (F=29.6, p=<0.001 for abundance; F=24.4, p=<0.001 for biomass) and Edgewater (F=19.2, p=<0.001 for abundance; F=19.7, p=<0.001 for biomass), the two hotel-managed sites (Table 3.4). However, no significant or near significant differences were detected for either of the other two sites when compared with the same controls.

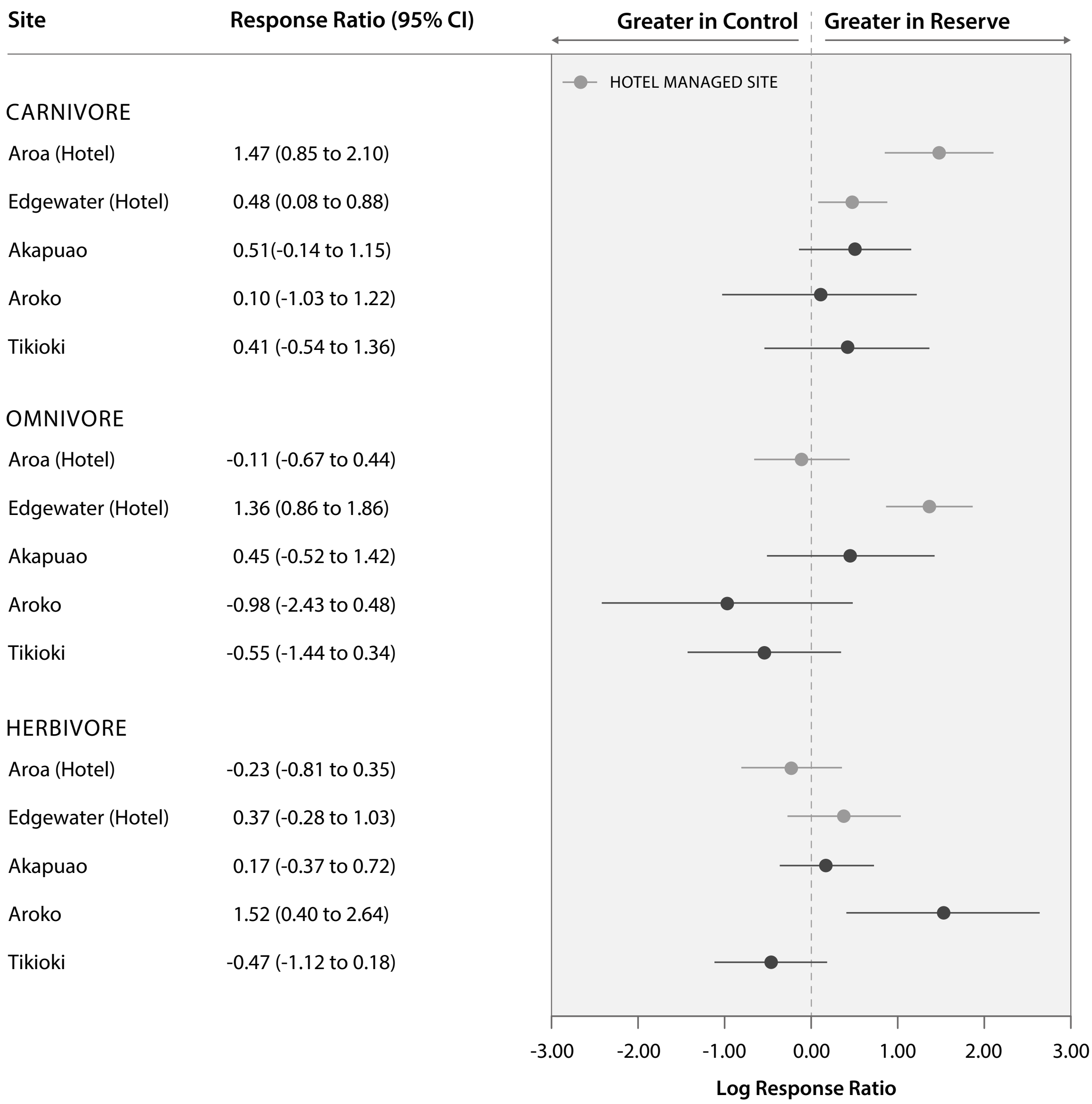

FIGURE 3.6 AVERAGE RESPONSE RATIOS (BIOMASS INSIDE/BIOMASS OUTSIDE) FOR EACH TROPHIC GROUP ACROSS FIVE LOCALLY MANAGED MARINE AREAS (*RA'UI*) IN RAROTONGA, COOK ISLANDS. POSITIVE VALUES INDICATE TROPHIC GROUPS WITH HIGHER BIOMASS INSIDE RESERVES, WHEREAS NEGATIVE VALUES INDICATE TROPHIC GROUPS WITH HIGHER BIOMASS OUTSIDE RESERVES. ERROR BARS ARE 95% CONFIDENCE INTERVALS. A RESPONSE RATIO IS CONSIDERED SIGNIFICANTLY DIFFERENT FROM ZERO IF ITS CONFIDENCE INTERVAL DOES NOT INCLUDE ZERO. RESPONSE RATIOS ARE NATURAL LOG TRANSFORMED.

TABLE 3.4 RESULTS OF ASYMMETRIC PERMANOVAS EXAMINING DIFFERENCES IN DENSITY AND BIOMASS OF TARGETED FISH SPECIES BETWEEN INDIVIDUAL *RA'UI* AND MULTIPLE CONTROLS (N=4 PER *RA'UI*) IN RAROTONGA, COOK ISLANDS

Site name	Pseudo-F statistic	Significance
(A) ABUNDANCE		
Aroa (Hotel managed)	29.622	**<0.001**
Edgewater (Hotel managed)	19.179	**<0.001**
Akapuao	0.503	0.486
Tikioki	1.813	0.178
(B) BIOMASS		
Aroa (Hotel managed)	24.402	**<0.001**
Edgewater (Hotel managed)	19.709	**<0.001**
Akapuao	0.124	0.729
Tikioki	1.826	0.187

Significant p values are shown in bold

3.6 Discussion

3.6.1 Management implications of different analyses

The results presented here come to opposing conclusions, with little indication of an LMMA effect at the network-level, but some evidence from both the site-level and asymmetric analyses that the two hotel-supported sites have been effective at increasing abundance and biomass of target species. Which is the valid result? Is the network-level analysis an under-estimate of biophysical effectiveness, or are the site-level and asymmetric analyses overestimates? The conflicting nature of the findings introduces the danger that management response will differ depending on which analysis receives consideration. A manager relying on the more positive site-level or asymmetric analyses might devote resources to supporting and expanding hotel management around Rarotonga. In contrast, a manager faced with the lack of evidence of an effect provided by the network-level analysis might elect to target resources at finding alternatives to the *ra'ui* system instead, a course of action that would miss the potential role that hotel-managed sites can play. To determine which of these outcomes is likely to be more valid, this discussion considers in detail the merits, challenges and implications of each approach, before going on to identify priority actions for enhancing marine resource management and conservation in the Cook Islands. Given the results of the site-level and asymmetric analyses, I also discuss the potential of coastal hotels as a funding and support mechanism for LMMAs in tropical developing countries.

3.6.2 Network-level analysis: effectiveness of the *ra'ui*

At the network level, this study provides little evidence that Rarotonga's network of no-take LMMAs has had beneficial effects on both species targeted by fishers and on the broader reef fish assemblage. Whilst abundance and biomass of targets species were a respective 1.9x and 3.1x higher within *ra'ui* sites than at controls, these differences were not statistically significant in either case (ANOVA, F=0.38, p=0.56 for abundance, F=0.69, p=0.43 for biomass). Across the network, I found no significant difference in community structure between *ra'ui* sites and their controls at the family level, though I did detect a significant difference for higher tropical level species (effect size of 0.39 for carnivores, 95% CI 0.04 to 0.74).

Evidence from other studies that have investigated the *ra'ui* is similarly mixed. Egerton (2005) found that overall density, overall biomass and biomass of targeted fish species were all significantly higher within the *ra'ui* than at unmanaged sites, whereas Miller (2009) concluded that responses to protection were highly variable for functional groups and species, and that targeted fish species were significantly more abundant at *ra'ui* sites than at corresponding controls for only two of the six sites she surveyed. In addition, Hoffman (2002) argued that the *ra'ui* were an effective conservation management tool and had improved coral reef health, though this observation was based on key informant interviews and an assessment of coral status at one *ra'ui* site only.

Overall, aside from the detection of a significant difference between higher trophic level species (Hamilton et al., 2010; Micheli et al., 2004; Samoilys et al., 2007b), my findings did not accord with the literature, where several studies and meta analyses have shown that marine reserves like the *ra'ui* can lead to statistically significant increases in biomass and abundance (Claudet et al., 2008; Halpern, 2003; Lester et al., 2009; Molloy et al., 2009; Stewart et al., 2009).

The strength of this network-level analysis lies in its spatial replication. Because I compared five *ra'ui* and five controls rather than just a single *ra'ui*-control pair, it is more likely that the overall differences I observed for higher trophic level species were an effect of protection, rather than the result of a pre-existing difference between sites, such as habitat (i.e. an area effect). However, the small sample size (n=5), high heterogeneity of the data and non-random siting of both *ra'ui* and control sites precludes this latter possibility from being disregarded altogether (Quinn and Keough, 2002). Two other factors may also make it challenging to observe a difference between *ra'ui* and controls should one exist.

First, when the *ra'ui* were reintroduced in the late 1990s in Rarotonga, they enjoyed broad support from the local community, in part due to an active education programme (Hoffmann, 2002; Miller, 2009). However, the semi-structured interviews in this study revealed awareness-raising activities have declined in recent years to such an extent that little public information about the *ra'ui* is currently available and there are no schools activities, beyond the annual 2-day Lagoon Day event. This has led to confusion about the status of the *ra'ui* amongst locals and to an overall drop in support for them (Tiraa, 2006).

Secondly, for much of the last 20 years, the most frequent occurrences of ciguatera fish poisoning (CFP) anywhere in the world have been recorded in Rarotonga (Rongo and van Woesik, 2013). For the effectiveness of the *ra'ui*, the consequences have been twofold. First, whilst the lagoon remains an importance source of fish for subsistence purposes, CFP has reduced the number of species that can be safely harvested and consumed (Pinca et al., 2009). Several authors (Côté et al., 2001; Edgar and Barrett, 2012; Jennings et al., 1995; e.g. Roberts and Polunin, 1991) have suggested that low fishing pressure in non-reserve areas may explain similarities in abundance and biomass between reserves and control sites. Secondly, the species targeted by fishers have varied in response to incidences of CFP. Because Rarotonga is a small island of less than 10,000 inhabitants, cases of poisonings, the species involved and the location of the catch spread rapidly through informal social networks, causing certain species in certain locations to be avoided at times (Rongo and van Woesik, 2013; Statistics Cook Islands, 2015) and making it more challenging to disentangle the effects of protection from the consequences of CFP.

3.6.3 Site-level analysis: effectiveness of the ra'ui

Of the five sites I studied, only the two hotel-managed reserves at Aroa and Edgewater were statistically different from their controls. Abundance and biomass of species targeted by fishers were 2.5 and 5.4 times higher within the *ra'ui* than at the control site at Aroa, and 3.0 times and 4.3 times higher at Edgewater. At both locations, the fish community structure and the responses of higher trophic level species were significantly different from controls. In contrast, the other three sites –Akapuao, Aroko and Tikioki – showed no clear differences in targeted abundance and biomass, community structure or trophic groups.

The key shortcoming of this particular analysis is its lack of true replication. If we want to determine whether a difference exists between the Aroa *ra'ui* and its control, for example, then it is valid to statistically compare biomass and abundance as I have done here. However, if we want to go one stage further and attribute the differences in the two indicators to the effect of the LMMA, then our analysis becomes pseudoreplicated (Hurlbert, 1984). This is because the experimental unit is in reality the *ra'ui*-control pair, not the 16 transects sampled at each of the two sites, and since there is only a one Aroa *ra'ui* and one Aroa control, the sample size is not 16, but 1. Without proper replication, it becomes impossible to statistically separate a treatment effect from a location effect, reducing the scope of inference and potentially invalidating the conclusions that result (Heffner et al., 1996).

This does not mean that the site-level investigation is without scientific merit, however. Many published studies have focused on a single watershed, lake or protected area, yet achieved useful inferences, principally through careful but subjective assessment of other possible confounding factors (Hurlbert, 2010, 2009). For this analysis, there are two reasons to suppose that the differences between *ra'ui* sites and controls could be an effect of the *ra'ui* rather than a consequence of pre-existing differences.

First, I used broad-scale benthic habitat mapping and transect-level observations of structural complexity to control for habitat and to ensure *ra'ui* and control areas were as similar as possible. It has long been recognised that habitat complexity and heterogeneity can be a primary driver of between-site differences (Chapman and Kramer, 1999; Edgar and Barrett, 1997; García-Charton et al., 2004; Garcia-Charton and Ruzafa, 1999), but few studies quantitatively control for habitat, in spite of increasing calls to do so (Claudet and Guidetti, 2010; Côté et al., 2001; Miller and Russ, 2014; Osenberg et al., 2011). By partitioning the natural variability due to habitat from the variability due to the protection afforded by the *ra'ui*, I increase the likelihood that the observed differences between *ra'ui* and control sites are attributable to protection rather than habitat. Indeed, Aroa and Edgewater were not statistically significantly different with respect to benthic habitat.

Secondly, at the two sites where biomass and density were significantly different from their controls (Aroa and Edgewater), the magnitude of this difference was large. In terms of density, for example, few studies of reef fish assemblages in marine protected areas have demonstrated an increase of more than 100% (Lester et al., 2009; Willis et al., 2003). However, for Aroa and Edgewater, density was a respective 2.5 times and 3 times higher than at controls. Biomass differences were even larger, exceeding control figures by 5.4 times at Aroa and 4.3 times at Edgewater. In the early ecological literature, there are several examples of studies that persuade not with the rigour of their experimental design, but with the magnitude of the effects they demonstrate (Raffaelli and Friedlander, 2012). For example, in a series of experiments, Paine (1974) found that removal of the predatory starfish *Pisaster* led to a rapid colonisation of the shore by mussels, causing numerous secondary extinctions (Raffaelli and Moller, 2000). Although the majority of Paine's experiments had no controls or replication, and would therefore be unlikely to meet the rigorous standards of peer review today, the magnitude of the effect was such that it is doubtful that anybody would question that the cause lay with the loss of the keystone consumer, the starfish (Raffaelli and Moller, 2000).

3.6.4 Asymmetric analysis: effectiveness of the *ra'ui*

When combined with the habitat matching, the magnitude of the differences observed between the two hotel-supported *ra'ui* and controls in the site-level analysis appears to suggest a reserve effect at these two sites. However, since the underlying analytical design lacked true replication, this interpretation, however compelling, is not logically defensible in the strictest sense, and more evidence is required to unambiguously discount the possibility that the observed findings were due to the effects of area, rather than protection.

Such evidence is provided by the asymmetric analysis, which also detected significant differences in both the biomass and abundance of target species between *ra'ui* and controls at Aroa and Edgewater, but not at two sites where there was no hotel involvement. The analysis compared each *ra'ui* to four control sites, partitioning the variability due to protection measures from that due to habitat/area, and so helping to avoid the issues of spatial confounding that undermine the site-level investigation (Claudet and Guidetti, 2010; Glasby, 1997; Underwood, 1994). It therefore improves upon the site-specific analysis from both a quantitative and logical perspective.

The asymmetrical analyses used here is based on the multiple-control beyond BACI (Before-After-Control-Impact) designs described by Underwood (1994, 1992, 1991), but modified to account for the lack of 'before' data (Glasby, 1997). As such, it shares the same primary limitation of both its parent and of the network-level analysis: it assumes random sampling of LMMAs and controls (Stewart-Oaten and Bence, 2001). This assumption has come in for some scrutiny, since sites cannot be selected at random for a pool of all possible sites and assigned into treatment (i.e. LMMAs) and controls, as would be the case with a classic experiment (Chapter 4, Osenberg et al., 2011).

Nonetheless, the asymmetric analysis helps to overcome the shortcomings of the site-level investigation by introducing spatial replication, and better accounts for the variability that made it challenging to detect an effect in the network-level analysis. As such, it provides the clearest and most robust indication both of a reserve effect at the hotel-supported sites, as well as of a lack of effect at the other LMMAs.

I further suggest that the reserve effect highlighted by the analysis is likely to be due to enforcement, revealed by the key informant interviews to be effective only at Aroa and Edgewater. Such an interpretation finds broad support in the literature: though few studies of marine protected areas present data on enforcement and compliance (Campbell et al., 2012; Samoilys et al., 2007b; but see Walmsley and White, 2003), there is strong evidence that reserves only work when fishing is effectively prevented (Claudet and Guidetti, 2010; Daw et al., 2011; Guidetti et al., 2008).

However, whilst an enforcement regime is in place at Aroa and Edgewater, it lacks legal support, and compliance remains an issue. Although staff at both hotels confront people fishing in *ra'ui* areas and ask them to leave, they have no legal powers to do so. This has led to unpleasant confrontations in the past, with fishers refusing to leave and staff unable to eject them. At present, the *ra'ui* are without legal foundation and rely instead on respect for traditional chiefly authority, or *mana* (Miller et al., 2011). Yet this underlying belief system is being eroded in the Cook Islands and is considered no longer sufficient to ensure adequate protection of marine resources in Rarotonga (pers. comm. B Ponia). In the semi-structured interviews, hotel staff members said that incidences of poaching were increasing but reported feeling reluctant to challenge trespassers for fear of abuse.

A further issue relates to the activities that the hotels themselves encourage in the *ra'ui*. For example, supplementary fish feeding at both sites is widespread (although not at levels likely to generate the apparent effects observed here), despite uncertainty over its impact (Feitosa et al., 2012; Hémery and McClanahan, 2007; Ilarri et al., 2008; Milazzo et al., 2006, 2005). Reef walking is similarly popular, in spite of evidence that the practice can be highly damaging (Hawkins and Roberts, 1993).

3.6.5 Priority management actions for the Cook Islands

Detecting changes in marine assemblages caused by human activities such as the creation of LMMAs requires that changes resulting from such activities can be distinguished from natural sources of variation (Curley et al., 2013). BACI designs, which attempt to deal with both temporal and spatial variation by sampling Control and Impact sites both Before and After LMMA establishment, generally provide the most reliable measures of reserve effects and are therefore considered the ideal sampling design for assessing no-take areas (Osenberg et al., 2011, 2006)

Here, as with many LMMAs, robust baseline 'before' data were not available, so I relied instead on progressively deconstructed approaches with replicated and multiple controls (Claudet and Guidetti, 2010; Osenberg et al., 2006). Other marine researchers have also adopted asymmetric designs (E.g. Airoldi et al., 2015; Curley et al., 2013; Fraschetti et al., 2012; Glasby, 1997; Guidetti et al., 2005; Harasti et al., 2014; Hoskin et al., 2011; Lincoln-Smith et al., 2006; Olds et al., 2012), but their use is not widespread in the literature, perhaps because asymmetric analyses can be complex to construct and often require manual calculation (Glasby, 1997). However, given their benefits, and since automated and straightforward techniques such as PERMANOVA can account for asymmetric designs and, when based on Euclidean distance matrices, give equivalent results to classical ANOVA, there is no reason that such designs should not receive greater attention in future (Airoldi et al., 2015; Curley et al., 2013).

Irrespective of the analysis considered, it is clear that the *ra'ui* system has not been an unqualified ecological success. I suggest that the myriad management issues presented here might be best addressed through the framework of the forthcoming Cook Islands Marine Park. Announced in August 2012, the 1.1 million km^2 protected area will cover the southern half of Cook Islands waters and will be zoned for multiple use, including fishing, tourism and even deep-sea mining of manganese nodules, providing such uses can be deemed to be sustainable (Chapman-Smith, 2014; Koteka, 2012). The park has a nascent education programme and recently opened an "information hub" in

Rarotonga to increase public awareness and encourage participation in the planning process. To improve compliance and reduce poaching, the remit of this centre could be extended to cover the *ra'ui* and could be complemented by a programme of educational talks in schools (Chapter 2) and a long-term research programme to better understand the effectiveness of the *ra'ui* system. In addition, the park's legislative mandate is currently being developed in consultation with stakeholders including traditional leaders, NGOs, government agencies and industry associations. This mandate could also give legal recognition to the *ra'ui,* regulate fish feeding and reef walking and provide a framework for strengthening enforcement capacity (Chapter 2). Such a step would be timely, given that ciguatera cases have been steadily declining since peaking in 2004, increasing per-capita fish consumption and intensifying fishing pressure in the lagoon (Pinca et al., 2009; Rongo and van Woesik, 2013, 2011).

3.6.6 Scalability of hotel management model to other locations

Beyond this study, there is some evidence that hotels can act as valuable stewards of marine resources (Table 3.5). For example, a recent survey of fish populations at the Whale Island Bay Resort in Vietnam concluded that the hotel-managed reserve there was as effective as comparable reserves managed by governments or collobaratively (Svensson et al., 2009). Density was 2.9 times higher than control sites and the number of species was 2.6 times higher, findings that compared favourably with similarly sized reserves at Sumilon Island (1.8x density; 1.2x species) and Apo Island Reserve (1.4x density; 1.15x species) (Svensson et al., 2009).

TABLE 3.5 EXAMPLES OF HOTEL INVOLVEMENT IN MARINE RESOURCE MANAGEMENT

Site	Country	Characteristics	Year	Size (ha)	Annual cost per ha ($US)	Reference
Navini Island	Fiji	12-month renewable lease agreement between Navini Island Resort and the chiefly clan of the Tui Lawa, but not formally recognised under government legislation. In addition to lease fees, resort provides materials to local primary school and supports community development projects	1988	25	97	(Niesten et al., 2013)
Anse Chastenet	Saint Lucia	No formal lease. Hotel closed off 2.6ha area of larger 'paper park'. Fishers using the park were asked to leave by hotel staff. Subsumed into larger, government-run Soufrière Marine Management Area in 1995	1992	2.6	-	(Roberts and Hawkins, 1997, 2000)
Chumbe Island Coral Park	Tanzania	10-year renewable lease between hotel and Zanzibar Government. Officially gazetted as an IUCN Category II protected area. Fishers are employed as park rangers and the hotel runs environmental education programmes for local communities and schools	1994	33	3,600	(Riedmiller, 2008; Nordlund et al., 2013)
Sugud Islands Marine Conservation Area	Malaysia	30-year lease. Officially gazetted as an IUCN Category II protected area. Reserve is managed a not-for-profit organization funded in part by a fee levied on visitors to Lankayan Island Dive Resort. Fees are used monitor and enforce the reserve, train personnel, and undertake conservation programs	2001	46,700	3	(Teh et al., 2008)
Whale Island Resort	Vietnam	Renewable 10-year lease agreement between local government and hotel. Two no-take reserves that are patrolled at night to prevent poaching. Enforcement support from coastguard, who deliver verbal warnings or confiscate fishing gear	2001	16	625	(Svensson et al., 2009)
Wakatobi Diver Resort	Indonesia	200 ha no-take zone and 500 ha buffer zone leased from local chiefs. Community representatives are responsible for enforcement. Proceeds from the lease are contingent upon community compliance with the regulations and are used to improve local infrastructure	2002	200	1,250	(Svensson, 2009; Svensson et al., 2010)
Misool Eco Resort	Indonesia	25-year lease agreement between local community and resort made under both Papuan *adat* (customary) law and Indonesian law. Resort acts as steward for the area and provides community members with employment, health insurance, job training and English lessons	2005	42,500	-	(Heinrichs, 2008; Niesten and Gjertsen, 2010)
Vamizi Island Marine Sanctuary	Mozambique	50-year lease, funded by conservation fees levied on guests who stay at the island's lodge. Reserve co-managed by the lodge and community leaders involved in fisheries, who receive US$3 for each dive boat that enters.	2008	1800	-	(Garnier et al., 2008; Rocliffe, 2010; Rocliffe et al., 2014)

However, since most hotel-managed approaches to date have been initiated by private organisations answerable to their investors, there is a danger that profit may be placed ahead of environmental and social concerns (Bottema and Bush, 2012; de Groot and Bush, 2010). An example of this is “greenwashing” – or perhaps more accurately, “blue-rinsing” – whereby a hotel presents an exaggerated picture of its environmental sustainability to the public (Weaver and Lawton, 2007). For instance, researchers at the hotel-managed Sandy Bay Reserve in Roatan, Honduras were ordered not to study the impact of the hotel’s dredging activity within the reserve, which may have dramatically increased fine sediment load on surrounding reefs (Colwell, 1998). A further criticism of HMMRs is that they are expensive and isolated undertakings and do not represent an integrated approach to coastal management (Colwell, 1998; de Groot and Bush, 2010). Typically, there is no formal input from regional or national planning authorities, and only limited participation from local communities, which can compromise wider sustainability goals and restrict such reserves to small areas (Svensson et al. 2009). Largely as a result of these factors, hotel involvement in marine resource management remains limited, despite rising tourist visitation and increasing numbers of protected areas globally (Spalding et al., 2013; Stolton et al., 2014; World Tourism Organization, 2014a).

The model of hotel participation I present here has shown some potentially encouraging results for Rarotonga. Unlike the majority of the examples profiled in Table 3.5, it is informal, collaborative and can be rapidly implemented at very low cost. It is instigated not by ecoresorts with explicit environmental and social programmes but by coastal hotels keen to safeguard the quality of the reef and associated fish populations for their guests.

So could this model scale to other locations more effectively than existing approaches? Two trends suggest that it might. First, the tourism industry as a whole is shifting away from ecotourism in favour of responsible tourism. Unlike ecotourism, a niche offering focussed on environmental conservation and improving the welfare of local people, responsible tourism seeks to improve the sustainability of the entire sector (Goodwin, 2011; Honey and Krantz, 2008). Crudely put, it is about all actors making small, continued sustainability improvements instead of a small niche making large advances. With responsible tourism gaining traction and with tourists increasingly seeking out environmentally conscious experiences, it is likely that more coastal hotels will be compelled to protect nearby marine resources. (Goodwin, 2011; United Nations Environment Programme and World Tourism Organization, 2012).

Secondly, in many tropical developing countries with high tourist visitation, marine resource management is increasingly being devolved to the community level, largely due to growing recognition of the relative strength of local institutions and because of disappointment with top-down, centralised government marine resource management efforts (Rocliffe et al 2014, Chapter 5). There are likely to be numerous locations around the world where these two strands will intertwine, where LMMAs that are associated with and supported by hotels could flourish. Although it is hard to determine accurate numbers given the informal nature of many LMMAs, evidence suggests that there are established initiatives in at least 19 tropical countries (Govan, 2009; Rocliffe et al., 2014). Taken together, these nations welcomed more than 10.8 million international arrivals in 2012 and are expected to steadily increase their market shares through to 2030 (World Tourism Organization, 2014a, 2011).

Of course, not all LMMAs are established in areas where there is tourist infrastructure so it is conceivable that an expanded role for hotels could serve to marginalise less accessible communities, who may paradoxically have a greater need for support. In Menai Bay, Tanzania, for example, resources from a marine conservation programme were unevenly concentrated, leading to positive outcomes in the well-connected village of Kizimkazi-Dimbani but no improvements in the more remote Fumba (World Tourism Organization, 2014b, 2011). It is also important to recognise that while the model I highlight here may offer some promise as a means to improve coastal marine resource management, it will never be sufficient in itself to overcome the numerous marine conservation challenges we face. It is therefore vital that LMMAs managed in association with hotels are never a cannibal of other, more established models of marine protection, but a complement to them.

3.7 Conclusions

Good evidence is critical for marine resource management and conservation, since without it, managers are not able to target and prioritise effectively or later defend their management decisions. Ideally, this evidence should be derived from well-designed studies that are properly conducted and analysed, with appropriate temporal and spatial replication (Levine, 2007). For spatially explicit tools like LMMAs, however, such designs are rare, principally because the informal and unfunded nature of many initiatives, together with the variability, complexity and dynamism of marine ecosystems, makes it challenging to isolate changes at LMMA sites from those due to natural processes (Underwood, 1994, 1992, 1991). As a result, managers may be forced to rely on less robust data, increasingly the likelihood of incorrect or sub-optimal priority setting.

Although an ideal sampling design may not be achievable, the evidence presented here suggests that there is greater value in adopting asymmetric designs with multiple controls than in relying solely on a single, spatially confounded control-impact comparison. While it is undoubtedly desirable to strive for better evidence of the ecological effects of area-based approaches like LMMAs, lack of formal proof of effects should not, however, be used to delay or excuse remedial activities. In data poor, informal and increasingly threatened systems, the perils of inaction are likely to outweigh the costs of action.

4 In the dark? A call for greater use of blind assessment in conservation biology

4.1 Preface

In the previous chapter, I used a long-established network of LMMAs in Rarotonga, Cook Islands as a model to explore how different research designs could lead to different management responses and highlighted the challenges of relying on single-point-in-time comparisons. In this chapter, I continue to examine the role of research design in improving assessments of area-based management initiatives, with a novel investigation into blind assessment, which is widely used in medical science to reduce important experimental biases, but has yet to be adopted in conservation science.

4.2 Abstract

Double blinding plays a vital role in modern medicine to reduce the risk of human expectation affecting findings, but is yet to be applied to all areas of scientific research. Blinding usually refers to keeping study participants, and those involved in assessment, management, or data collection unaware of the allocated treatment or true hypotheses, in order to avoid influence caused by that knowledge. On average, trials that have not blinded assessors show larger treatment effects than properly

blinded studies. However, the idea of natural human bias also has important yet currently unaddressed implications for conservation biology, for example in assessments of marine reserve performance, where researchers may expect greater fish abundance in protected compared to unprotected sites, leading to a subconscious upward bias in their estimations. Here, I present the results of a two-group double-blind randomised controlled trial based on underwater video fish surveys on Indonesian coral reefs. An unblinded group of observers were told that half of the video transects were filmed in an unfished marine reserve and half were from fished controls, whereas a blinded group were not. I compared estimates of fish abundance from both groups and found that the unblinded group significantly and erroneously overestimated the effectiveness of the reserve, inflating their fish counts by approximately 28% (95% CI 18.5% to 40.5%, $p>0.0001$). On control transects, there was no significant difference between the two groups. I conclude that blinding is a valuable tool for both conservation scientists and managers and can be used to maximise accuracy in experimental design and discern the true performance of management strategies. I also call for the development of guidelines to encourage best practice.

4.3 Introduction

The accurate estimation of wildlife abundance is of central importance to conservation science and is a critical component of effective conservation and management strategies (Gladstone et al., 2012; Ransom et al., 2012). Accuracy, however, can be compromised by the extent to which target organisms are available and detectable, as well as by the precision of those doing the estimating (Elphick, 2008; Thompson and Mapstone, 1997). Taking coral reefs as an example, if a reef fish is cryptic (Bozec et al., 2011), moves out of a study area prior to detection (Ward-Paige et al., 2010) or actively avoids (Dickens et al., 2011) or is attracted to an observer (Chapman et al., 1974), a fish count that needs to consider this species will be affected. In addition, an observer may inadvertently miscount (Alldredge et al., 2008; Frederick et al., 2003) or misidentify (MacSwiney G. et al., 2008) certain organisms, or make recording or analysis errors (Elphick, 2008).

In comparative studies, for example when assessing differences between a protected area and a control site, such biases will reduce overall accuracy and precision, but will tend to affect both estimates roughly equally and so rarely confound results greatly (Edgar et al., 2004). Further, they can typically be accommodated and minimised with techniques such as distance sampling (Newson et al., 2008), using multiple observers (Alldredge et al., 2008; Jenkins and Manly, 2008), or through comprehensive calibration and training of observers (Thompson and Mapstone, 1997). However, not all biases will act consistently across comparisons and so may cause serious misinterpretations if they are not recognised and incorporated into study designs (Edgar et al., 2004).

An important bias in this regard is expectation bias (also called ascertainment bias, information bias or detection bias), which occurs when study outcomes are shaped by the conscious or unconscious predispositions of the researcher (Hrobjartsson et al., 2013). Information that confirms one's expectations or hypotheses is often favoured, leading to a more positive assessment of the experimental group than the control (Burghardt et al., 2012; Hrobjartsson et al., 2013; Tuyttens et al., 2014). That experimenters can all too easily find what they are looking for has long been recognised: in the early 1960s, Rosenthal and Fode (1963) showed that researchers who expected rats to perform better in a maze found that they did so.

In fields as diverse as psychology (Cesario et al., 2006; Doyen et al., 2012), medicine (Bland, 2005; Day and Altman, 2000; Miller and Stewart, 2011), animal behaviour (Burghardt et al., 2012; Tuyttens et al., 2014) and the peer review of scientific literature (Budden et al., 2008; Fisher et al., 1994; Van Rooyen et al., 1999), double blinding is a vital technique for combatting expectation bias. Though usage varies (Devereaux, 2001; Moher et al., 2010; Schulz and Grimes, 2002a), the term usually refers to keeping study participants, those involved in assessment or management, and those collecting and analysing data unaware of the allocated treatment or true hypothesis of the study, so that they are not influenced by that knowledge (Boutron et al., 2007; Day and Altman, 2000). A closely related technique is randomisation, whereby participants are randomly assigned to a control or treatment group at the start of a study (Boutron et al., 2007). The use of both techniques in a double-blind randomised control is widely regarded as the best way to minimise expectation bias and thereby provide the most rigorous and reliable results about the effectiveness of a treatment or intervention (Moher et al., 2010; Schulz and Grimes, 2002a).

There is convincing evidence that not blinding observers can result in systematic and significant differences in outcome assessment (Boutron et al., 2006). In a multiple sclerosis trial, Noseworthy et al. (1994) had all patients examined by both a blinded and an unblinded neurologist, with only the unblinded neurologists' scores demonstrating a significant treatment benefit. Recent meta-analyses have confirmed this finding: randomised trials have substantially larger treatment effects if the observer is not blinded (Hrobjartsson et al., 2013, 2012).

The magnitude of this potential expectation bias effects varies (Day and Altman, 2000; Schulz and Grimes, 2002a), but blinding tends to be more important when the observer has strong expectations or a vested interest in the result and when outcome measurement is subjective (Hrobjartsson et al., 2012; Tuyttens et al., 2014). These factors are often present in ecological field research, particularly in assessments of protected area effectiveness, where studies tend to adopt a Control Impact or quasi-experimental Before-After-Control-Impact approach (Osenberg et al., 2011). Such studies usually compare conditions within a protected site to those at a control site using visual survey techniques such as belt transects. Typical results are that protected areas show statistically significant increases in species abundance compared to their controls (Lester et al., 2009). However, since researchers know which sites have received protection (i.e. treatment) and which are controls, there is a danger that they may unconsciously overestimate the effects of protection, or even detect a difference where none exists.

In this study, I investigated the potential for expectation bias in conservation research. Postgraduate students in ecology and environmental science unknowingly participated in a double-blind randomised controlled trial during a workshop to measure the effectiveness of marine protected areas. I tested two hypotheses: i) that non-blinded observers with knowledge of which site is a control and which is a marine reserve would record larger reserve effects in terms of fish abundance than blinded observers; and ii) that this effect would vary with fish abundance, with greater effects registered on transects where fish abundance was higher, a phenomenon termed "subjectivity" in medical studies (Boutron et al., 2006; Day and Altman, 2000; Moher et al., 2010; Wood et al., 2008a).

4.4 Materials and methods

4.4.1 Participants and ethics

It is difficult to properly blind field studies of marine reserves because researchers are almost always aware if they are working at a protected or unprotected site. I controlled for this by basing my experiment on underwater video recordings rather than live observations. Twenty postgraduate students from the University of York, UK

(13 women, 7 men) between the ages of 21 and 40 (median age: 23.5) took part in the experiment, which formed part of a compulsory workshop for a course on the management and design of protected areas. The experiment and its associated protocols were approved by the Ethics Committee of the Environment Department at the University of York.

Using a computer-generated list of random numbers, I randomly extracted 30 one-minute video transects from an ongoing reef fish monitoring programme conducted by Operation Wallacea on the reefs surrounding Hoga Island Marine Research Station in Southeast Sulawesi, Indonesia. Fifteen of these were randomly designated as "marine reserve" transects and 15 as "control" transects. In other words, there was no reserve as such. The 50 x 5m belt transects followed the clearly visible reef crest at each site at a depth of 1m to 6m and were filmed in high definition (HD), using Canon HFS21 cameras in an underwater housing.

4.4.2 Design and procedure

The trial participants (herein "observers") were first given a 30-minute workshop covering marine reserve assessment design, and techniques for surveying coral reef communities including point counts, belt transects and video-based methods. Observers then received a short training session on how to estimate fish abundance from video transects and were misleadingly informed that they would be using these skills in an experiment to assess the effectiveness of a new high definition video technique for estimating fish populations.

For this experiment, I randomly split the observers into two groups of equal size (n=10): an unblinded group, who were told which transects were from the dummy reserve and which were from controls through the use of onscreen titles; and a blinded group, who were not told. Randomisation was achieved through use of a computer generated random number list prepared by an investigator (LGA) with no involvement in the experiment itself (Jadad et al., 1996; Moher et al., 2010). Both groups watched the same 30, one-minute long video transects on individual computer workstations and counted all non-cryptic reef-associated fish using a standard underwater visual census (UVC) protocol based on English (1997). The seating plan was structured to ensure that observers could only see the screens of those in the same treatment group as themselves (Day and Altman, 2000).

As far as possible, the experiment was designed to replicate field conditions in studies using UVC. For example, observers were not allowed to speed up, slow down or pause transects, nor confer with each other about their fish counts. To ensure that observers all experienced the same counting conditions, the specification of each computer and monitor were identical. To avoid order effects, transects were stratified into control and reserve and randomised, meaning transect order consistently alternated between control and reserve.

After they finished the fish counts, the observers were given a questionnaire to test their perception of the study (Bang et al., 2010; Fergusson, 2004). This included the question "The task you just completed was a series of video transects. What do you think the purpose of this task was?". Responses showed that no observer was aware of the true nature of the research objective. The trial assessor (SR) then fully debriefed the two groups, explaining the experimental hypothesis verbally as well as giving each observer a written version. Written informed consent was also sought and observers were offered the opportunity to refuse the use of their data. All consented.

To conclude the trial, two of the research team (SR and LGA) independently counted the actual number of fish on each of the one-minute transects. The team paused, rewound and slowed down the recordings as necessary for accurate counts. Both were blinded as to which transects were from dummy controls and which were from dummy reserves. There was no significant difference between the two observers' estimates (paired T test, $p=0.4432$).

4.4.3 Levels of blinding and randomisation

In medicine, randomised control trials typically have participants (usually the patient receiving the treatment or the control/placebo), observers/investigators (typically a doctor or healthcare provider), assessors (those running the trial) and data analysts. Table 4.1 outlines how these apply to the specific focus of this project.

TABLE 4.1 LEVELS OF BLINDING IN MEDICINE AND CONSERVATION BIOLOGY

	MEDICINE		CONSERVATION BIOLOGY		
Level	Medical trial	Can be blinded?	Marine reserve assessment	Normally blinded?	Blinded in this study?
Participant	Patient receiving treatment or placebo	Yes	Treatment is a no-take marine reserve; placebo is the control site	NA	NA
Observer/Investigator	Doctor assessing treatment	Yes	Observer recording fish abundance on reserve and control transects	No	Yes
Assessors	Person running the trial	Yes	Person coordinating observers/running the experiment	No	Yes
Data analysts	Person analysing data	Yes	Person analysing data	No	Yes

In addition to randomising the observers into two groups and blinding both to the aims of the study until after completion of the experiment, I introduced several further levels of blinding to help ensure reliability of the conclusions reached (Table 4.1). First, to prevent the research team from subconsciously transferring attitudes for or against marine reserves to the student observers (Doyen et al., 2012; Schulz et al., 2002; Wolf, 1950), I recruited an independent invigilator from a non-ecological discipline to administer the experiment. The invigilator was blinded to the aims of the study, which I confirmed with a post-trial debrief and questionnaire.

Second, we blinded the assessor (SR) prior to and during the trial. This was to prevent him from influencing which observer was assigned to which group, for example as a result of unconscious knowledge that a particular observer was extremely passionate about marine reserves and likely to register a higher bias (Schulz and Grimes, 2002b). This technique is known in medicine as allocation concealment (Pildal et al., 2008).

Third, to prevent knowledge of whether observers were blinded or unblinded from influencing the selection of methods and analytical approaches, SR was also blinded during data analysis (Boutron et al., 2007; Polit, 2011). In both the data analysis and the concealment of allocation, LGA acted as a confederate, maintaining a password protected spreadsheet of treatment allocation, providing data with this allocation masked, and revealing which group was which only after the analysis had been completed.

4.4.4 Statistical analysis

I performed three main analyses to explore the influence of expectation bias on abundance counts. First, I constructed a series of generalized linear mixed effects models (GLMMs) to determine whether blinding influenced counts of fish abundance on control and reserve transects. In all models, I treated the response variable abundance (count data) and blinding (categorical factor with two levels) as fixed effects and transect and observer as random effects. I assumed a Poisson error distribution and used a log link function, allowing for the over-dispersion suggested by initial data exploration by modelling observer as an individual-level random effect (Elston et al., 2001). I assessed model assumptions with graphical procedures, evaluating homogeneity of variance by plotting residuals against fitted values, and normality of residuals with quantile-quantile plots and histograms. I fitted all GLMMs using the Laplace approximation. Models were evaluated with likelihood ratio tests comparing the full models to null models with the predictor variable (whether the observer was blinded) removed. I estimated p-values for the full versus null model comparisons with parametric bootstrapping (10,000 iterations).

Secondly, to determine the overall effect of the dummy marine reserve I calculated the natural logarithm of the Response Ratio (ln *RR*). The response ratio, a common measure of effect size in marine reserve studies (e.g. Côté *et al.* 2001; Halpern 2003; Samoilys *et al.* 2007; Lester *et al.* 2009; Miller *et al.* 2011) is defined as the ratio of the mean fish abundances in experimental (protected) and control (fished) areas. Ratios greater than 0 suggest higher fish abundance within reserves than in controls; values less than 0 indicate the reverse – i.e. a negative response to protection. For each of the three subgroups of blinded observers, unblinded observers and the research team, I pooled individual observer abundance estimates by meta-analysis using fixed effects models, thereby assuming that the true effect size for all estimates was identical, and treating variations among observers as a form of sampling error. Effect sizes were regarded as significantly different from zero when the confidence interval of the grand mean did not overlap zero. To aid interpretation, I back transformed and converted ratios to a percentage increase or decrease.

I also used log response ratios to test the hypothesis that the magnitude of the expectation bias would be greater on those reserve transects with higher fish abundance. Here, I defined ln *RR* as the ratio of the mean abundances from blinded and unblinded observers per reserve transect. By using random effects models, I allowed the effect size to vary among transects (Jauni et al., 2014). Results were interpreted and calculated in the same way as in the effect size analysis. All analyses were performed in R (version 3.1.1 R Development Core Team, 2014) with the packages lme4, pbkrtest, metafoR and boot.

4.5 Results

Results from the assessment of fish abundance carried out by the research team indicated that the 30 transects on which the observers estimated abundance were highly heterogeneous, with a range of between 10 and 234 fish per transect. Overall, there was no significant difference between fish abundance on “control” and “reserve” transects (Mann-Whitney U Test, p=0.619).

4.5.1 Effects of blinding on estimates of abundance

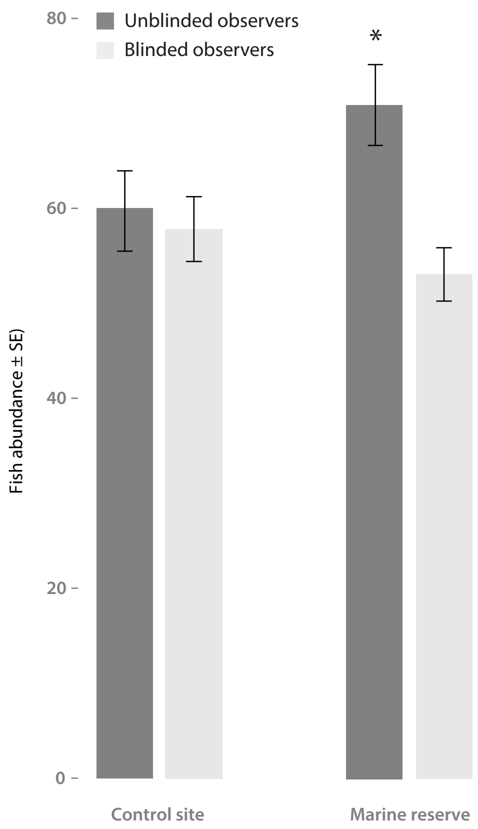

FIGURE 4.1 FISH ABUNDANCE ESTIMATES FOR BLINDED OBSERVERS (LIGHT GREY BARS) AND UNBLINDED OBSERVERS (DARK GREY BARS) FOR A DUMMY MARINE RESERVE AND A DUMMY CONTROL SITE. UNTRANSFORMED DATA

* denotes a significant difference between blinded and unblinded observations. Error bars are ± SE. Unblinded observers knew which transects were from the “marine reserve” and which were from the “control”. Blinded observers did not know which transects were from the “reserve” or “control”.

On “control” transects, abundance estimates for blinded and unblinded observers were not significantly different from each other (p = 0.872, Table 4.2, Figure 4.1). However, on “reserve” transects, failing to blind observers had a highly significant (p < 0.001) and large positive effect on counts of fish abundance.

TABLE 4.2 EFFECTS OF BLINDING ON ABUNDANCE: GENERALIZED LINEAR MIXED MODEL (GLMM) PARAMETER ESTIMATES, STANDARD ERRORS (SE), BOOTSTRAPPED *P* VALUES AND R^2 GOODNESS OF FIT STATISTICS FOR “CONTROL” AND “RESERVE “TRANSECTS.

Transect Group	Estimate	SE	P value	Conditional R^2
Fished “control”	0.008	0.048	0.872	0.944
Protected “reserve”	0.273	0.043	<0.001	0.958

For “control” and “reserve” transect groups, conditional R^2 values were close to one, suggesting high goodness-of-fit for both models (Johnson, 2014; Nakagawa and Schielzeth, 2013). Prior to modelling, I checked the data for homogeneity of variance. Fligner–Killeen tests revealed no compelling evidence of heteroscedasticity among treatments (control and reserve, p = 0.214) or among subgroups (blinded control, blinded reserve, unblinded control, unblinded reserve, p = 0.233).

4.5.2 Reserve effects

The actual log response ratio of the dummy marine reserve, derived from the repeated blinded estimates of two of the research team was -0.149, meaning there were 13.8% fewer fish on the “reserve” transects than on “control” transects (Figure 4.2. The exponential of [response ratio-1] x 100). The log response ratio for the blinded observers was similar in magnitude and direction to the actual ratio. For this group, the overall effect size was -0.07 (95% CI -0.20 to 0.05), indicating that blinded observers estimated 7.2% fewer fish (95% CI -18% to 4.92%) in the dummy reserve than in the dummy control site (Figure 4.2). This was neither significantly different from zero nor from the actual response ratio.

Dataset	*RR* (95% CI)	SE	N
Unblinded observers	0.178 (0.04 to 0.316)	0.070	10
Blinded observers	-0.075 (-0.199 to 0.048)	0.063	10
Actual response ratio	-0.149 (-)	0.159	2

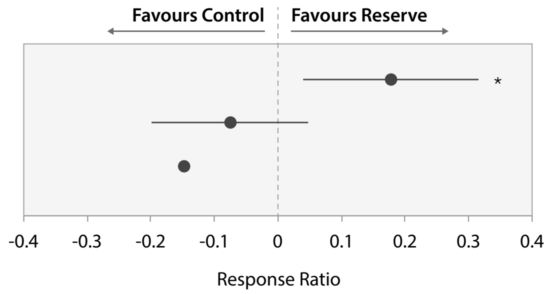

FIGURE 4.2 OVERALL EFFECT SIZES FOR DIFFERENCES IN FISH ABUNDANCE BETWEEN "CONTROL" AND "RESERVE" SITES. *RR* NATURAL LOG RESPONSE RATIO, *SE* STANDARD ERROR, *N* NUMBER OF EFFECT SIZES USED TO CALCULATE EACH GRAND MEAN

Unblinded observers knew which transects were from the "marine reserve" and which were from the "control". Blinded observers did not know which transects were from which. Black circles are grand means. The bars around the black squares depict 95% Confidence Intervals. An effect size is considered significantly different from zero if its confidence interval does not overlap zero. Positive effect sizes indicate that "reserve" sites have a greater average abundance than "control" sites. * Effect sizes for unblinded observers were both significantly positive and significantly different from the other two moderators (omnibus test of moderators p= 0.0147).

In contrast, the overall response ratio for the unblinded observers was 0.18 (95% CI 0.04 to 0.32), meaning that this group estimated 19.5% more fish (95% CI 4.08% to 37.2%) in the "reserve" than in the "control". This effect size was both significantly positive and significantly different from the blinded response ratio and the true response ratio (omnibus test of moderators p= 0.0147). In all models, no significant within-group heterogeneity was observed (Q=0.49, p=1.00 for blinded observers and Q=1.66, p=0.99 for unblinded observers).

Overall, the actual response ratio revealed that there were fewer fish on transects I had designated as reserve compared to those I had designated as control transects. However, whilst results from the blinded group of observers agreed with this finding, the unblinded group estimated that there were significantly more fish in the "reserve" than in the "control". Therefore, in this case, by blinding the observers, I prevented a statistical Type one error.

4.5.3 Effects of fish abundance on magnitude of expectation bias

I used meta-analytic random effects models and log response ratios to further explore the effects of blinding on counts of fish abundance on "reserve" transects (Figure 4.3). The overall effect size was 0.25 (95% CI 0.17 to 0.34). Thus, unblinded observers recorded significantly greater fish abundances compared to blinded observers on "reserve" transects (p=<0.0001). Model heterogeneity was low and non-significant (I^2 = 25.58%, p = 0.27). I also used this approach to determine whether expectation bias was greater on "reserve" transects with higher numbers of fish than on transects with fewer fish. To this end, the effect sizes in Figure 4.3 are sorted in descending order by transect mean. If expectation bias were larger on transects with higher fish abundance, I would expect the individual effect sizes in Figure 4.3 to be further away from the dotted line at the top of the figure and nearer to it towards the bottom, but this is not the case. Indeed, although the overall effect size on the seven transects with the highest fish counts is marginally greater than on the eight least abundant transects (0.269 vs. 0.238), the difference is not statistically significant (p = 0.73).

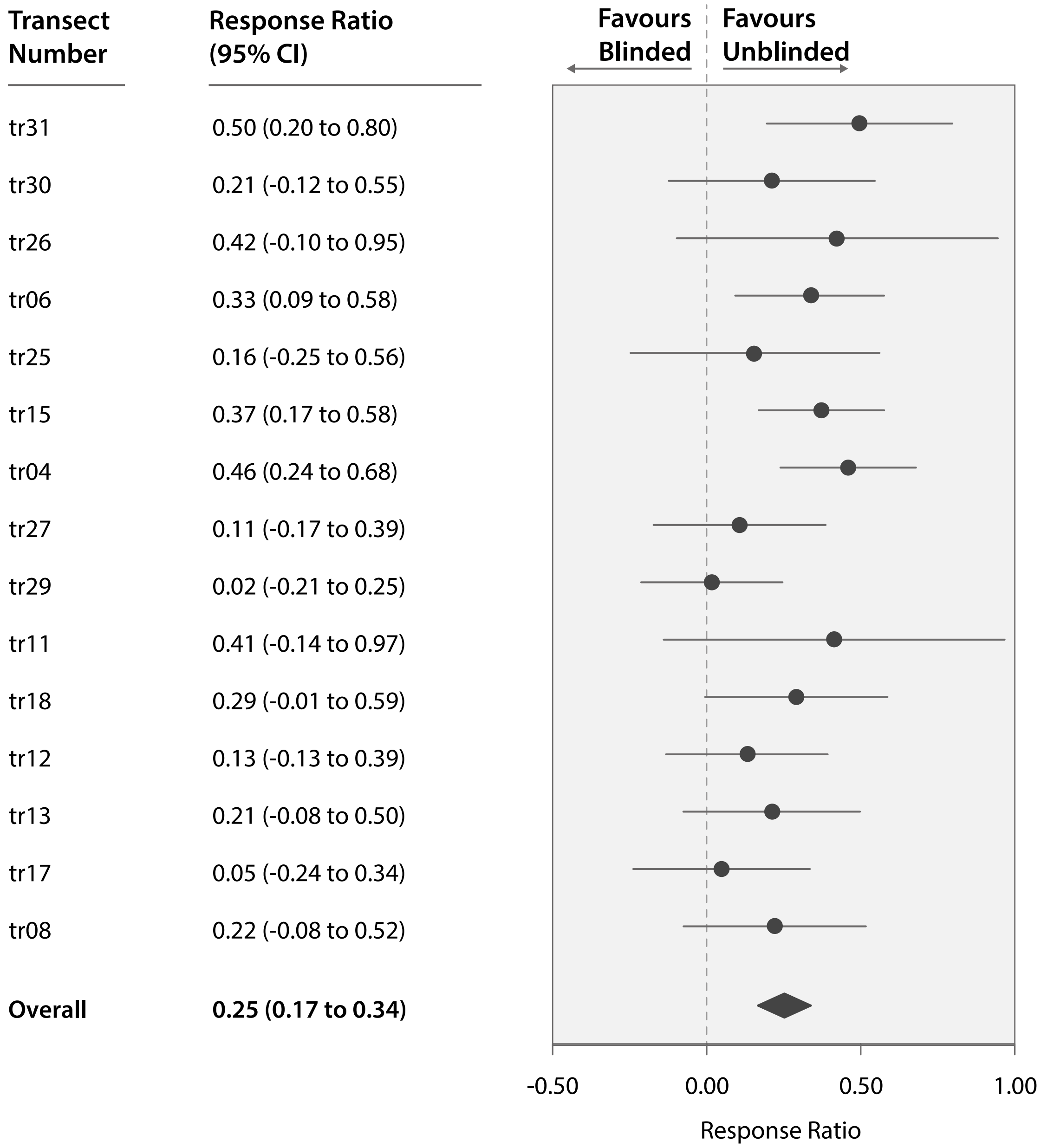

FIGURE 4.3 OVERALL EFFECT OF BLINDING ON FISH ABUNDANCE RECORDED ON "RESERVE" TRANSECTS, AND THE EFFECTS OF FISH ABUNDANCE ON MAGNITUDE OF EXPECTATION BIAS

Note: The bars around the black circles depict the 95% Confidence Intervals (CIs). An effect size is considered significantly different from zero if its confidence interval does not include zero. Positive effect sizes indicate that unblinded observers recorded greater fish abundance than blinded observers on the same transect. The effect sizes are sorted by transect mean in descending order. Larger means do not yield more positive effects than smaller means. The *Overall* effect size was derived from a random effects model with low, non-significant heterogeneity (I^2 = 25.58%, p = 0.27) and was significantly positive (p=<0.0001). Response ratio is natural log transformed.

4.6 Discussion and conclusion

This study provides clear evidence that expectation bias could significantly confound results in assessments of the ecological effectiveness of protected areas. To my knowledge, this is the first quantitative demonstration of this bias in conservation biology. Our analyses show that observers who were told which video transects came from a dummy marine reserve and which from an unprotected dummy control site (the unblinded group) counted significantly more fish on "reserve" transects than observers who were not given this information (the blinded group), inflating their counts by approximately 28% (95% CI 18.5% to 40.5%), even though there was no reserve as such and the real counts were not different for reserve and control transects. On "control" transects, there was no significant difference between the two groups. This suggests a one-way bias: although knowledge of which video transects came from "reserves" resulted in increased estimates of fish abundance on those transects, knowledge of which transects came from "controls" did not appear to cause a corresponding decrease in fish counts from observers.

In my experiment, the reserve-allocated transects had by chance 13.8% *fewer* fish than those allocated to the control site. Though, the blinded group's findings agreed with this result, the unblinded group recorded significantly more fish on "reserve" transects than on "controls". In a real world scenario, this would lead to inaccurate feedback to conservation managers and thus likely hinder their ability to react to changes in local fisheries. I interpret this as a substantial risk of bias resulting from a failure to blind observers, though the magnitude of error from this single study is relatively conservative when compared to other disciplines. In medicine, for example, a meta-analysis investigating the impacts of blinding reported that non-blinded observers exaggerated effect sizes by about 68% (Hrobjartsson et al., 2013).

Evidence from medicine also suggests that expectation bias is larger when outcomes are more subjective (Boutron et al., 2006; Day and Altman, 2000; Moher et al., 2010; Wood et al., 2008a). With respect to the present study, I hypothesised that the bias would be larger on transects with higher fish abundance, since these may be harder to assess accurately than those with fewer fish. However, my analyses did not find a significant relationship between level of bias and fish abundance. This could suggest that blinding is equally important in areas of both high and low fish density, but it could also be argued that the absence of an effect was due to the heterogeneous nature of the transects and the small sample size. I would therefore encourage future research to address this question more fully.

An important consideration is how my findings apply to real world ecological surveys; to what extent does the expectation bias reflect that found in practising marine conservation surveys? Most researchers involved in the collection of data from underwater surveys would tend to be trained to a higher level than the students who participated in the study. It is conceivable that the subjects would have been more optimistic about the potential of MPAs than more experienced scientists, and therefore more biased in their assessments. Conversely, as established researchers may have deeply held convictions or substantial investments in a particular research outcome or conservation success, they may in fact be more susceptible to expectation bias than the students who participated in the experiment (Marsh and Hanlon, 2004; Tuyttens et al., 2014). This may be especially true in disciplines like conservation biology and environmental science, where political and financial pressures to translate findings into policy continue to swell the ranks of advocate scientists (Caveen et al., 2014; Kaiser, 2000; Nelson and Vucetich, 2009). Accordingly, future research should as a priority seek to replicate this study across a large sample of practicing researchers. To guard against potential confounding in other research designs, I also encourage replication of this study in situations that compare between protected and unprotected sites and (a) use subjective methods like DAFOR (dominant, abundant, frequent, occasional or rare) or percentage cover, (b) estimate transect or point-count cylinder boundaries, or (c) assess organism size as well as abundance.

Although the evidence presented here suggests that blinding is clearly desirable to reduce experimenter biases in comparative assessments of MPAs, it is difficult to establish and maintain when in-situ techniques such as UVC are used. Even if those conducting the census were initially unaware of the management status of a particular site, external information (e.g. maps or signs) would inevitably cause a breaking of the blind. As such, if MPA researchers are to adopt blind assessment, video-based methods will need to be more widely embraced.

Compared to UVC techniques, stereo-video systems have been shown to increase accuracy of fish length estimates and definition of transect boundaries, allow quicker surveying in water and provide a permanent record of the transect that can be independently validated or reanalysed by other observers (Harvey et al., 2004; Pelletier et al., 2011). By allowing transects to be completed more quickly, diver operated stereo-video is also likely to reduce overestimation of highly mobile species that move along the transect entering and exiting the survey zone (Ward-Paige et al., 2010). Recent work has also concluded that diver operated stereo-video systems are more suitable than underwater visual census for detecting fishing impacts on fish communities (Goetze et al., 2015).

One downside is that video footage may not be of sufficient quality for accurate species identification. This has resulted in lower reported species richness than for UVC, though improvements in technology continue to reduce this problem (Holmes et al., 2013; Pelletier et al., 2011). Further, while video techniques require less time in water than UVC, post-survey data processing time is a concern. Stereo-video fish surveys have been found to take 2-3 times longer to obtain data from than UVC (Holmes et al., 2013), though when using a defined species list the total time for both techniques has been shown to be comparable (Goetze et al., 2015).

Assuming the data collector and analyst are different members of the research team and video surveys are already being used, the additional costs, time and logistical challenges associated with implementing a blinding protocol would be negligible, especially given the consequential reduction in bias and improvement in the reliability of the outcomes. Where the objectives, logistics, or budget of the research favour UVC over video methods, a blended approach could be adopted. This would involve having a subset of the data analysed by blinded video observers and unblinded UVC surveyors and then using the differences between the two as a correction factor on the rest of the UVC dataset.

In conclusion, my analyses demonstrate that expectation bias may present a potentially serious problem to assessments of protected area effectiveness. Although I have focused on marine reserves, there is no reason to believe this bias would not arise in other areas of conservation biology that involve comparative estimates of abundance using techniques prone to observer bias. Accordingly, if we want to be certain that any differences calculated between two groups stem from a given treatment rather than from the biases of the observer, I suggest that blind assessment should receive greater attention from conservation biologists. Further, I recommend the development of guidelines to help ecological researchers to better understand under what circumstances blinding may benefit research design, as well as to provide direction on how to properly implement a blinded protocol. Given that conservation biology has already learnt much from medicine, particularly in the adoption of systematic reviews and meta-analysis (Pullin and Knight, 2001; Pullin and Stewart, 2006), such a document could be based on the CONSORT statement, which provides guidance to medical researchers on improving the reporting of randomised controlled trials (Schulz et al., 2010).

Ultimately, expectation bias results from the predispositions of the observers, and these may fluctuate unpredictably from study to study and observer to observer (Hrobjartsson et al., 2013, 2012). As such, I would caution against using the pooled average of the difference between blinded and unblinded observers as a correction factor for existing and future comparative studies of MPAs (Hrobjartsson et al., 2012). Although these findings may at first glance appear to be questioning the integrity of protected area researchers, the expectancy effects demonstrated here are often subconscious (Day and Altman, 2000). When double-blind randomised controlled trials were first proposed in medicine, they were vehemently opposed by some clinicians as an affront to their own expert judgement (Haynes et al., 2012). Medicine has since grown to embrace such methods enthusiastically. Based on the findings I report here, conservation biology must urgently consider doing the same.

5 TOWARDS A NETWORK OF LMMAS FOR THE WESTERN INDIAN OCEAN

5.1 Preface

The World Database on Protected Areas (WDPA) provides a publically accessible overview of the location, status, age and size of the world's marine protected areas and is used to track progress towards international biodiversity targets. Because the IUCN – the body responsible for the WDPA – does not consider that all LMMAs are automatically MPAs, many are not included in this database, and as such, little is known about the status or extent of LMMAs outside the Western and Central Pacific region, where the overwhelming majority of research effort has been concentrated to date. In this chapter, I address this gap by conducting the first assessment of LMMAs in the Western Indian Ocean, examining them in terms of geography, numbers, size, governance structures and the legal frameworks that underpin them.

5.2 Abstract

In the Western Indian Ocean, local communities are increasingly assuming responsibility for inshore marine resources either on their own or through collaborative management arrangements with governments or non-state actors. In this paper, I trace the evolution and expansion of community management in the WIO and present the first ever inventory and assessment of the region's locally managed

marine areas. I compare the key attributes of these areas to those under government stewardship and assess their relative contributions to progress towards the Convention on Biodiversity target of 10% of marine and coastal ecological regions to be effectively conserved by 2020. I also explore the legal frameworks that underpin locally managed marine initiatives in Kenya, Madagascar, Mozambique and Tanzania to assess the potential for future expansion. A principal finding is that whilst LMMAs protect more than 11,000 square kilometres of marine resource in the WIO, they are hampered by underdeveloped local and national legal structures and enforcement mechanisms. In my recommendations to improve local management, I suggest establishing a network of LMMA practitioners in the WIO region to share experiences and best practice.

5.3 Introduction

Despite their value to humans, marine ecosystems worldwide are threatened by a range of anthropogenic pressures, including pollution, habitat loss, climate change and overfishing (Halpern et al., 2008; Jackson, 2008; Lester et al., 2009). These impacts have drained populations of culturally and economically important fish stocks and reduced structural complexity of various marine communities across a rich range of habitats, species and trophic levels (Fraschetti et al., 2011; Graham et al., 2008; Lester et al., 2009).

In the Western Indian Ocean (WIO) as throughout the world, Marine Protected Areas (MPAs) have been a primary management approach in attempts to alleviate anthropogenic pressures (IUCN, 2004). An MPA is defined by IUCN as: "A clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values." (Day et al., 2012). Solid evidence from MPAs, particularly for No-take Zones (MPAs that allow no extraction), shows that protection can increase average size, diversity, abundance and biomass of species (Lester et al., 2009; Roberts and Hawkins, 1997; Russ and Alcala, 1996) and that some of this biomass can be exported beyond protected boundaries (Gell and Roberts, 2003; Russ et al., 2004; Russ and Alcala, 2010). MPAs can also play an important role in climate change adaptation, enhancing ecosystem resilience and protecting vital ecosystem services (Hastings et al., 2012; Van Lavieren and Klaus, 2013).

In 2002, international leaders at the World Summit on Sustainable Development set the first target for the establishment of a global system of MPAs (World Summit on Sustainable Development, 2002). This target was formally quantified four years later, when the parties to the Convention on Biological Diversity (CBD) committed to effectively conserving 10% of each of the world's ecological regions by 2012 (CBD, 2006). In 2010, the parties pushed back the deadline to 2020 and adopted Aichi Biodiversity Target 11, with a revised goal of conserving "at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures" (CBD, 2010).

Initial progress towards these targets was slow: based on the rate of MPA expansion to 2008, Wood estimated that the 10 percent figure would not be achieved until 2047 (Wood et al., 2008b). In contrast, the most recent analysis (Spalding et al., 2013), paints a more optimistic picture. MPA coverage has increased dramatically, quadrupling between 2002 and 2012 (*ibid.*). MPAs now cover 8.3 million km^2, 2.3% of the global ocean area and 7.9% of the continental shelf and equivalent areas (i.e. less than 200m deep) (*ibid.*). So pronounced is the increase that the 10 percent Aichi CBD target could be reached, even before 2020 (*ibid.*). However, a few very large MPAs are largely responsible for this apparent reversal of fortunes, a trend that looks set to continue as new super-sized protected areas come online in Cook Islands and New Caledonian waters (Pala, 2013; Spalding et al., 2013). Further, most of these MPAs are located in uninhabited or low-population-density areas (Spalding et al., 2013) and/or in developing countries where enforcement is weak to non-existent (Samoilys et al., 2007b).

Although their popularity continues to increase, marine protected areas often fall short of their original goals and sometimes fail entirely, though published negative evaluations are rare. Inadequate long-term funding and widespread management failure have resulted in unenforceable and ineffectual "paper parks" (Jennings, 2009). The most recent global evaluations suggest that less than 16% of MPA managers feel they have adequate funding for effective conservation (Balmford et al., 2004) and that just 15% of coral reef MPAs are effectively managed (Burke et al., 2011). Regional evaluations have reached similar conclusions. In a recent review of marine conservation successes in the WIO, for example, Samoilys & Obura (2011) only mention one example of successful government-established MPAs: those of Kenya.

5.3.1 Locally Managed Marine Areas

As a result of disappointment with top-down, centralised government interventions, and facilitated by increasing recognition of the relative strength of local institutions (Christie, 2004; Cinner et al., 2009; McClanahan et al., 2006) local communities throughout the Indo-Pacific are increasingly assuming responsibility for inshore marine resources through collaborative partnerships with governments and/or non-state actors (Alcala and Russ, 2006; Cinner et al., 2012a; Govan et al., 2006; Wamukota et al., 2012). In the Pacific, areas where marine resources are at least in part under community control are usually termed "locally managed marine areas" (LMMAs) (Govan, 2009). In the WIO, the terms used vary more widely, encompassing not only LMMA, but also Collaborative Fisheries Management Area and Community Conservation Area, as well as local names such as *tengefu*, *hifadhi za kijamii* and *vilindo vya wenyeji* in Kenya (Anderson, 2012; Samoilys and Obura, 2011, Table 1.4). Although the level of community involvement and the overall management model is context-specific to an extent, a key aspect is local control. Technical support may be provided by government agencies, private sector stakeholders or non-governmental organisations, but it is the resource users themselves who make most of the management decisions, including the location of any protected areas (Evans et al., 2011; Gutierrez et al., 2011; Tawake, 2007).

Despite its relatively recent popularity, the approach of managing and conserving marine resources at the local level is actually centuries old (Christie and White, 1997; Cinner et al., 2009; Johannes, 1978). In many tropical nations, especially in Pacific Island countries, informal systems of community marine management were in place prior to colonialism (Christie and White, 1997; Johannes, 1978). Further, the tradition of customary marine tenure (CMT) – the right to control access to local fishing grounds – in Pacific Island countries provided an ideal socio-cultural platform on which modern day LMMAs could evolve (Johannes, 1981). However, although there are sacred coastal sites that are protected for spiritual reasons in places like Kenya and Tanzania (Burgess et al., 1998; Metcalfe et al., 2010) as well as several taboos around fishing (Glaesel, 2000), there is no tradition of CMT and this may partly explain why the establishment of LMMAs is a more recent phenomenon in the region.

In the Pacific, more than 500 communities in 15 countries manage 12,000km^2 of coastal resources, 1,000km^2 of which constitutes full no-take protection (Govan, 2009). In the Western Indian Ocean, little is known about the status or extent of

LMMAs. In this paper, I present the first inventory of LMMAs in the WIO and assess them in terms of geography, numbers, size and governance structures. I compare the key attributes of these areas to those under government stewardship and evaluate potential contributions to international biodiversity commitments. To determine prospects for future LMMA expansion, I also explore the legal frameworks that underpin locally managed marine initiatives in Madagascar, Kenya, Mozambique and Tanzania. Finally, I make recommendations for improving local marine management, including the establishment of a regional network of practitioners to facilitate the sharing of experiences and best practice.

5.4 Methods

5.4.1 Locally Managed Marine Areas: a definition

Following Govan et al (2009), I define a locally managed marine area as

> *"An area of nearshore waters and coastal resources that is largely or wholly managed at a local level by the coastal communities, land-owning groups, partner organizations, and/or collaborative government representatives who reside or are based in the immediate area."*

Under this definition, LMMAs are managed for sustainable use rather than for conservation *per se* (Burke et al., 2011). Many LMMAs employ a combination of management techniques, including periodic closures, gear restrictions, species specific reserves and permanent fully protected (closed) no-take zones (Abunge, 2011; Mills et al., 2011).

This wide variety of approaches and focus on sustainable use has lead some to question whether LMMAs should qualify as protected areas and thereby count towards international biodiversity targets (Govan and Jupiter, 2013). For the IUCN, for example

> *"only those areas where the main objective is conserving nature can be considered protected areas; this can include many areas with other goals as well, at the same level, but in the case of conflict, nature conservation will be the priority" (Day et al., 2012).*

Others (Govan and Tawake, 2009; Secretariat of the Convention on Biological Diversity, 2004), notably the CBD, are perhaps mindful of the informal nature of many LMMAs and less concerned about the need for an overarching conservation objective.

By assuming that LMMAs can help WIO nations to meet their CBD obligations, this second approach is the one I adopt here.

Based on categories suggested by Sen and Raakjaer Nielsen (1996) I developed a typology that classifies sites in the WIO along a four-point spectrum according to the extent to which resource management is shared between government and user groups.

Level 1: *Central:* Governments or non-state actors designate and manage the area. No mechanisms exist for dialogue with users and decisions are taken by resource managers.

Level 2: *Consultative:* Governments or partner organisations designate and manage the area. Whilst mechanisms exist for dialogue with users, in practice, most decisions are taken by resource managers.

Level 3: *Cooperative*: Local communities and governments or non-state actors cooperate together as equal partners in decision making.

Level 4: *Local:* In this type of arrangement, government has delegated management authority to local communities. The remit of government or partner organisations is largely restricted to providing advice and endorsing management decisions made by local communities.

For this paper, I classified level 3 and 4 sites as LMMAs and level 1 and 2 sites as MPAs. For the sake of clarity, I refer to the four levels collectively as Marine Managed Areas (herein MMAs) (Govan and Tawake, 2009).

5.4.2 Study site

The Western Indian Ocean region refers to the African coastal states of Somalia, Kenya, Tanzania, Mozambique and South Africa, together with the Indian Ocean island states of Comoros, Madagascar, Mauritius and Seychelles, as well as the two French overseas departments Mayotte and Réunion (Figure 5.1). Basic geographic and socio-economic information for the region is summarised in Table 5.1.

The mainland WIO area stretches for 13,000km along the coast from Somalia in the north to South Africa in the south. The island states consist of more than 400 islets and islands with a combined coastline of 6,360km. The region is ecologically and socio-economically diverse. Overall species composition is rich, exceeding 11,000 species of marine flora and fauna, 60-70% of which are endemic to the Indo-Pacific ocean

(UNEP/Nairobi Convention Secretariat, 2009; WWF, 2004). There are at least 369 species of coral, 10 mangrove and 12 seagrass, 2,200 coastal fishes, 3,000 molluscs, 450 crabs, 400 echinoderms and five of the world's seven marine turtle species (Obura, 2012; UNEP/Nairobi Convention Secretariat, 2009; WWF, 2004).

The WIO region has a population of around 152 million people, of which approximately 48.3 million (31.8%) live within 100km of the coast (Table 5.1). Population density is diverse, ranging from 15 people per square kilometre in Somalia to 395 in Comoros and 631 in Mauritius. Economically, excluding the French territories (Mayotte, Réunion), GNI per capita totals in the island nations are significantly higher than on the mainland, with figures from the Seychelles more than

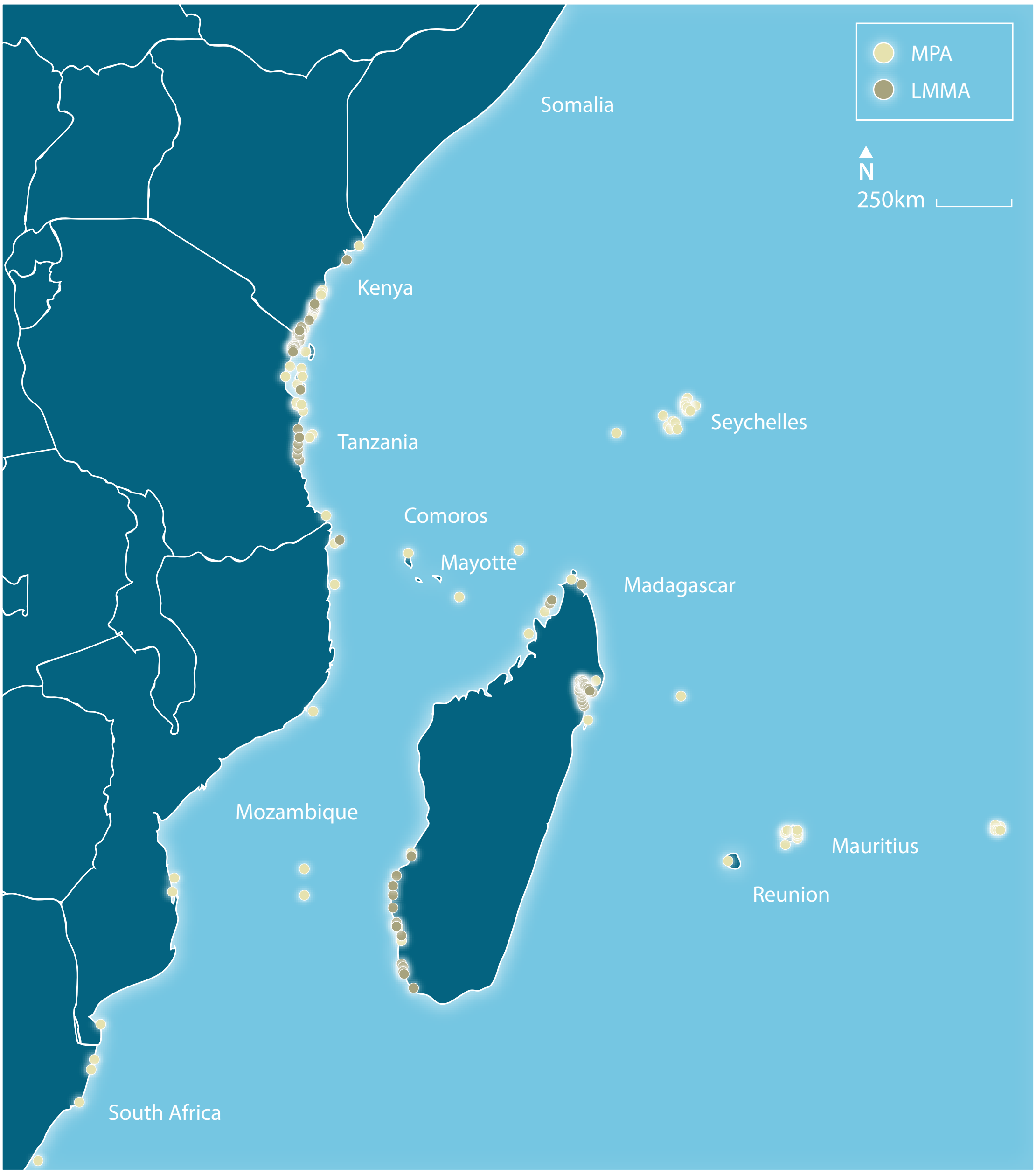

FIGURE 5.1 LMMAS AND MPAS IN THE WESTERN INDIAN OCEAN.

12 times those of Kenya and more than 18 times those of Tanzania, two of the largest economies on the mainland. The higher level of socio-economic development in Mauritius and the Seychelles is also underscored by human development index scores, with Seychelles ranked 52 in the world and Mauritius 77 – 91 places higher than Kenya, which is ranked third in the WIO region.

TABLE 5.1 GEOGRAPHIC SCOPE, POPULATION AND SOCIO-ECONOMIC CHARACTERISTICS OF WESTERN INDIAN OCEAN (WIO) NATIONS

Country	Area (km^2) [1]	Coastline (km)	Pop. (m) 2010	% Coastal pop. 2000 [2]	Pop. density km^2	GDP 2010 (US$bn)	GNI per cap. 2010 (US$) [3]	HDI* 2011
Comoros	1,860	340	0.73	100	395	0.54	750	0.433
Kenya	569,140	536	40.51	7.7	71.2	31.41	790	0.509
Madagascar	581,540	4,828	20.71	50.5	35.6	8.72	430	0.48
Mayotte [4]	374	-	0.19	100	-	-	-	-
Mauritius	2,030	372	1.28	100	631.1	9.73	7,750	0.728
Mozambique	786,380	2,470	23.39	55.9	29.7	9.59	440	0.322
Réunion [5]	2,513	-	0.83	100	331.7	-	-	-
Seychelles	460	491	0.09	100	188.1	0.94	9,760	0.773
Somalia	627,340	3,025	9.33	62.7	14.9	-	-	-
South Africa[6]	94,361	-	10.27	37.3	108.8	57.46	6,090	0.619
Tanzania	885,800	1,424	44.84	19.8	50.6	23.06	530	0.466

Source: The World Bank (2010), except for coastal population and HDI, which are provided respectively by CIESIN (2007) and UNDP (2011).

Note: South Africa only includes KwaZulu Natal as the Province that borders the WIO

* HDI=Human development index (http://hdr.undp.org/en/statistics/indices/hpi/)

1. Excludes area under inland water bodies, national claims to continental shelf, and exclusive economic zones
2. Percentage of population living within 100km of a coastline
3. Gross national income, calculated using World Bank Atlas Method
4. Figures for Mayotte are calculated from INSEE (2010). Population estimate from 2009
5. Figures for Réunion are calculated from INSEE (2011).
6. South Africa only includes KwaZulu Natal, the Province that borders the WIO. Area and population figures from Statistics South Africa (2012); GDP figures calculated from Statistics South Africa (2011)

5.4.3 Data compilation

To gather data on level 1 and 2 sites, I synthesised a list of MPAs in the eleven territories under consideration from the academic literature, government agencies, non-government and intergovernmental organisations' reports, and from the World Database on Protected Areas. I captured information on IUCN category, size and age.

To calculate threat levels to marine resources and MPA effectiveness, I undertook a region-specific re-analysis of global spatial data from Burke et al. (2011)'s *Reefs at Risk Revisited,* thus confining ourselves to coral reef habitats. I extracted data on reef extent, reef threats, MPA extent and MPA effectiveness for each nation (one province for South Africa) in the WIO. The MPA effectiveness data for the Western Indian Ocean in Burke et al. (2011) cover 59 coral reef MPAs, 66% of the MPAs I documented in this analysis. Because effectiveness was determined through a rapid review using scores from regional experts rather than from field practitioners, there may be a sampling bias toward better-known sites, with a potentially higher proportion of ecologically effective sites than would be found overall (*ibid.*). Where one or more of the authors of this study had in-depth and more recent experience of one of the sites, these ratings were updated where necessary (n=6) to give a more accurate picture. All spatial modelling was performed with the ArcGIS™ 10.1 Geographic Information System software, and the ArcGIS™ Spatial Analyst extension.

To assess contributions towards international biodiversity commitments, I measured the progress of each country, except France, towards achieving the Convention on Biological Diversity's (CBD) target of 10% of marine and coastal ecological regions to be effectively conserved by 2020 (CBD, 2010). Progress was assessed by calculating the percentage coverage by MPAs and LMMAs of the continental shelf to 200m depth after Wells et al. (2007).

The list of level 3 and 4 sites, the LMMAs, was based on information gathered from LMMA workshops at the Seventh WIOMSA (Western Indian Ocean Marine Science Association) Scientific Symposium, held in Mombasa, Kenya in October 2011 and the Madagascar LMMA Forum, held in June 2012 in Madagascar. The initial workshop outputs were supplemented by an extensive electronic search for published literature using several electronic databases (including Web of Science, ScienceDirect, EconLit, WorldFish Library Catalog and CAB Direct). To capture potential LMMA sites documented in grey literature, I used Google.com and Google Scholar and examined the first 100 hits from each of the searches. Criteria for inclusion of an LMMA in the final list were that its area under management had been formalised through some form of legislation, usually a by-law. I combined the outputs of the workshop and literature searches to create the inventory of a total of 136 MMAs in the Western

Indian Ocean and liaised with key individuals and officials to determine or verify locations, sizes and governance structures of these areas.

5.5 Results & Discussion

5.5.1 MPAs in the Western Indian Ocean

Formally recognised MPAs have been around in the WIO since 1965, when the Ilhas da Inhaca e dos Portugueses Faunal Reserve (now part of the Ponta do Ouro Partial Marine Reserve) was gazetted in Mozambique (UNEP-WCMC, 2010). Both Madagascar and Kenya followed suit over the next three years, establishing Nosy Tanikely, the Malindi and Watumu Marine National Parks and the Malindi-Watumu Marine National Reserve (UNEP-WCMC, 2010). Today, all WIO countries have gazetted MPAs, except for Somalia, where the lack of a central administration has made it very difficult to practice conservation (Barrow et al., 2007; IUCN, 2000)

The 74 MPAs identified have a total coverage of 133,273km^2. Early protected areas tended to be smaller than their contemporary counterparts and were often designed to protect a specific habitat such as a turtle-nesting beach (Wells et al., 2007). Indeed, the smallest MPA in the WIO, the 0.01km^2 Cousin Island Special Reserve in the Seychelles dates from 1975. Table 5.2 shows how the average size of protected areas in the region increased by a factor of six from 49.1km^2 between 1965-1974 to 292.4km^2 between 1995-2004 as the emphasis shifted to larger, zoned multiple-use sites such as the Quirimbas and Bazaruto Archipelago National Parks in Mozambique. The two largest MPAs in the region are also among the newest: the Marine Park of the Glorieuses, designated in February 2012, and the neighbouring Marine Park of Mayotte, designated two years previously (Personal communication, Pascale Chabanet). The combined area of these new parks alone is more than 100,000km^2, constituting more than 84% of the total MPA coverage in the WIO region. This trend towards larger marine protected areas is continuing, with the newly designated Primeiras and Segundas MPA in Mozambique covering over 10,000km^2 (MPA News, 2012). Although declarations of large MPAs represent a step forward in marine biodiversity conservation, knowledge of the effectiveness of these areas is needed to properly assess achievements in addressing the Aichi CBD targets.

TABLE 5.2 MEAN AREA OF MPAS IN THE WESTERN INDIAN OCEAN AND NUMBERS EXISTING BY DECADAL CREATION DATE

Year created	Number of MPAs created	Average size (km^2)
1965-1974	7	49.1
1975-1984	16	36.5
1985-1994	10	25.3
1995-2004	26	292.4
2005-2014	15	8920.8

5.5.2 Effectiveness of protection of MPAs

Even with the large increase in coverage afforded by the Mayotte and Glorieuses reserves, the 74 MPAs (outlined in Table B1) protect just 7.3% of the continental shelf in the region (Table 5.3).

Seventy six percent of reefs in the WIO are at risk from local threats, with half (49.7%) rated at high or very high risk. The problem is most acute in the mainland coasts of Somalia and Tanzania and the islands of Réunion and Comoros, where more than 90% of reefs are threatened. The largest single threat is overfishing, which affects 72% of coral reefs, particularly on the densely populated coastlines of southern Kenya and Tanzania (Table 5.3). Watershed-based pollution is a problem in places like Madagascar, where widespread deforestation has caused extensive erosion and siltation in coastal areas (Burke et al., 2011). Dynamite fishing is also an issue, primarily on the mainland Tanzania coast where it has occurred for decades (Samoilys and Obura, 2011; Wells, 2009), but it also occurs in Mohéli Marine Park in Comoros (MS pers. obs. 2010).

Only 29.6% of reef-related MPAs assessed in the region were found to be effective. Whilst the effectiveness of MPAs within the WIO appears to range from 0-100% effective, it is important to recognise that artefacts occur at both extremes. For example, Madagascar's MPAs receive a 100% rating because only one of the seven MPAs was actually appraised, while Mayotte's score of 0% predates the establishment of the new marine park in 2010 (Personal communication, Pascale Chabanet).

5.5.3 LMMAs in the Western Indian Ocean

Four of the eleven nations under consideration have active LMMA projects (Table B2). Unlike the region's legislated MPAs, the WIO's LMMAs are newer endeavours. Of the

62 sites identified, 60 (96.8%) were established after the year 2000, in line with the passing of legislation to decentralise marine resources management in Kenya, Tanzania, Mozambique and Madagascar.

TABLE 5.3 EXTENT OF MPA COVERAGE AND EFFECTIVENESS FOR CORAL REEF MPAS IN THE WESTERN INDIAN OCEAN.

Region/ Country	Continental shelf (km^2) [1]	MPAs (km^2) in shelf area	MPA coverage (%) [2]	Effective MPAs (%)[3]	Reef Area (km^2)	Reef area % of global	Overfished reefs (%)[4]
Global	24,285,959.0			15.0	249,713.0	100.0	55.0
Indian Ocean				29.0	31,543.0	13.0	60.0
Comoros	1,415.7	404.0	28.5	0.0	399.0	0.2	100.0
Kenya	8,460.1	835.3	9.9	55.6	620.0	0.2	93.8
Madagascar	96,652.8	2603.3	2.7	100.0	3,934.0	1.6	86.9
Mauritius	27,372.8	139.2	0.5	28.6	976.3	0.4	62.6
Mayotte	1,100.0	1,100.0	100.0	0.0	643.8	0.3	78.1
Mozambique	73,299.7	14,551.3	19.9	0.0	2,435.5	1.0	78.5
Réunion	965.1	35.0	3.6	0.0	27.5	0.0	100.0
Seychelles	31,478.7	201.8	0.6	30.0	1,904.3	0.8	9.6
Somalia	40,391.7	0.0	0.0	NA	546.5	0.2	100.0
South Africa[6]	16,093.8	960.9	6.0	50.0	5.0	0.0	100.0
Tanzania [7]	8,951.6	1,160.8	13.0	0.0	2,089.0	0.8	94.7
Tanzania – Zanzibar	8,951.6	1,000.5	11.2	25.0	922.0	0.4	99.7
WIO*	**315,133.6**	**22,185.8**	**7.0**	**29.6**	**14,502.8**	**5.8**	**76.4**

1. Source: World Resources Institute (2007)
2. Percentage of continental shelf within marine protected areas
3. Percentage of coral reef MPAs judged to be effective in a rapid appraisal by regional experts. Developed from Burke et al. (2011). Both the Burke analysis and this one use the same definition of an MPA.
4. Percentage of reefs at medium or high risk from overfishing and destructive fishing. Developed from Burke et al. (2011).
5. At 68,332km^2, Mayotte Marine Park is much larger than its continental shelf area. Until 31 March 2011, Mayotte was not an overseas department of France and separate figures for continental shelf were not attainable. The figure used as a conservative proxy is the size of its lagoon (INSEE, 2010).
6. South Africa only includes KwaZulu Natal, the Province that borders the WIO. As continental shelf values for the province weren't available, a conservative proxy of 10% of South Africa's total shelf area was used.
7. Figures for continental shelf of the mainland and Zanzibar were unavailable, so an arbitrary split of 50% of Tanzania's total shelf was used.

* WIO values are totals

The 62 LMMAs for which I obtained reliable estimates of size amount to 11,329 km^2 in total. These varied across four orders of magnitude: the largest, at 1,966.7 km^2, is Madagascar's Ankivonjy; the smallest, at 0.118km^2, Mkwakwani/Tradewinds in Kenya. Mean LMMA size was 183km^2, with a quarter of sites smaller than 2.12km^2 and with a median size of 20.75km^2. However, these figures obscure important differences between countries, highlighted in Table 5.4. For example, Kenya's 14 nascent LMMAs protect a total of 109.6km^2 of marine resource, 37 times less than the 4,096.5km^2 under management in Tanzania and 61 times less than Madagascar's LMMA coverage of 6,635.3km^2.

5.5.4 Combined coverage and progress towards international targets

In the Western Indian Ocean, MMAs cover a combined 34,321.4 km^2 of the continental shelf (10.9% –Table 5.4). Assuming that percentage of the shelf is an acceptable proxy for the Aichi Biodiversity Target 11 target of 10% of marine and coastal areas protected by 2020, then Comoros, Kenya, and Tanzania have already achieved the target (with respective figures of 28.5%, 11.2% and 37.6%). Mozambique (19.9%) has also achieved the target, primarily due to recent designation of Primeiras and Segundas MPA, whilst Madagascar is on course to do so, should the Barren Isles LMMA be established as scheduled in 2014. In total, LMMAs in the WIO cover 11,329.4km^2, 3.6% of the region's continental shelf. The differences between LMMA and MPA coverage are particularly pronounced in mainland Tanzania and Madagascar, where LMMAs cover 3.5 and 2.6 times more area than MPAs, respectively.

5.5.5 Locally Managed Marine Areas in the WIO: legal context

Marine Areas exist primarily in Kenya, Madagascar, Mozambique and Tanzania. To determine potential for future expansion, here I assess the co-management systems in these four countries by exploring legal context, quantifying successes and identifying barriers to replication (Table 5.5).

TABLE 5.4 EXTENT OF MPA (LEVEL 1 AND 2) AND LMMA (LEVEL 3 AND 4) COVERAGE AND CURRENT PROGRESS TOWARDS ACHIEVING INTERNATIONAL BIODIVERSITY CONSERVATION TARGETS

Country*	No. of MPAs	MPA coverage (%) [1]	No of LMMAs	LMMA av. Size (km^2)	LMMA coverage (%) [2]	LMMA + MPA coverage (%) [3]
Comoros	1	28.5	0	NA	0.0	28.5
Kenya	9	9.9	14	7.83	1.3	11.2
Madagascar	8	2.7	34	195.16	6.9	9.6
Mauritius	13	0.5	0	NA	0.0	0.5
Mayotte	1	100.0	0		0.0	100.0
Mozambique	6	19.9	1	18	0.0	19.9
Réunion	1	3.6	0	NA	0.0	3.6
Seychelles	14	0.6	0	NA	0.0	0.6
Somalia	0	0.0	0	NA	0.0	0.0
South Africa [4]	4	6.0	0	NA	0.0	6.0
Tanzania	10	13.0	12	341.38	45.8	58.7
Tanzania -- Zanzibar	3	11.2	1	470	5.3	16.4
WIO**	**69**	**7.0**	**62**	**182.73**	**3.8**	**10.9**

1. Percentage of continental shelf within marine protected areas (level 1 and 2)
2. Percentage of continental shelf within locally managed marine areas (level 3 and 4)
3. Percentage of continental shelf within marine protected areas and locally managed marine areas
4. South Africa only includes KwaZulu Natal, the Province that borders the WIO

* Excludes Îles Éparses

** All are regional totals, except LMMA av. Size (km^2) which is a mean

5.5.5.1 Kenya

Before the recent introduction of Beach Management Units (BMUs) under Legal Notice 402 of the Fisheries Act, marine resource management took a centralised, top-down approach in Kenya (Government of Kenya, 2007). BMUs are a co-management tool for small-scale fisheries initially developed to improve inland fisheries on Lake Victoria (Signa et al., 2008). A BMU is an association of fishers, fish traders/mongers, boat owners, fish processors and other fishery stakeholders centred on a coastal landing site and formally led by an executive committee of stakeholders (Government of Kenya, 2007; Lamprey and Murage, 2011). With the support and permission of officials from the Department of Fisheries, these BMUs are able to devise and enforce by-laws to govern their fishery, allowing them to delineate its boundaries and for example, exclude non-registered fishers or boats from the area (Government of Kenya, 2007; Nelson, 2012). There are presently around 60 BMUs along the Kenyan coastline (Lamprey and Murage, 2011). Many of these exist only in name, and are yet to formalise their areas of jurisdiction or develop their by-laws (Lamprey and Murage, 2011; Nelson, 2012).

The 2007 BMU Regulations provide a legislative framework to establish Locally Managed Marine Areas in Kenya. Kenya's first LMMA dates from 2006, when the community of Kuruwitu on the central Kenyan coast established the Kuruwitu Community Managed Conservation Area (Griffin, 2012; Maina et al., 2011). Through a local umbrella organisation, the Kuruwitu Community Welfare Association, residents designated a 0.29km^2 no-take zone (Maina et al., 2011; Nelson, 2012). Since establishment, live hard coral cover within the LMMA has increased by an estimated 30%, whilst fish numbers have grown by 200% (Nelson, 2012).

Predating the 2007 BMU Regulations, Kuruwitu lacked legislative support and depended on acceptance from nearby communities and support from the East African Wildlife Society (Maina et al., 2011; Murage, 2012). Nonetheless, the reserve's success has attracted interest from other fishing communities along the Kenyan coast, and it is likely that Kuruwitu, along with community exchange visits to the Collaborative Management Areas in Tanga, northern Tanzania, helped to catalyse the development of the 2007 BMU regulations and the designation of further LMMAs (Maina et al., 2011; Nelson, 2012; Samoilys and Obura, 2011). At present, there are 14 operational LMMAs in Kenya, covering 110km^2 (Table B2).

This rapid increase in LMMA numbers suggests that local communities perceive them to be beneficial, and has occurred in spite of a lengthy development process involving consultations, community surveys, mapping and management plan creation (Abunge, 2011). However, several issues remain, including insufficient capacity for effective monitoring, control and surveillance in local communities, lack of funding and alternative livelihoods, conflicts of interest between stakeholders, legislative overlap and conflicting mandates, and a poor understanding among local communities of the legal procedures involved in designating an LMMA (Anderson, 2012; Maina et al., 2011; Samoilys et al., 2011). A further challenge relates to lack of land ownership. In Kuruwitu, for example, the Association hopes to establish a community-run eco-lodge to accommodate visiting tourists, but has so far found it impossible to obtain rights to coastal land on which to construct it (Maina et al., 2011; Nelson, 2012).

TABLE 5.5 KEY FEATURES OF LMMA INITIATIVES IN THE WESTERN INDIAN OCEAN

Country	Formal LMMAs	LMMA success[1]	LMMA potential[2]	Key local-level institutions	Key enabling legislation	Local name for LMMAs
Comoros	No	-	Low-Medium	Village fishing associations	-	-
Kenya	Yes	Medium	High	Beach Management Units (BMUs)	Beach Management Unit Regulations 2007	Community Conservation Areas, *tengefu*, Local Marine Management Areas (also LMMAs)
Madagascar	Yes	High	High	Village and multi-village level fishing associations. Village councils (Fokontany), Communes	Gestion Locale Sécurisée (GELOSE), *dina,* Décret d'Application No 848-05	LMMA, Community Managed Protected Area
Mauritius	No	-	Low	-	-	-
Mayotte	No	-	Low	-	-	-
Mozambique	Yes	Low	Medium	Fishing Community Councils (CCPs – Conselho Comunitário de Pescas) and Co-management Committees (CCG – Comité de Co-Gestão)	2003 Regulation on Marine Fisheries	-
Réunion	No	-	Low	-	-	-
Seychelles	No	-	Medium-low	Praslin Fishers Association	-	-
Somalia	No	-	Low	-	-	-
South Africa	No	-	Medium-high	Local Subsistence Co-Management Committees	Policy for the Small Scale Fisheries Sector in South Africa	Small Scale Fishing Community Area *
Tanzania	Yes	High	High	Beach Management Units (BMUs)	2003 Fisheries Act and its principal Regulations of 2009	Collaborative Management areas, Collaborative Fisheries Management Areas (CFMAs)
Tanzania -- Zanzibar	Yes	Medium	Medium-low	Village Fisheries Committee (VFC), Village Conservation Committee (VCC)	Environmental Management for Sustainable Development Act 1996, The Marine Conservation Unit Regulations **	-

1. Level of success in establishing LMMAs to date.
2. Potential to establish more LMMAs in future.

* Forthcoming. Communities will be able to apply several control measures within this area, including quotas and gear restrictions, as well as closed seasons and areas.

** In draft, awaiting finalisation.

Recent reforms to land policy offer some hope (Samoilys et al., 2011). If these can be combined with guidelines to standardise LMMA establishment, as well as with education and awareness programmes at the local level, then Kenya will be well-placed to lead the LMMA revolution along the coasts of mainland East Africa (Maina et al., 2011).

5.5.5.2 Tanzania – mainland

In mainland Tanzania, co-management of marine resources dates back to the mid 1990s, when a collaborative approach was initiated in the coastal waters of the Tanga region (Verheij et al., 2004). The Tanga Coastal Zone Conservation and Development Programme operated with donor funding from 1994 until 2005 and has continued since as the Tanga Coastal Zone Resources Center, a District and Regional government initiative (Samoilys and Kanyange, 2008). Under this Programme six collaborative management areas were established between 1997 and 2001 covering a total of 1,604km^2 (Table B2). They have legal recognition in the form of a by-law, as well as formal endorsement from the Director of Fisheries (Samoilys and Obura, 2011; S. Wells et al., 2007a).

Each of the Tanga LMMAs has a no-take-zone collaboratively policed by fisheries officers and local communities. Ecological monitoring since 1999 showed that these closures – the first on the East African coast to be established and actively managed by local fishing communities – had higher densities of fish and invertebrates, leading to positive impacts on local livelihoods, at least until 2004 (Samoilys et al., 2007a; S. Wells et al., 2007b). Since then, dynamite fishing, which was almost completely eradicated in the region between 1998 and 2004, has returned (Samoilys and Obura, 2011).

Mainland Tanzania also has coastal Beach Management Units, established by the 2003 Fisheries Act and its principal Regulations of 2009 (United Republic of Tanzania, 2009, 2003). As in Kenya, BMUs empower communities to manage local fisheries resources, giving them the rights to restrict certain gears and control access through licencing (Mulyila et al., 2012). Co-management of fisheries resources has spread rapidly in Tanzania, and there are currently 179 coastal BMUS, of which 68 have management plans and 39 have legal recognition through by-laws (Otsyina et al., 2010; Sobo, 2012).

BMUs are increasingly establishing Collaborative Fisheries Management Areas (CFMAs) as a higher-level mechanism to manage their shared resources (Otsyina et al., 2010). CFMAs, a type of LMMA, can protect the fishing grounds of an individual BMU or, more commonly, the shared resources of several (Fisheries Development Division and WWF, 2009). Typically, BMUs with legal recognition first consult with neighbouring Units to determine the boundaries of the CFMA, before establishing a Co-ordination Committee. The Committee, which is composed of representatives from each BMU within the proposed area, synthesises management proposals from individual Units into a draft CFMA management plan, and acts as a networking mechanism for BMUs (Fisheries Development Division and WWF, 2009; Mwangamilo, 2012). Once the draft plan is approved by each participating BMU, the CFMA can be given legal recognition in the form of a District Council by-law. At present, there are six CFMAs in mainland Tanzania, all established with the assistance of the WWF as part of a programme in the Rufiji, Mafia and Kilwa Districts of central Tanzania (Anderson, 2012; Sobo, 2012). The six areas cover a total 2,498 km^2 across 21 BMUs, 2.5% of which (61.2km^2) has no-take protection, initially for a 2-year period (Mwangamilo, 2012). The programme is so far showing some promise: incidences of illegal dynamite fishing and seine netting have decreased, and there is a perception among resource users that fish abundance is starting to recover (Tanzania Natural Resource Forum, 2012).

5.5.5.3 Tanzania – Zanzibar

BMUs (and CFMAs) do not exist in Zanzibar, a semi-autonomous region of the United Republic of Tanzania with separate environmental law and policy. The most comparable institution is the Village Fisheries Committee (VFC), a local-level organisation comprising 10 elected members (Anderson, 2012). VFCs were formed in all coastal fishing villages when devolution of marine resource management began in 1994 (Cinner et al., 2012b). VFC jurisdiction depends on village boundaries and distance covered by local fishers. Responsibility for enforcement of regulations is shared between the Committee and the Department of Fisheries and VFCs can draw-up by-laws to manage resource use (Cinner et al., 2012b; Mwaipopo, 2008).

In Zanzibar, the legal basis for MPA establishment is provided by the Environmental Management for Sustainable Development Act 1996 (Government of Zanzibar, 2007). The Act recognises the need to involve local communities in MPAs, enabling co-management arrangements analogous to LMMAs to develop at Misali Island and Menai Bay, albeit with a degree of State oversight (Levine, 2007; Mwaipopo, 2008; Wells et al., 2007).

In recent years, however, it appears that conservation of marine resources has become more centralised, less transparent and less participatory (McLean et al., 2012). For example, Misali Island, a community-initiated attempt to resist tourism development, was subsumed into the larger Pemba Channel Conservation Area in 2005, with a consequential decline in community involvement (Levine, 2007; Lindhjem, 2003; McLean et al., 2012; WWF, 2004). Further, whilst the forthcoming Marine Conservation Unit regulations (Government of Zanzibar, 2012) include provisions to promote community involvement in marine resource management, the overall approach is top-down in nature (McLean et al., 2012). These regulations will need a degree of revision if local management is to flourish in Zanzibar.

More broadly, both mainland Tanzania's BMUs and Zanzibar's VFCs lack capacity in many crucial areas including conflict resolution, financial management, project planning and marine ecology (Anderson, 2012; Sobo, 2012; Tanzania Natural Resource Forum, 2012). There are clear cultural, legal, political and institutional similarities between Kenya and Tanzania, so the efforts that are presently underway in both countries to address capacity constraints and promote sustainable financing would likely benefit from sharing experiences and resources.

5.5.5.4 Madagascar

One of the traditional values recovered following Madagascar's independence in 1960 was the social code. In rural communities, this social code – known as the *dina* – is a community law, generally communicated through oral tradition, though written down in some cases (Rakotoson and Tanner, 2006). In 1996, the Malagasy Government introduced the Gestion Locale Sécurisée (GELOSE), a legal framework designed to integrate the *dina* with governmental laws to enable community-based management of natural resources (Rakotoson and Tanner, 2006).

Seven years later, at the fifth World Parks Congress in Durban, South Africa, the Malagasy president recognised the need to protect the country's unique natural assets and committed to the Durban Vision, a national conservation plan to triple the amount of protected area coverage (Durbin, 2007). This was codified into law shortly afterwards as a new decree (Décret d'Application No 848-05) for the existing Code des Aires Protégées (Durbin, 2007). The decree set up a System of Protected Areas of Madagascar, which simplified and redefined the legal process used in protected area creation (IRIN, 2006). Under this more flexible model, community organisations, NGOs and the private sector are permitted to manage protected areas, in addition to the parastatal protected areas agency Madagascar National Parks (Rabearivony et al., 2010). Since then, several LMMAs have been established along the coast by NGOs working with local communities (Table B2).

The Velondriake Community Managed Protected Area in southwest Madagascar is the country's oldest LMMA (Harris, 2007). Velondriake spans nearly 1,000 km^2 of coral reefs, mangroves, lagoons, beaches and sea grass beds, making it one of the largest marine managed areas in Madagascar (Harris, 2007; Westerman and Gardner, 2013). Home to around 7,500 semi-nomadic Vezo, Velondriake unites 25 coastal villages in the co-management of local marine resources (Harris, 2011; Westerman and Gardner, 2013). It is legally recognised as an IUCN category V MPA and was granted definitive protected status by inter-ministerial decree in late 2012 (Westerman and Gardner, 2013). Velondriake began as an initiative to improve the sustainability of the octopus fishery, but had since expanded to include aquaculture, temporary closures and the designation of eight permanent no-take marine reserves totalling 0.8km^2 (Harris, 2011; Westerman and Gardner, 2013).

The initiative is largely guided and managed by local communities, with technical and financial support provided by the British NGO Blue Ventures. Resource use and access rights within the area are governed by a legally recognised *dina* rather than the GELOSE framework (Andriamalala and Gardner, 2010). The *dina* bans destructive fishing practices including beach seining and poison fishing, regulates temporary and permanent closures and grants conflict resolution and enforcement powers to local communities, allowing them to impose fines and utilise the regional court system in cases where conflict resolution is unsuccessful (Harris, 2011; Westerman and Gardner, 2013).

Velondriake's perceived success has triggered widespread replication of the LMMA approach. Over the last 7 years, 34 LMMAs have been established along Madagascar's northern, western and southern coasts. Taken together, these initiatives presently cover 6.9% of the seabed, 6,635.3km^2. In 2014, the Barren Islands is expected to add a further 4,290km^2 to the total. This scaling up is unparalleled in the Western Indian Ocean, yet it has been achieved at low cost, without financial support from central government (Harris, 2011). With severe constraints continuing to inhibit the country's capacity for environmental governance, Madagascar's LMMAs may offer an encouraging and locally acceptable solution to the challenges of marine resource management (Harris, 2011).

5.5.5.5 Mozambique

In Mozambique, the concept of fisheries co-management is enshrined in the 2003 Regulation on Marine Fisheries, which establishes co-management institutions at the provincial, district and local levels, as well as banning non-artisanal fisheries within three nautical miles of the coast (Government of Mozambique, 2003). The Decree introduces two types of institution: Fishing Community Councils (CCPs – Conselho Comunitário de Pescas) and Co-management Committees (CCG – Comité de Co-Gestão) (Government of Mozambique, 2003). Both were later formally established through legislation adopted in 2007 (Government of Mozambique, 2007; Swennenhuis, 2011).

CCGs are multi-stakeholder committees formed principally at the provincial or district levels, but also at the local level (Government of Mozambique, 2007; Russo de Sá, 2011). Their principal objectives include deciding closed seasons and permissible types of gear and protecting endangered marine resources as well as advising on conflict resolution among fishers, fishing licences and fee collection (Government of Mozambique, 2003). CCPs are community-based associations of elected community members involved in artisanal fisheries (Government of Mozambique, 2007). Analogous to Kenya and Tanzania's BMUs, CCPs give local stakeholders rights to establish boundaries, control access and promote the sustainable use of marine resources (Government of Mozambique, 2003; Rosendo et al., 2011). Once members have been elected and the CCP established, they can apply for formal legal recognition, which, if granted, empowers them to assume responsibility for fishing licences and enforcement, functions otherwise administered at the district level (Government of Mozambique, 2007; IFAD, 2012).

CCP adoption has largely been driven through several donor-led artisanal fisheries programmes, especially in the Sofala Bank area, where the National Institute for the Development of Small-Scale Fisheries and the International Fund for Agricultural Development (IFAD) have worked with local communities to establish 65 Councils along a 950-kilometre stretch of coastline fronting the provinces of Sofala, Zambezia and Nampula (IFAD, 2012, 2010). Largely as a result of these initiatives, there are presently at least 156 CCPs along the coastline of Mozambique (National Institute for the Development of Small Scale Fisheries, 2012). Taken together, it is estimated that these Councils provide a degree of representation to almost all coastal fishing communities in the country (MRAG, 2010). However, very few existing councils (approx. 20) are officially recognised by the Ministry of Fisheries (IFAD, 2012, 2010), whilst many are unaware of their rights and responsibilities, and frequently lack the human resources, technical capacity and financial support necessary for effective management and enforcement (Swennenhuis, 2011; Wilson, 2012). In a study of compliance with centrally declared fisheries controls in the Sofala Bank area, for example, Wilson (2012) found that fewer than 10% of inspected nets complied with minimum mesh size regulations, whilst none of the fishers interviewed ceased fishing or reduced effort during the closed season.

Similar technical and financial constraints have also plagued the higher level Co-management Committees. Outside of areas where they have direct support, very few CCGs are functioning, and of those that are, even fewer are endowed with the resources and awareness of community-level rights and obligations they need (Russo de Sá, 2011; Swennenhuis, 2011). So far, there has been little monitoring of CCG operations at the central level and mechanisms for co-ordination across all levels of governance are practically non-existent (Russo de Sá, 2011).

As a result of these issues, this study was only able to identify one example of a functioning LMMA in Mozambique: the Vamizi Marine Sanctuary. This no-take reserve is managed by the Vamizi CCP, with technical and financial support from a partnership between an eco-lodge on Vamizi Island and WWF (Garnier et al., 2008). The partnership is helping to build community capacity for effective marine resource management by training members in reef and fish monitoring, developing alternative livelihood projects and providing environmental education (Garnier et al., 2008). The Council has delineated the boundaries of the Sanctuary with marker buoys and receives a $3 fee for each dive boat that enters (Personal Communication, Isabel da Silva).

Vamizi's success suggests if capacity constraints were addressed, more LMMAs could be established. Closed areas have already been trialled in other CCPs, and research suggests that these initiatives, together with restrictions on gear, length and species, may enjoy broad community support in Mozambique, particularly in the Pemba region (McClanahan et al., 2013, 2012). To this end, a new phase of the IFAD-financed ProPESCA initiative is aiming to strengthen the capacity of the CCPs and CCGs, whilst the IUCN is considering an intervention to support the creation of community no-take zones (IFAD, 2012; Swennenhuis, 2011).

5.6 Conclusions

This analysis shows that although MPAs in the Western Indian Ocean cover 133,273km^2, only 7.0% of the region's continental shelf is protected by them. Less than 30% of reef-related MPAs in the WIO were found to offer effective ecological protection, though this compares favourably with global figures, where 15% were graded effective (Burke et al., 2011).

I found active LMMA initiatives in four countries that have passed legislation to decentralise marine resource management, with good potential for scaling up these initiatives in Kenya, mainland Tanzania and Madagascar, and lower potential in Zanzibar and Mozambique (Table 5.5). Due to underdeveloped legal structures supportive of local management in the other seven countries, no other formal LMMA initiatives were found to be in place, though the Seychelles (Clifton et al., 2012; UNDP, 2010), Comoros (Hauzer et al., 2013) and especially South Africa (Government of South Africa, 2012) have all acknowledged the potential of devolved management.

At 11,329.4km^2, LMMAs cover 3.6% of the region's continental shelf, with particularly pronounced differences between LMMA and MPA coverage in mainland Tanzania and Madagascar, where LMMAs cover 3.5 and 2.6 times more area than MPAs respectively. Assuming that percentage of continental shelf covered by LMMAs and MPAs is an acceptable proxy for the CBD's 10% 2020 coverage targets, Comoros, Kenya, Mozambique and Tanzania have already achieved the target, whilst Madagascar is on course to do so.

Three caveats apply here. First, LMMA data were based on outputs from a workshop at the Seventh WIOMSA Scientific Symposium in Kenya in October 2011 and the Madagascar LMMA Forum in June 2012 (Blue Ventures, 2012) in addition to information gathered from a search of the published and grey literature and first-hand knowledge (MS, SP). Because many LMMAs, especially in Kenya and Mozambique, are small-scale, informal arrangements, information about them can be difficult to source. Accordingly, calculations of LMMA coverage should be considered as conservative estimates.

Secondly, the CBD 2020 targets call for 10% of the world's ecological regions to be effectively conserved. For the same reasons that the coverage estimates are conservative, the extent to which the region's LMMAs can be considered effective conservers of resources is largely unknown. Globally, empirical evidence that co-management arrangements achieve ecological (Cinner et al., 2012b; McClanahan et al.,

2008) and social (Evans et al., 2011; Syakur et al., 2012; Wamukota et al., 2012) goals is scarce and inconclusive, though this is likely to be due in part to the recently established nature of many LMMA initiatives. For example, in an examination of 42 marine co-management systems in Kenya, Tanzania, Madagascar, Indonesia and Papua New Guinea, Cinner et al. (2012c) found that although 88% of resource users surveyed reported high levels of compliance and 54% perceived a benefit to their livelihoods, co-management could also create a degree of social inequality by favouring wealthier users. They additionally found that fish biomass in co-managed areas was generally greater than in areas without local management, though substantially lower than in no-take marine reserves in the same countries (Cinner et al., 2012c).

Thirdly, although this analysis suggests that the LMMAs are increasingly numerous and could help WIO nations to meet international biodiversity commitments, I am not necessarily advocating for physically larger LMMAs with greater institutional complexity. Although the perceived success of the Velondriake Community Managed Protected Area in Madagascar provides evidence that large LMMAs with multiple stakeholders can function effectively, (Harris, 2011; Westerman and Gardner, 2013) other studies have reached differing conclusions. In an examination of ten fisheries cooperatives in Mexico, for example, McCay et al (2014) found that few stakeholders and a small spatial scale were critical success factors in community-based management of the commons.

Evidence from a recent global review of Territorial Use Rights for Fisheries (TURFs) by Auriemma et al (2014) is more equivocal. LMMAs and TURFs, the latter defined by the study as "an area in which individuals or communities are given some level of exclusive access to marine resources within a defined boundary", overlap in many key areas because managed access rights are often implemented by communities within LMMAs (Auriemma et al., 2014). The analysis, which drew on 103 case studies in 29 countries to test, among others, the hypothesis that larger TURFs are less successful due to increasing difficulties of enforcement, found no overall effect (Auriemma et al., 2014). The question of whether bigger LMMAs are better is thus largely unresolved and would benefit from additional research.

From the country-specific analysis of LMMA implementation I present here, it is clear that a lack of organisational capacity, skills and money can all compromise the effectiveness of locally managed marine areas. And where areas are successful, further challenges may arise. For example, as biomass and fish numbers increase, so too may poaching, leading to further strain on resources devoted to enforcement (Aalbersberg et al., 2005; Guilbeaux et al., 2008).

Over the short-term, these issues may be best addressed through the establishment of an information-exchange forum to enable LMMA practitioners to share experiences and best practice, to offer training and exchange visits, and to promote local management to other communities and governments, especially in countries that have yet to devolve marine resource management to the local level. The forum could be modelled on the Pacific LMMA network and be complemented by research initiatives to better understand under what circumstances LMMAs may achieve their social and ecological objectives. Over the longer-term, the forum could form the basis for scaling-up LMMAs in the region towards a network that is lasting, effective and representative, and one that is complementary to centralised systematic conservation efforts.

6 General Discussion

Throughout the tropics and subtropics, increasing numbers of communities are taking responsibility for the management of coastal resources under arrangements known as "locally managed marine areas" or LMMAs (Govan, 2009; Rocliffe et al., 2014, Chapter 5). Broadly analogous to MPAs, LMMAs are managed for sustainable, long-term use rather than biodiversity conservation itself, may incorporate secure tenure rights for resource users, and typically employ a range of management techniques, including periodic closures, gear restrictions, species-specific reserves and permanently closed, fully protected areas to achieve diverse aims (Jupiter et al., 2014; Mills et al., 2011).

These locally centred interventions have emerged as a potential solution to many of the challenges of small-scale fisheries management commonly faced by coastal communities in developing countries, and have arisen chiefly because of disenchantment with top-down, centralised government approaches (Cohen and Foale, 2013; Govan et al., 2006; Wamukota et al., 2012). When effective, evidence suggests that LMMAs can encourage responsible fishing, strengthen compliance and may help to safeguard food security and increase resource abundance (Bartlett et al., 2009; Cinner et al., 2005; Gutierrez et al., 2011; Léopold et al., 2013; Levine and Richmond, 2014; Pollnac et al., 2001; Weiant and Aswani, 2006).

However, the relatively young and informal nature of many initiatives means that several questions remain unanswered. In particular, little is known about the long-term effectiveness of LMMAs at achieving their ecological and social objectives, with initiatives often lacking financial resources to sustain both effective management and the evaluation and collection of robust evidence on which such management frequently depends (Frid and Crowe, 2015; Gutierrez et al., 2011; Jupiter et al., 2014; Léopold et al., 2013; Levine and Richmond, 2014). Moreover, little is known about the status and scale of LMMAs outside the Western and Central Pacific region, where the majority of research attention has so far been focused (e.g. Govan, 2009).

This thesis combines social and ecological research to explore these key research gaps and to offer suggestions for improving the effectiveness of LMMAs. Using a cross-disciplinary approach that integrates methods from ecology, conservation science and socio-economics, as well as from medicine, psychology and business management, I suggest novel approaches for improving assessments of (Chapters 3 and 4) and support for (Chapters 2 and 3) LMMAs, and examine their status and extent in a region where they are rapidly emerging as tool of choice for local marine resource management and conservation (Chapter 5).

My findings expand our knowledge of LMMAs in the Western and Central Pacific and Western Indian Ocean and provide important baselines against which to evaluate future efforts to develop and implement management strategies in the Cook Islands and WIO. I highlight the importance of capturing visitor perceptions of marine resource management and protection efforts in tourism-dependent island states, and provide some evidence of the role coastal hotels can play in supporting and sustaining LMMAs in tropical developing countries. Finally, I demonstrate that both a common analytical flaw and a previously un-researched bias may confound results in evaluations of the ecological effectiveness of protected areas, underscoring the need to incorporate blind assessment and to more carefully consider potentially confounding variables in future research designs.

In this concluding chapter, I summarise and contextualise the findings of the preceding data chapters and make recommendations both for further research and for improving LMMAs. The limitations and implications of the research are discussed throughout and conclusions are drawn.

6.1 Sustaining LMMAs: a role for tourism?

Since many LMMAs lack the necessary financial resources to sustain effective management and a robust evidence base, yet exist in tropical developing countries with high tourist visitation, Chapters 2 and 3 considered the interplay both between tourism and LMMAs from both a social and an ecological standpoint. Both were based on primary field research in the tourism-dependent Cook Islands, which relies on its marine life and beaches to sustain a sector with an estimated total contribution to GDP of between 75 and 90% (pers. comm. Metua Vaiimene). Taken together, these findings underscore the importance of gathering robust visitor perception data in tourism-centred island states, provide a baseline against which to evaluate the effectiveness of future management strategies and initiatives in the Cook Islands, and offer cautious backing for the use of coastal hotels as a mechanism for LMMA funding and support.

In Chapter 2, I surveyed 468 visitors to the Cook Islands with respondent completion questionnaires to investigate awareness of and support for marine conservation initiatives in the Islands and to quantitatively examine satisfaction with the marine environment. My findings suggest substantial visitor disappointment with the marine environment on the main island of Rarotonga, with around 40% of those surveyed expressing dissatisfaction with fish diversity, abundance and size, and around half not satisfied with coral diversity or cover. Forty-six percent of respondents were aware of the network of no-take LMMAs in the Cook Islands (known locally as *ra'ui*), but none could name all six of Rarotonga's sites. However, support for marine protection efforts was high, with 85.5% keen to see at least 30% of Cook Islands waters afforded full protection from fishing, and 93.6% scoring marine reserves as important or very important.

Building on these results, in Chapter 3, I used underwater visual census techniques to assess the ecological effectiveness of Rarotonga's LMMAs. A particular focus of the study was the two sites that are co-managed by hotels, as well as on the potential to scale this approach in other tropical developing countries. My results indicated that abundance and biomass of species targeted by fishers were 2.5 and 5.4 times higher, respectively, within the LMMAs than at the control at the Aroa hotel site, and 3.0 times and 4.3 times higher at the Edgewater hotel site. At both locations, the responses of higher trophic level species and overall fish community structure were also significantly different from controls. The sites without hotel involvement showed no clear differences in targeted abundance and biomass, community structure or higher trophic level responses. There is therefore evidence to suggest that hotel co-management can be effective at achieving the aims of these LMMAs.

Unlike the majority of the examples in the literature to date (E.g. Niesten and Gjertsen, 2010; Nordlund et al., 2013; Riedmiller, 2008b; Svensson, 2009; Svensson et al., 2010; Teh et al., 2008), the model of hotel involvement I explore in this chapter is collaborative and informal, and without significant cost overheads. The actors are coastal hotels with a desire to ensure that their guests can enjoy vibrant and diverse beachfront reefs, rather than ecoresorts with explicit environmental and social programmes. As such, it avoids many of the criticisms traditionally associated with hotel stewardship of marine resources including expense (Nordlund et al., 2013) "greenwashing" (Weaver and Lawton, 2007) and a lack of integration with broader coastal management approaches (Colwell, 1998; de Groot and Bush, 2010).

The moderately encouraging results shown by this study are underpinned by increasing adoption of responsible tourism practices by hotels (Goodwin, 2011; United Nations Environment Programme and World Tourism Organization, 2012) and growing numbers of LMMAs worldwide (Rocliffe et al 2014, Chapter 5) so it is likely that, in locations where these two trends intersect, there is an opportunity for LMMAs supported by hotels to flourish. These results also add to a growing base of evidence that tourism can provide a strong economic foundation to some LMMAs and can be a key driver in persuading local communities to participate in management (Cohen et al., 2014; Horowitz, 2008; Jupiter et al., 2014; Niesten and Gjertsen, 2010; Vianna et al., 2012; Weeks and Jupiter, 2013).

However, since not all LMMAs are established in areas where there is tourist infrastructure, it is conceivable that expansion of a hotel-supported model could be detrimental to more remote communities (Levine, 2007). Moreover, as more hotels partner with coastal communities to co-manage resources, it will be important to guard against the temptation to "ocean-grab", whereby a hotel could acquire exclusive use rights, depriving small-scale fishers from accessing the resources they need and negatively impacting food security (Bennett et al., 2015).

For the Cook Islands, the central recommendation that arises from both studies is reform legislation to give its LMMAs legal recognition and strengthened enforcement. The *ra'ui* have historically relied on chiefly authority, or *mana*, to prevent poaching (Miller, 2009; Miller et al., 2011), a system which has become less effective as chiefly power has waned on Rarotonga (pers. comm. B Ponia). Legislation has been drafted to give the *ra'ui* legal recognition (Government of the Cook Islands, Draft), but this is currently stalled amid uncertainties over the extent to which the enforcement and declaration of *ra'ui* areas is devolved to the local community.

While the specific mechanics of implementing the legislation fall outside the scope of this thesis, it is important to highlight that the Cook Islands is far from the only country to struggle in this regard. Many Pacific Island nations are legally pluralist, with more than one system of law operating concurrently (Techera, 2015; Vukikomoala and Jupiter, 2012). Developing appropriate legal mechanisms for marine resource management therefore requires a hybridisation of state-based legislation and customary law, which is a complicated and challenging undertaking (Techera, 2015). A variety of hybrid approaches have been adopted in the Pacific, but differing cultural, environmental and socio-political contexts mean that no dominant mechanism has emerged as of yet (Clarke and Jupiter, 2010; Govan, 2009; Mills et al., 2012; Techera, 2015; Vierros et al., 2010; Vukikomoala and Jupiter, 2012).

6.1.1 Research impact

The interdisciplinary research design employed in Chapters 2 and 3 was designed to be of maximum value to marine stakeholders in the Cook Islands. Together with NGO staff and community members, government officials with expertise in tourism, conservation and marine resource management were consulted during the first of two field seasons in the Cook Islands, and their responses used in part to generate and refine the research questions. The ecological dataset from Chapter 3 has already been shared with the Ministry of Marine Resources (MMR) (which has overall responsibility for the monitoring of the *ra'ui*), and forms a core part of their on-going monitoring efforts. Since the dataset represents best practice in data design, MMR staff have been trained how to use it effectively, helping to reduce the time spent on future monitoring, and to increase the robustness of outputs.

Furthermore, Chapters 2 and 3 are in the processing of being rewritten as policy briefs to be shared with relevant government ministries, tourism stakeholders and community representatives, as well as members of the committee responsible for the establishment and monitoring of the Cook Islands Marine Park. Preliminary results from Chapter 2 were shared in Feb 2014, and have already led to the implementation of improved signage for the *ra'ui*, a recommendation arising from the Chapter.

Finally, I am also intending to present my results to a general public audience on Rarotonga via video link in Summer 2016, and to lead an open Q&A on the future of the *ra'ui*.

6.1.2 Key recommendations

- Investigate opportunities for hotels to support and sustain LMMAs in other tropical developing countries.

- Improve compliance and reduce poaching by initiating an LMMA education and awareness programme targeting residents and visitors alike in the Cook Islands.
- Use research into visitor perceptions of marine resource management to help guide sustainable tourism development efforts, both in the Cook Islands and in other tourism-dependent island states.

6.2 Sustaining LMMAs: improving assessments

If the status of LMMAs is to be defended in a world where there are increasing pressures on marine resources, then arguments as to their effectiveness (or otherwise) need to be based on the soundest possible evidence. As Lawton (1996) has pointed out, evidence from research that involves experimental falsification of hypotheses tends to be much more persuasive to others than modelling studies, observations, logical argument and anecdote, which are increasingly less-sound. However, experimental falsification demands high standards of both data collection and experimental design if such evidence is to stand up to scrutiny. Achieving those high standards becomes more challenging the larger the spatial scale (extent) of focus (Raffaelli and Friedlander, 2012; Raffaelli and Moller, 2000), because there is usually a trade-off made between spatial scale and the degree of replication of experimental and control areas that can be achieved. In addition, rigorous analysis of such experiments presumes that the underlying data have been collected in an unbiased way. I have explored these issues in Chapters 3 and 4.

In Chapter 3, I used a long-established network of LMMAs in Rarotonga, Cook Islands as a model to examine how different research designs may lead to different management responses. I conducted three analyses to explore differences in the abundance and biomass of fish species targeted by fishers in LMMA sites and at paired controls. In the first, I used a spatially replicated design to assess the *overall* differences across all of the sites (the network-level analysis). In the second, I used an essentially unreplicated design to assess the differences between each of the *individual* LMMAs and their controls (the site-level analysis). In the third, I used an asymmetric design to examine the differences between each LMMA and multiple control sites. The network-level analysis found no effect of protection: there were no significant differences in overall fish community structure between LMMAs and control sites, and neither the biomass nor abundance of species targeted by fishers was significantly higher. Conversely, the site-level analysis revealed significant differences in targeted biomass (4.3-5.4 times higher) abundance (2.5-3 times higher), and community structure between LMMAs and controls at two of the sites, both of which had engaged hotels as co-management partners.

The key shortcoming of the site-level analysis is its lack of true replication, without which it becomes impossible to statistically separate a treatment effect from a location effect. This reduces the inferences that can be made and risks invalidating the resulting conclusions, though it does not in itself drain the analysis of its scientific merit. Many published studies have focused on a single managed area yet achieved valid inferences, predominantly through subjective but careful assessment of other variables that might confound (Hurlbert, 2010, 2009). In my analysis, the use of broad-scale benthic habitat mapping and transect-level observations of structural complexity to help control for habitat (a primary driver of between-site differences: Chapman and Kramer, 1999; Edgar and Barrett, 1997; García-Charton et al., 2004; Garcia-Charton and Ruzafa, 1999), together with the magnitude of the observed differences between the two hotel-managed *ra'ui* and controls (much greater than those reported by most studies of reef fish assemblages in marine protected areas: Lester et al., 2009; Willis et al., 2003) suggests a reserve effect at these two sites.

Although this interpretation is persuasive, it is not logically defensible, since the underlying analytical design lacks true replication. More evidence is therefore required to unequivocally discount the possibility that the observed differences were artefacts of area, rather than effects of protection.

This evidence comes from the asymmetric analysis, which also found significant differences in both the abundance and biomass of target species between LMMA sites and controls at Aroa and Edgewater, but not at two sites without hotel involvement. This analysis compared each LMMA to four control sites, partitioning the variability due to protection measures from that due to habitat/area. In so doing, it improves upon the site-level investigation by introducing spatial replication, as well as on the network-level analysis by better accounting for the variability that made it challenging to detect an effect. It therefore provides a more robust indication of a reserve effect at the hotel-supported sites than the other analyses.

Given that the results of the three analyses conflict, there is a risk that the management response could vary in accordance with whichever analysis receives attention. A manager who gives consideration to the site-level and asymmetric analyses, both of which suggest that the two hotel-supported *ra'ui* have been effective at increasing biomass and abundance of targeted species, might elect to allocate some resources to scaling and sustaining hotel management around Rarotonga. In contrast, since the network-level analysis offers little evidence of a *ra'ui* effect, a manager relying on this assessment might instead devote resources to finding alternatives to the *ra'ui* system, a course of action that would miss the potential role that hotel-supported sites can play.

Irrespective of which analysis is acted on by managers, there are several further issues that could undermine the accuracy of the transect surveys on which the analyses both depend. In general, these issues – specifically, biases – can be minimised through careful training of those who perform the surveys and will tend to affect the estimates from control and managed sites roughly equally, rarely confounding results greatly (Edgar et al., 2004). However, some biases can disproportionately influence one such estimate over another, and so have the potential to hinder accurate interpretation if they are not recognised and incorporated into study designs (Edgar et al., 2004).

Expectation bias is one such bias. Defined as the influencing of research outcomes by the conscious or unconscious predispositions of the researcher (Hrobjartsson et al., 2013), expectation bias often leads to a more positive assessment of the experimental intervention than the control, since findings that confirms one's hypotheses or expectations can be favoured (Burghardt et al., 2012; Hrobjartsson et al., 2013; Tuyttens et al., 2014).

In Chapter 4, I used underwater video fish surveys from Indonesian coral reefs to investigate the potential for expectation bias in conservation research. Postgraduate students in ecology and environmental science were randomly split into two groups and unknowingly participated in a double-blind randomised controlled trial during a workshop to measure the effectiveness of marine protected areas. An unblinded group of observers were told that half of the video transects were filmed in an unfished marine reserve and half were from fished controls, whereas a blinded group were not given such (mis) information.

I compared estimates of fish abundance from both groups and found that while there was no significant difference between the two groups on the control transects, the unblinded group significantly and incorrectly overestimated the number of fish present on the reserve transects by approximately 28% (95% CI 18.5% to 40.5%, $p>0.0001$). This study therefore provides clear evidence that expectation bias could seriously confound results in assessments of the ecological effectiveness of protected areas. To my knowledge, this is the first quantitative demonstration of this bias in conservation biology.

Since the study was an experiment in a computer lab, an important consideration is how applicable the findings are to real world ecological surveys. In other words, to what extent is the expectation bias in my sample reflective of that found amongst practising marine conservation biologists? As relative newcomers to underwater surveying, it is conceivable that the subjects used in my study could have been more optimistic about the potential of protected areas than more established researchers, and therefore more biased in their estimates. In contrast, since experienced scientists may have more significant investments in a conservation success or given research outcome, they could be more vulnerable to confirmation bias than the subjects I used (Marsh and Hanlon, 2004; Tuyttens et al., 2014). Only by repeating this kind of study with different groups might it be possible to attribute exact reasons for bias.

Of course, it is not only scientific researchers who conduct ecological evaluations of marine protected areas: both managers and resource users often do so too. A study by Léopold et al. (2009) compared estimates of abundance from resource users in Fiji with those from scientific divers and found that the users overestimated the abundance of target taxa. The researchers attributed this result to a deliberate upward bias on the part of users, who, as members of the community where the survey team were staying, were indirectly financially incentivised by the study (*ibid.*). However, they did not investigate the possibility that the scientific divers themselves may also have provided biased estimates, albeit subconsciously.

Although I focused on protected areas in Chapter 4, there is no reason to suppose that other areas of conservation biology involving comparative estimates of abundance would not be susceptible to the bias that I found. As such, if conservation scientists wish to be certain that any differences estimated between two groups stem from a given treatment rather than from the biases of the observer, I suggest that blind assessment should receive far greater attention.

My findings in Chapters 3 and 4 also touch on some broader issues about whether and how communities and conservationists monitor and evaluate marine resources (Hockley et al., 2005). Good evidence is clearly vital for effective marine resource management, and with LMMAs receiving increasing attention from international agencies, NGOs and the donor community, the need for robust research is stronger and more pressing than ever. Yet, although improving the evidence base is a key research area for LMMAs (Cohen et al., 2014), there is a danger that doing so might divert scarce resources away from management or conservation or that outputs will remain of insufficient quality to support effective decision making and priority setting (Sheil, 2001, Hockley et al., 2005). Given the informal nature and capacity constraints of many LMMAs (Rocliffe et al., 2014, Chapter 5), it is possible that the costs to the communities themselves of a rigorous LMMA evaluation programme might not outweigh the perceived benefits, leading to unmonitored exploitation over the longer term (Hockley et al., 2005). As a consequence, the findings of these chapters are likely to be of greater relevance to conservation scientists and NGO managers than to LMMA practitioners.

6.2.1 Research impact

In addition to the Cook-Islands specific impacts and outputs detailed in section 1.1.1, findings from Chapter 4 have been presented at regional and international conferences, including the 2014 International Marine Conservation Congress in Glasgow, UK, and the 2015 WIOMSA Symposium in Durban, South Africa. Researchers from the Centre for Evidence-Based Conservation at Bangor University have expressed interest in collaborating on and publishing guidelines for conservation practitioners on incorporating blind assessment into research design, a central recommendation of Chapter 4. Moreover, the marine conservation NGO Blue Ventures is currently exploring the possibility of integrating blinding into its emerging conservation tourism programme in Timor-Leste, which is using both underwater visual census and diver-operated video to monitor coral cover and fish populations.

6.2.2 Key recommendations

- Strive for better evidence of the ecological effects of spatially explicit tools like LMMAs but do not use a lack of formal proof of effects to delay or excuse remedial activities.
- Control-Impact designs that compare a single managed site to a single control and attribute any difference between sites to the effect of management are not necessarily without merit but should contain a very careful assessment of possible confounding factors.
- Replicate Chapter 4 with a large sample of practicing protected area researchers.
- Implement blind assessment where video-based survey methods are already in use.

- Where research objectives, logistics, or budget do not favour video methods consider adopting a blended approach. This would involve having a subset of the data analysed by blinded video observers and unblinded UVC surveyors and then using the differences between the two as a correction factor on the rest of the dataset.
- Replicate Chapter 4 for other designs that compare between protected and unprotected sites and use subjective methods like DAFOR (dominant, abundant, frequent, occasional or rare) or percentage cover.
- Develop guidelines to help conservation biologists to better understand under what circumstances blinding may benefit research designs, as well as to provide direction on how to properly implement a blinded protocol.

6.3 Scaling LMMAs: using practitioner networks

In Chapter 5, I shift geographic focus to the Western Indian Ocean, tracing the evolution and expansion of community management, and presenting the first-ever inventory and assessment of the region's LMMAs. I compared the key attributes of these areas to MPAs under government stewardship and assessed their relative contributions to progress towards the Convention on Biodiversity target of 10% of marine and coastal ecological regions to be effectively conserved by 2020. I also explored the legal frameworks that provide a foundation for locally managed marine initiatives in Kenya, Madagascar, Mozambique and Tanzania to assess the potential for future expansion of LMMAs in the region.

My analyses highlight that the number of LMMAs has increased rapidly over the last decade so that LMMAs now cover more than 11,000km^2 or 3.6% of the region's continental shelf. LMMAs are emerging as a tool of choice in Madagascar and mainland Tanzania, where they cover 4.2 times and 3.5 times more area than government-led MPAs respectively. Assuming that the percentage of continental shelf covered by LMMAs and MPAs is an acceptable proxy for the CBD's 10% 2020 coverage targets, Comoros, Kenya, Mozambique and Tanzania have already achieved the target at the country level, whilst Madagascar is on course to do so.

Despite the rapid expansion in coverage in the region, the country-specific analysis revealed that LMMAs were hampered by underdeveloped legal structures and enforcement mechanisms, and that a lack of organisational capacity, robust evidence and financial support all presented barriers to effectiveness of many initiatives.

In addition to these issues, I found that LMMAs in the region had tended to operate in relative isolation, with little communication or coordination between support organisations or implementing communities. The principal recommendations of the chapter are therefore to establish a network of LMMA practitioners in the WIO region to share experiences and best practice, and to invest in research to better understand how LMMAs can meet their long-term goals. I suggest that this network should be modelled on the Asia-Pacific LMMA Network, which has proven effective at facilitating information sharing and peer-to-peer learning amongst coastal communities (Cohen et al., 2014; Rocliffe and Peabody, 2012).

6.3.1 Research impact

These recommendations have been acted upon by regional implementing organisations, and work is presently underway to establish an LMMA network for the WIO, as well as a country-specific network for Madagascar (Rocliffe and Harris, 2014). MIHARI, Madagascar's nascent network, is developing quickly and already counts more than 130 LMMA villages as members (Mayol, 2013; Rocliffe and Harris, 2015).

Efforts are also in progress to integrate the WIO network with the Asia-Pacific Network. As part of this initiative, the first global LMMA workshop was held on 10th September 2012 at the IUCN World Conservation Congress in Jeju, South Korea. The meeting brought together 17 community leaders from LMMAs in the South Pacific, western and central Indian Ocean, Caribbean and Central America. It explored best practices, lessons learned and common challenges in LMMA management and laid the foundations for a global network aimed at facilitating the sharing of timely and relevant information to improve LMMA management (Rocliffe and Peabody, 2012).

Such a network is not without its challenges, however. Firstly, it would require significant time commitments from participating institutions and would need long-term donor support (Rocliffe and Peabody, 2012). Secondly, the network would need to respond to the needs of local communities and partners, which will be highly variable across different regions and countries (*ibid.*). Thirdly, the lack of a common language between LMMA communities and practitioners around the world would also present a major challenge (*ibid.*).

As well as comprising a PhD Chapter, the study has been published in the peer-reviewed literature (Rocliffe et al., 2014) and presented at several regional and international conferences, including the Third International Marine Protected Areas

Congress in Marseille, France (2013), the Ninth WIOMSA Scientific Symposium in Maputo, Mozambique (2013), the 2013 Fuller Symposium in Washington, DC, and the Ninth Pacific Islands Conference on Nature Conservation and Protected Areas in Suva, Fiji (2013).

6.3.2 Key recommendations

- Develop regional and country-specific networks of LMMA practitioners in the WIO to share best practice on management topics such as financing and evaluation and to encourage the development of further LMMAs.
- Integrate these networks with existing Asia-Pacific LMMA Network.
- Invest in research to better understand how LMMAs can meet their long-term goals.

6.4 From adversaries to associates: bringing marine conservation and fisheries management together

Food security, economic opportunities from fishing and tourism and many of the other goods and services provided by a health ocean are increasingly threatened by overfishing and global climate change. Many of the fisheries most at risk are those located along the coastlines of tropical developing countries, where a lack of management, financing, data and research attention is endangering both the biodiversity of the marine environment as well as the lives and livelihoods of those who depend upon it. Approximately ninety-seven percent of the world's 120 million fishers live in developing countries, and more than 90% of them work in the small-scale subsector (The World Bank, 2012; UN FAO, 2014). Seafood is a primary source of animal protein for hundreds of millions of people in the tropics, an estimated 5.8 million of whom earn less than US$1 per day from fishing (The World Bank, 2012)

6.4.1 Step one: reform fisheries legislation

Science-led, interdisciplinary approaches to management and conservation are emerging in response to these challenges, providing a nascent roadmap towards more sustainable policies and practices that better reconcile the management of marine ecosystems with the needs of those that depend upon them (Worm et al., 2009). The first step on this path towards ending overfishing and rebuilding depleted marine resources is the development of progressive fisheries reforms built on catch limits determined through rigorous scientific research. Although a comprehensive global assessment of all such reforms is clearly beyond the scope of this thesis, recent years have seen three instruments in particular be developed and implemented: the US Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006, the EU Common Fisheries Policy overhaul of 2013, and the 2015 *Voluntary Guidelines for Securing Sustainable Small-Scale Fisheries in the Context of Food Security and Poverty Eradication*, produced by the UN's Food and Agriculture Organisation (herein SSF guidelines).

Both the EU and US reforms share a common theme of ending overfishing and rebuilding depleted stocks; both impose strict, scientifically determined catch limits to help achieve these aims; and both give the option of transitioning to rights-based fisheries (RBF) management approaches (Barner et al., 2015), a practice they also share with the SSF guidelines (Food and Agriculture Organization of the United Nations, 2015).

6.4.2 Step two: adopt rights-based fisheries management approaches

RBF management is the second step on the path. Two of the most frequently employed strategies are individual transferable quotas, whereby each fisher owns a known fraction of the total resource and must therefore buy additional fractions from other fishers should (s)he want to fish more than his/her allotted amount; and Territorial Use Rights in Fisheries (TURFs), which assign secure tenure rights to harvest in a spatially discrete area (Auriemma et al., 2014; Barner et al., 2015). In both cases, rights can be assigned to individuals, as well as communities or cooperatives (Barner et al., 2015).

Evidence from both empirical research (Grimm et al., 2012) and modelling studies (Costello et al., 2008) suggests that, when effectively designed, enforced and monitored, RBF may offer a promising way forward for previously failing or unmanaged fisheries. Assigning secure tenure rights incentivizes sustainable fishing behaviour, overcoming the tragedy of the commons by removing the race to fish – the advantage gained from outcompeting other fishers – that has sadly all to often characterized traditional fisheries management (Barner et al., 2015; Grafton et al., 2006; Hardin, 1968). Fishers work collaboratively towards fisheries management and conservation goals for the simple reason that the value of their rights hinges on the abundance of the stock (Hilborn, 2007b).

RBF approaches have been implemented around the world and accounted for nearly a quarter of all landings by biomass in 2015 (Barner et al., 2015). In particular, ITQs have been used successfully in New Zealand (Mace et al., 2014), the US (Grimm et al., 2012; Kaplan et al., 2013) and Namibia (Oelofsen, 1999), while TURFs have shown promising results in Chile (Aburto et al., 2013; Gelcich et al., 2012) and Samoa (Young, 2013), among others.

However, despite their demonstrated potential to help end overfishing, RBF strategies are not without their shortcomings. They can be costly to implement and enforce, may not be suitable for highly migratory species, and when designed without sufficient consideration of local conditions, can favour larger boats and communities, creating inequitable outcomes (Barner et al., 2015; Olson, 2011).

Whilst these issues may often be the result of flawed design rather than an inherent weakness in the RBF approach, they nevertheless suggest that a third tactic is needed, one which can protect biodiversity and ecosystem functioning, and help to absorb the shocks of imperfect design and implementation.

6.4.3 Step three: better MPAs

Marine protected areas, then, are the third step on the path to ending overfishing. Since MPAs have been discussed at length throughout this thesis, I will consider them only briefly here. Evidence has shown that MPAs, particularly marine reserves in which all forms of extractive activity are permanently prohibited, can have both social and ecological benefits, increasing the average biomass, density, size and diversity of species within their boundaries (Halpern, 2003; Lester et al., 2009), and exporting some of this biomass beyond protected boundaries, with positive consequences for surrounding fisheries (Gell and Roberts, 2003; Harrison et al., 2012; Pelc et al., 2010; Russ and Alcala, 2010).

Despite these benefits, MPAs are not a comprehensive solution to the problem of overfishing. Their limitations fall into three broad and interrelated groupings: those relating to effectiveness, those relating to siting and those relating to underlying drivers. First, many MPAs are nothing more than ineffective paper parks unable to achieve conservation goals (Halpern, 2014). The reasons for this are numerous and multi-faceted, but include insufficient funding and active management, as well as a failure to adequate consider both biological and social goals, and the inherent tradeoffs in so doing (Balmford et al., 2004; Burke et al., 2011; Christie and White, 2007).

Secondly, whilst so-called NEOLI MPAs – No-take, well-Enforced, Older than ten years, Larger than $100km^2$ and in Isolated locations – are considered to be the most effective type of marine protected area (Edgar et al., 2014), the preference for these design features unfortunately creates a disconnect that severely undermines the potential of MPAs. From an ecosystem recovery and fisheries management standpoint, the need for MPAs is arguably greatest along crowded tropical coastlines where resources and reefs are the most degraded, and where food security and income are often most at risk.

Paradoxically, it is precisely these contexts where it is the most challenging to establish and sustain effective MPAs. Rising coastal populations, together with pressure from local communities concerned about inadequate consultation, and insufficient compensation for loss of access to fishing grounds, mean that the creation of durable and successful MPAs along coastlines is increasingly improbable (Smith et al., 2010).

Thirdly, while MPAs are often established proactively to safeguard valuable, high biodiversity areas, they do not deal with the underlying causes of the threats that they endeavour to mitigate. To borrow from the medical language of Chapter 4, this is somewhat akin to treating patients who have contracted a water-borne bacterial disease like cholera with rehydration salts. While this action is undoubtedly effective, it does nothing to prevent the disease from occurring in the first place. Today, cholera has all but been eliminated from industrialised countries not as a result of rehydration treatment, but because of the introduction of modern sewage and water treatment systems.

6.4.4 The promise of LMMAs

The lesson here is clear. In spite of their benefits and increasing use, marine protected areas are not enough to halt ocean declines, and must be paired with rights-based management approaches to deal with the underlying drivers of resource degradation. Herein lies the promise of LMMAs. The evidence presented in Chapter 5 and elsewhere (Afflerbach et al., 2014; Govan, 2009) suggests that LMMAs are often a hybrid of spatially discrete secure tenure rights (TURFs) for local fishers and marine reserves and/or other marine spatial closures. By combining fisheries management and conservation at the local level, this type of LMMA, also known as a TURF-Reserve (TR), theoretically eliminates many of the downsides of either approach when used in isolation, potentially offering a way to optimise both conservation and fisheries objectives (Costello and Kaffine, 2010).

Whilst an accurate assessment of this potential is difficult given that many initiatives are informal, recently established and under-studied, TRs are starting to receive attention from both scientists (Afflerbach et al., 2014; Yamazaki et al., 2015) and NGO practitioners. Early research has focussed primarily on stage-setting: bioeconomic modelling to explore the potential impacts of TRs prior to implementation (Costello and Kaffine, 2010) and reviews of the status and geographic extent of TRs (Afflerbach et al., 2014, Chapter 5). One intriguing finding so far is that the establishment of spatially explicit tenure rights in coastal communities can inspire resource users to create their own marine reserves, suggesting that TRs may have synergistic benefits (Afflerbach et al., 2014; Barner et al., 2015; Oliver et al., 2015; Ovando et al., 2013). The positive nature of these early results has led some NGOs to establish TR-focussed programmes. For example, both Blue Ventures Conservation and Fish Forever – a collaboration between the NGOs Rare and Environmental Defense Fund and scientists from the University of California, Santa Barbara – are working to establish and scale LMMAs that pair secure tenure and marine reserves.

6.5 A more hopeful future for fisheries: research priorities

From the evidence presented in this chapter and throughout this thesis, it is apparent that scaling and sustaining effective LMMAs will require policy reform to promote rights-based management at the local level, as well as sustainable financing and long-term, cross-disciplinary programmes of monitoring and evaluation.

6.5.1 Towards community-centred policy reform

From a policy perspective, there are several reasons to be hopeful. The recently introduced SSF guidelines, for example, state that secure, equitable, and socially and culturally appropriate tenure rights are vital for coastal fishing communities and call on governments to ensure that such rights are granted (Food and Agriculture Organization of the United Nations, 2015). Similarly, Goal 14 of the UN Sustainable Development Goals (UN, 2015) concerns itself with conservation and sustainable use of the oceans and calls for improved access to marine resources for small-scale fishing communities. Further, whilst not specifically focused on secure tenure, the Promise of

Sydney, the main output of the 2014 World Parks Congress, makes repeated mention of the need to increase community engagement in and stewardship of marine protected areas, and argues for the introduction of new approaches to incentivise them (IUCN, 2014a, 2014b, 2014c).

At the national level, the results of Chapter 5 show that governments in the Western Indian Ocean are increasingly devolving management to the local level, granting coastal fishing communities the right to control access to valuable marine resources. Although, these experiences have been supported by other regional (Govan, 2009) and global analyses (Auriemma et al., 2014), more needs to be done. This is why enhancing the role of LMMA networks, a key recommendation of Chapter 5, is so vital. Not only do such initiatives promote the sharing of experiences and best practice, they can also unify previously disparate and marginalised communities into a powerful force to advocate for policy reform. As a result of the development of the LMMA network in Madagascar, for example, the country's president recently committed to a tripling of marine protected area coverage, with a special emphasis on community-centred approaches. Understanding under what conditions practitioner networks can most effectively catalyse policy reform at the local level should therefore feature strongly in future LMMA research programmes.

6.5.2 Towards sustainable financing of LMMAs

Chapters 3 and 4 explored the interplay between tourism and LMMAs and argued in particular for a greater role for models that involve coastal hotels. Although the evidence I present suggests that such an approach shows undoubted promise, it is only one of many options for involving the tourism sector. To date, tourism-centred efforts to provide both alternative livelihoods for fishers and sustainable financing for management and conservation initiatives have focused largely on community-based marine ecotourism (CBMET). The promise of this approach lies in its emphasis on the triple bottom line: minimising the *environmental* impacts of tourism whilst benefitting local communities, *socially* and *economically* through employment, community projects and capacity building (Durham, 2009; Morris, 2002; Stronza, 2009; Svensson et al., 2009). As a result, it has proven to be a popular means of supporting conservation and development, especially in developing countries, and has a wide base of support.

CBMET is a broad church and can encompass a range of potential initiatives. Taking just one region, the Western Indian Ocean, as an example, in the Bay of Ranobe in southwest Madagascar, the NGO Reef Doctor has worked with local communities to establish LMMAs at Massif des Roses (Rose Garden) and Ankaranjelita (Belle et al., 2009). FI.MI.HA.RA, an association of local stakeholders, is responsible for managing both sites and is fully funded by an entrance fee of 5,000 ariary (USD$1.60) levied on tourists visiting Massif des Roses (Reef Doctor, 2012). Across the Mozambique Channel, and away from reefs in Gazi, Kenya, a local women's group has established a 350m-long mangrove boardwalk. Tourists pay a fee to use the boardwalk, the profits from which are returned to the community, and have been used to improve healthcare, sanitation and school facilities, as well as to provide school scholarships for children from the village (UNEP-Nairobi Convention and WIOMSA, 2015).

It is clear from these examples that community-based marine ecotourism can produce revenues for local communities, and can lead to positive conservation and development outcomes, notions which are broadly supported by the literature (Bookbinder et al., 1998; Brightsmith et al., 2008; Cater and Cater, 2007; Durham, 2009; e.g. Goodwin, 1996; Goodwin and Santilli, 2009; Sarr et al., 2008; Stronza, 2009).

Despite this promise, however, such initiatives are not without shortcomings. There have been few thorough evaluations of CBMET and much of the evidence that exists lacks statistical rigor and appears and project reports or workshop proceedings rather than in peer-reviewed publications (Kiss, 2004; Stronza, 2009). The marine environment is sensitive and easily impacted by tourists, who may inadvertently damage reefs whilst diving or snorkelling, and whose demand for seafood may increase pressure on already vulnerable stocks (Barker and Roberts, 2004; Cesar et al., 2003; Hawkins et al., 1999; Hawkins and Roberts, 1993).

From a financial perspective, an important question for conservationists is to what extent CBMET provides communities with an effective incentive to promote conservation. Here too, the evidence is equivocal. In many cases, projects have delivered only a small boost to local livelihoods and employment, and have led to little change in resource use (Kiss, 2004; Stonich, 2000; Stronza, 2009). Initiatives also often increase local reliance on a single revenue stream, a large proportion of which may be lost through "leakage" to other regions or countries (Bookbinder et al., 1998; Stonich, 2000; Stronza, 2009).

There are cases where the benefits of CBMET are sufficiently attractive so as to outcompete other livelihoods, but on account of their success, such instances often appeal to multiple external stakeholders, increasing resource pressure and diluting benefits (Kiss, 2004; Wunder, 2000). As a result, few coastal communities have realised sufficiently high benefits from CBMET to provide effective incentives for conservation, and many remain dependent on external support for long to indefinite periods (Kiss, 2004; Stronza, 2009). Accordingly, if LMMAs are to be sustainably financed – and communities incentivised to manage them effectively – other approaches beyond tourism are needed.

Such approaches include: informal systems of fisheries taxes or fees; use of fines levied on rule breakers; Marine Stewardship Council eco-certification; community-based aquaculture; conservation trust funds; blue carbon; and short-term spatial closures for fast-recovering invertebrate species (Latham and Rocliffe, 2016). In particular, aquaculture, blue carbon and short-term closures appear to be gaining traction in the context of LMMAs.

Evidence suggests that community-based aquaculture – the cultivation or rearing of aquatic plants and animals, usually as an alternative livelihood to fishing – can make a significant contribution to economic growth and food security, and can also improve nutrition and water resource management (FAO, 2014; Frankic and Hershner, 2003; Purcell et al., 2010; Rougier et al., 2013; WWF, 2015). However, several challenges including habitat modification and pollution, shortages of seed and feed supplies, as well as limited investment, government support and technical capacity undermine aquaculture's potential scalability (Ateweberhan et al., 2014; Bower, 2000; Eriksson et al., 2012; Halling et al., 2013; Rougier et al., 2013).

For blue carbon – efforts to unlock and market the carbon sequestration value of marine vegetation – the picture is similarly mixed (AGEDI, 2014). Whilst pilot projects and emerging research suggest a high potential for such initiatives to incentivise mangrove LMMAs, blue carbon is yet to be included in any regulatory frameworks and standards are still in their infancy (Donato et al., 2011; Hamrick and Goldstein, 2015; Huxham, 2013; Jones et al., 2014). In addition, due to the high costs and multi-year time frames required for project establishment, there are very few functioning pilot sites, and the volatility of carbon markets makes it difficult to forecast returns (Hamrick and Goldstein, 2015). More research and larger, more numerous projects are needed in order to better determine the potential for blue carbon to sustainably finance LMMAs.

A temporary ban on the harvesting of marine resources in a designated area, short-term or periodic closures are increasingly being used in community-based management initiatives across the Indo-Pacific and are supported by a growing evidence base (Bartlett et al., 2009; Cinner et al., 2006; Cohen and Foale, 2013). Recent research suggests that periodic closures can be particularly effective tools for the management of short-lived, fast-reproducing species. For example, Oliver et al. (2015) found that a long-established system of periodic closures used in the management of the reef octopus *Octopus cyanea* in Madagascar could improve fisher catches and income. Octopus landings increased by more than 700% in the month following the lifting of a closure, boosting the catch per fisher per day by almost 90% over the same period (Oliver et al., 2015). The apparent success of these closures, which typically cover 25% of a community's overall octopus fishing grounds and are in place for 2-3 months at various times of year, led to other communities following suit. As of March 2016, more than 250 closures have taken place.

Aside from increasing the income and catches of the communities that implement them, there is a tantalising possibility that community-led periodic closures can catalyse broader conservation actions at the local level. Following the successful establishment of the octopus closures in Madagascar, small-scale fishing communities across the country have grouped together to establish more than 60 LMMAs covering over 11% of the island's seabed, many of them incorporating community-enforced marine reserves permanently off limits to fishing.

Of course, the most effective conservation strategy for any given site must be developed based on a clear understanding of the social, economic and environmental context, as well as on a careful and pragmatic evaluation of the options (Kiss, 2004). Ecotourism, aquaculture, blue carbon and periodic closures are not mutually exclusive models for funding LMMAs, and in many cases, may prove to be most effective when integrated together. As the portfolio of financing options grows, so too must the body of research devoted to evaluating the effectiveness of these emerging approaches, both individually and in combination.

6.5.3 Towards long-term, integrated monitoring of LMMAs

A common thread that binds together this thesis is the need for more rigorous and multi-disciplinary evaluations of area-based conservation and management initiatives. Indeed, this is a specific focus of Chapters 2, 3 and 4. Chapter 4 took a *social* research approach rooted in medicine and psychology – the double-blind randomized controlled trial – to explore a bias with the potential to negatively affect *ecological* assessments of protected or managed marine areas. Chapters 2 and 3 also combined the social and the ecological, using respondent-completion questionnaires and semi-structured interviews, as well as underwater visual census techniques to explore marine resource management and conservation in the Cook Islands.

Taking a more holistic approach not only allows a more complete picture of conservation challenges to emerge, but also facilitates the development of potentially more innovative and effective solutions than would have been possible had the analysis been constrained to the silo of a single discipline. After all, had my work in the Cook Islands been restricted to an ecological study centred on counting and sizing reef fish on transects inside and outside LMMAs, I would have been unable to establish enforcement histories of the different sites, or to determine which species were targeted by local fishers, making it challenging to ascertain overall ecological effectiveness. Moreover, I would have completely overlooked the potential role for hotel-based management there. Similarly, had I limited my analysis solely to tourist perceptions of the marine environment in the Cook Islands, I would have had no way of knowing whether or not these perceptions had a basis in ecological reality.

Nevertheless, while the use of multi-disciplinary research approaches in the thesis has arguably both increased its overall value and engendered a broader and more nuanced array of conclusions and recommendations, the subject matter for the individual chapters based was guided less by theoretical considerations than by on-the-ground realities. At its heart, this thesis is a practical beast, and with the exception of Chapter 4, reflects the priorities of the resource managers and NGO practitioners I consulted who work with coastal communities in the Cook Islands and Western Indian Ocean.

While such an approach is not without merit, with MPAs continuing to fail and with LMMAs receiving increasing donor scrutiny as their popularity grows, future research in this field could undoubtedly benefit from adopting a more structured and integrated design unconstrained by the three-year time period of UK doctoral research. In particular, it is recommended that forthcoming programmes examine LMMAs more holistically, perhaps through the lens of common-pool resource theory. To date, some authors have sought to identify the fundamental social, economic, and institutional attributes of effective co-management schemes, with many taking Ostrom's institutional design principles (1990) as a starting point (E.g. Agrawal and Benson, 2011; Brooks et al., 2012; Cinner et al., 2012c; Gutierrez et al., 2011; Pollnac et al., 2001). Although this is not the approach adopted by this thesis, the strengthened institutional capacity of LMMAs that may result from practitioner networks (Chapter 5) together with the improved assessments and sustainable financing explored in Chapters 2,3 and 4 could increase the likelihood that the expected benefits of management will outweigh the costs. This would satisfy the most fundamental test provided by Ostrom's framework for analysing sustainability of social-ecological systems (Ostrom, 2009).

Initiatives should also consider the notion of equity, an area that has received little research attention to date in the context of LMMAs. Not everyone will be affected by LMMAs in the same way; impacts and trade-offs between groups will be unequal and may vary over time (Chaigneau and Brown, 2016). Whilst early indications are that LMMAs may favour wealthier resource users (Cinner et al., 2012c), a more explicit treatment, ideally built upon a multi-dimensional framework such as that advocated by McDermott et al. (2013) could prove to be a fruitful avenue for further research.

Finally, to maximise the certainty that assessments of LMMAs are not confounded by temporal variations in marine environment, researchers should strive for long-term and scientifically credible monitoring of LMMAs. Such an undertaking is not without its difficulties, however. Even for centrally managed MPAs receiving budgetary support from governments, there are very few long-term ecological monitoring programmes (Addison, 2011) and fewer still that incorporate a social dimension. High costs and burdensome information requirements are part of the problem, as is the ever-evolving nature of social-ecological systems theory, which could conceivably derail a long-term programme, should new insights emerge that contradict the framework on which the programme was orginally built. A further issue comes from the culture of modern science itself, which favours new findings over the maintenance of existing work and dissuades early-career researchers from assuming responsibility for long-established initiatives (Lindenmayer and Likens, 2010) Taken together, these attitudes can fragment project leadership, and create considerable disincentives to sustain long-term monitoring (Lindenmayer and Likens, 2010).

In many ways, the Cook Islands could provide an ideal study site for implementing the research programme outlined in the preceding paragraphs, in spite of the challenges of so doing. At present, the *ra'ui* system around the main island of Rarotonga is underperforming, and suffers from many of the issues that plague area-based conservation initiatives, including a lack of financial and stakeholder support (Chapters 2 and 3). However, the forthcoming Cook Islands Marine Park is set to create zones that ban commercial fishing around each of the islands within its boundaries. These zones, which will extend out to 50 nautical miles, promote rights-based management for local fishers in a structure analogous to a TURF + reserve, and therein have the potential to overcome many of the issues constraining the current *ra'ui* system through incentivising sustainable use.

Using these new locally managed sites, the research presented in Chapters 2 and 3 could be expanded into a long-term, quasi-experimental Before-After-Control-Impact research programme in which both the impact and control sites would be sampled before, throughout and after the establishment of the LMMAs. This assessment could include information on connectivity, habitat type and broader representativeness. From an ecological perspective, this would help to overcome the methodological limitations explored in Chapter 3, and could also combine video transects and underwater visual census techniques to better understand the biases around non-blinded assessment highlighted by Chapter 4. In addition, the social aspects of the research could be broadened to consider the perspectives of all stakeholders as opposed to just tourists' (Chapter 2). Such an approach has the potential not only to better incorporate local ecological knowledge but also to help determine under what circumstances LMMA benefits are equitably shared. Overall, given the recently established and under-researched nature of many LMMAs, integrating ecological and social data in this way in the Cook Islands could offer an unparalleled opportunity to more thoroughly evaluate LMMA effectiveness over the long-term, as well as to inform the design and implementation of future initiatives.

6.6 Concluding remarks

Our oceans, long considered inexhaustible, are complex, variable and threatened at a global scale by a multitude of anthropogenic pressures, including overfishing, population growth, climate change and habitat loss (Halpern et al., 2008; Jackson, 2008; Jackson et al., 2001). From a conservation standpoint, attempts to tackle ocean declines have centred on the use of marine protected areas (MPAs), which place areas off limits to exploitation or excessive use. MPAs are promoted as a win-win approach that can benefit both nature (Halpern, 2003; Lester et al., 2009), and people (Brander et al., 2015; Van Beukering et al., 2013) but often fail to do either (Caveen et al., 2014), existing only as 'paper parks' without active management or enforcement (Halpern, 2014; Mora et al., 2006).

This is not to say that MPAs should be discarded. In spite of their shortcomings, there are numerous examples of successful initiatives (Chapter 1) around the world, and the protected area strategy remains a cornerstone of conservation in the marine realm, just as it does on land (Kareiva and Marvier, 2012). To echo Marvier's (2014, p. 2) sentiments, "conservation absolutely needs protected areas, but it also needs new solutions that tackle the systemic root causes of planetary degradation". In essence, marine conservation should also set its sights on fisheries management, and experiment with new approaches.

LMMAs, especially those that pair rights-based management and community-led resource conservation areas, are one such effort in this vein. Through dealing with the underlying drivers of resource degradation, and by creating incentives that make conservation economically acceptable – and ideally attractive – to coastal communities, LMMAs have the potential to overcome many of the weaknesses of more conventional approaches to resource management and conservation. They may therefore be a promising new approach marine conservation, particularly in impoverished and densely populated coastal regions where there is high dependence on marine ecosystem services for food and income.

There are some (e.g. Soulé, 2013) who argue that such a human-centred approach to conservation is not worthy of the label at all, and that nature should be protected for its intrinsic worth rather than for its benefits to humanity. While this is a laudable aim, research suggests that – in the US at least – messaging centred on the protection of nature for its intrinsic value preaches only to the converted to a large extent (Marvier and Wong, 2012). Given this, if we want to maximise conservation's impact and reach, it is arguably better to accept a degree of self-interest and work to develop solutions that align the two (Marvier, 2014).

The results presented here are not a panacea for ocean ills. Overcoming the numerous stressors that threaten the oceans will require LMMAs to not only be strengthened, expanded and underpinned where possible by rigorous, long-term scientific monitoring and evaluation, but also integrated more effectively into broader marine spatial planning efforts (Aswani and Ruddle, 2013; Gaymer et al., 2014; Mills et al., 2012). It will require an ambitious network of effectively managed and well-enforced marine reserves, nestled within even more ambitious ecosystem-based management initiatives (Agardy et al., 2011; McCauley et al., 2015). It will require the continued development and implementation of progressive fisheries reforms built on catch limits and determined through rigorous scientific research. It will require a much greater uptake of carefully designed rights-based management approaches that assign secure tenure rights or otherwise incentivise sustainable fisheries (Costello et al., 2008).

Most crucially of all, it will require on-going collaboration. To manage fisheries, we must manage people (Hilborn, 2007b). And to do this most effectively, we must continually find new ways to bring together social scientists and ecologists, local communities and governments, NGOs and resource managers, seafood companies and coastal fishers, tourism businesses and resource users, researchers and LMMA practitioners.

Achieving this, while far from a simple endeavour, is the key to a more hopeful future for tropical coastal conservation in the tropics, one that at last addresses the gulf between theory and reality in win-win conservation, providing a gateway to sustainable fisheries, vibrant oceans, and improved food security for over a billion people.

7 APPENDICES

APPENDIX A: PUBLICATIONS ARISING FROM THESIS

Journal articles

Rocliffe S., Peabody S., Samoilys M., Hawkins J.P., 2014. Towards A Network of Locally Managed Marine Areas (LMMAs) in the Western Indian Ocean. *PLOS ONE.* 2014;9: e103000. doi:10.1371/journal.pone.0103000

Rocliffe, S., Anderson, L.G., Glass, L., Harris, A., 2015 (in review). Incentives for community-based tropical marine conservation: panacea or pipe dream. *Aquatic Conservation*

Reports

Rocliffe, S., and Peabody, S., 2013. Locally-managed marine areas: towards a global learning network workshop, World Conservation Congress, Jeju, South Korea, 10 September 2012. Blue Ventures. London.

Rocliffe, S., and Harris, A., 2015. Short-term fishery closures: experiences from around the world. Blue Ventures. London.

Conference presentations

Rocliffe, S., and Anderson, L.G., 2014. In the dark: should conservation biologists use blinding? Third International Marine Conservation Congress, Glasgow, Scotland, 14-18 August 2014.

Rocliffe, S., 2013. Participatory approaches to MPAs – achieving meaningful and effective engagement. Plenary at 3rd International Marine Protected Areas Congress, Marseille, France, 21-27 October 2013.

In preparation

Rocliffe, S., Anderson, L.G., Andradi-Brown, D., and Exton, D., (in prep). In the dark? A call for greater use of blind assessment in conservation biology

Rocliffe, S., Anderson, L.G., Coward, G., Williams, E., and Hawkins, J.P., 2015 (in prep). The importance of tourist perceptions of marine resource management in Small Island Developing States (SIDS)

Rocliffe, S., Anderson, L.G., Coward, G., Williams, E., and Hawkins, J.P., 2015 (in prep). Is there a role for hotels in supporting Locally Managed Marine Areas? A tale of two analyses

Other articles, reports and key presentations produced during my PhD candidature

Anderson, L.G., **Rocliffe S.,** Haddaway, N.R**,** Dunn, A.M, 2015. The role of tourism in the spread of non-native species: a systematic review and meta-analysis. *PLOS ONE.* 2015; 10:e0140833.doi:10.1371/journal.pone.0140833

Anderson, L.G., Dunn, A.M., **Rocliffe, S.,** Stebbing, P.D, 2015 (in review). Aquatic biosecurity best practice: lessons learned from a global leader. *Journal of Environmental Management*.

Anderson, L.G., **Rocliffe, S.,** Dunn, A.M., Stebbing, P.D. 2014. Aquatic biosecurity best practice: lessons learned from New Zealand. Report to the Department of Environment, Farming and Rural Affairs, London.

Rocliffe, S., 2012. Ensuring sustainable and equitable live baitfish fisheries for tuna pole-and-line. London: International Pole and Line Foundation. 57pp

Allsopp, M., Miller, C., Atkins, A., **Rocliffe, S.,** Tabor, I., Santillo, D., and Johnston, P. Review of the current state of development and the potential for environmental impacts of seabed mining operations. Greenpeace Research Laboratories Technical Report (Review) 03-2013. 50pp

Rocliffe, S., and Harris, A., 2015. The status of octopus fisheries in the Western Indian Ocean. Blue Ventures. London.

Rocliffe, S., 2014. Experiences of short-term fishery closures in other (non-octopus) fisheries. Scaling success in octopus fisheries management in the Western Indian Ocean workshop, 3-5 December 2014, Stone Town, Zanzibar.

Rocliffe, S., 2014. The status of octopus fisheries in the Western Indian Ocean. Scaling success in octopus fisheries management in the Western Indian Ocean workshop, 3-5 December 2014, Stone Town, Zanzibar.

Rocliffe, S., 2015. Harvesting deep sea resources: innovation or exploitation. Natural History Museum, London, UK, 24 April 2015

APPENDIX B: QUESTIONNAIRE, INTERVIEW CONSENT FORM AND INFORMATION SHEETS

THE UNIVERSITY of York

CONSENT FORM

KEY INFORMANT INTERVIEWS

University of York (UK)
Ministry of Marine Resources (Cook Islands)
Ra'ui research study

- I, the undersigned, have read and understood the Study Information sheet provided.
- I have been given the opportunity to ask questions about the study and have had these questions answered to my satisfaction.
- I have been given adequate time to consider my decision and I agree to take part in the study.
- I understand that I can withdraw from the study at any time and I will not be asked any questions about why I no longer want to take part.
- I understand that taking part in the study will include being interviewed and audio recorded.
- I understand that my personal details such as name and employer address will not be revealed to people outside the project.
- I understand that my words may be quoted in publications, reports, web pages and other research outputs but my name will not be used without my prior approval.

Name of Participant: ______________________________ Date: __________

Participant Signature: ______________________________

Researcher Signature: ______________________________ Date: __________

THE UNIVERSITY *of York*

STUDY INFORMATION

KEY INFORMANT INTERVIEWS

University of York (UK)
Ministry of Marine Resources (Cook Islands)
Ra'ui research study

- Thank you very much for agreeing to participate in this study. This Information Sheet explains what the study is about and how we would like you to take part in it.
- The purpose of this study is to investigate the effectiveness of the *Ra'ui*, with a specific focus on the potential of hotels to fund or manage community-run marine protected areas. There are three components to the research: i) an assessment of fish and invertebrate abundance and size in the *Ra'ui* and at non-*Ra'ui* sites using standard underwater survey techniques; ii) a self-completion questionnaire examining public attitudes, knowledge and perceptions of the *Ra'u*i; and iii) in-depth interviews with resource users, community leaders, government officials, tourism operators and NGO representatives.
- In order to elicit your views, we would like you to be interviewed by one of the researchers involved in the Study at the University of York (UK). If you agree to this, the interview will be audio recorded and will last approximately 45 minutes.
- The information provided by you in the interview will be used for research purposes. It will not be used in a manner which would allow identification of your individual responses without your prior approval.
- This study has been considered by an Institutional Ethics Committee at the University of York and has been given a favourable review
- Once again, we would like to thank you for agreeing to take part in this study. If you have any questions about the research at any stage, please do not hesitate to contact us.

Steve Rocliffe
University of York
Cook Islands mobile: 58536
Email: sr588@york.ac.uk

THE UNIVERSITY of York

STUDY INFORMATION
SELF-COMPLETION QUESTIONNAIRES

University of York (UK)
Ministry of Marine Resources (Cook Islands)
Ra'ui research study

- Thank you very much for agreeing to participate in this study. This Information Sheet explains what the study is about and how we would like you to take part in it.
- The purpose of this study is to find out about how people use the beaches and lagoon in Rarotonga. The results will be used to help improve management of the area.
- In order to elicit your views, we would like you to fill out a short self-completion questionnaire. The questionnaire should take no more than five minutes to complete and we do hope you can spare the time to give us your views.
- The information provided by you in the interview will be used for research purposes. It will not be used in a manner which would allow identification of your individual responses.
- This study has been considered by an Institutional Ethics Committee at the University of York and has been given a favourable review.
- Once again, we would like to thank you for agreeing to take part in this Study. If you have any questions about the research at any stage, please do not hesitate to contact us.

Steve Rocliffe
University of York
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LAGOON USER SURVEY

THE UNIVERSITY *of York*

This survey aims to obtain your views about Rarotonga's beaches and lagoon. It will take no more than five minutes to complete. The answers you give are completely confidential and will help manage these areas better. Thank you for agreeing to complete this survey.

Unless otherwise asked, please choose the ONE *answer that best represents your view by placing a tick in the appropriate box.*

A. ABOUT YOUR VISIT TO RAROTONGA

Q1. During this visit to Rarotonga, on how many days did you go to the beach?

- ☐ 1-3 days
- ☐ 4-6 days
- ☐ 7-9 days
- ☐ 9+ days
- ☐ Don't know

Q2. What beach-based activities have you participated in during this visit to Rarotonga?

	Yes	No
Beach walking	☐	☐
Canoeing/kayaking	☐	☐
Fishing	☐	☐
Kitesurfing	☐	☐
Lagoon Cruise	☐	☐
Picnicking/Barbecuing	☐	☐
Photography	☐	☐
Sailing	☐	☐
Stand-up paddle boarding (SUP)	☐	☐
SCUBA diving	☐	☐
Snorkelling	☐	☐
Sunbathing/Relaxing	☐	☐
Surfing/Bodyboarding	☐	☐
Swimming	☐	☐
Windsurfing	☐	☐
Other (please specify)	☐	☐

Q3. How often do you participate in (A) SCUBA diving and (B) Snorkelling?

	(A) SCUBA	(B) Snorkel
Weekly	☐	☐
Every two weeks	☐	☐
Monthly	☐	☐
Several times a year	☐	☐
Once a year	☐	☐
Less often	☐	☐
First time	☐	☐
Never *>> Go to question 5*	☐	☐

Q4. If you: (A) SCUBA dive or (B) Snorkel, how would you rate your ability for each?

	(A) SCUBA	(B) Snorkel
Very good	☐	☐
Good	☐	☐
Fair	☐	☐
Poor	☐	☐
Very poor	☐	☐
Don't know	☐	☐

Q5. Are corals...

	Yes	No	Don't Know
Animals?	☐	☐	☐
Plants?	☐	☐	☐
Alive?	☐	☐	☐

B. YOUR AWARENESS OF MARINE RESERVES

Q6. Have you heard of any community-run marine reserves (also called Ra'ui) around Rarotonga?

- ☐ Yes
- ☐ No *>> Go to question 10*

Q7. If YES, which of the following beaches are in Ra'ui?
Please tick ALL that apply

- ☐ Black Rock
- ☐ Edgewater Hotel
- ☐ Fruits of Rarotonga/Tikioki
- ☐ Muri
- ☐ Rarotongan Hotel/Aroa
- ☐ Club Raro
- ☐ Asunto
- ☐ Don't know

Q8. How did you find out about the Ra'ui?
Please tick ALL that apply

- ☐ Brochure/leaflet
- ☐ Buoy markers
- ☐ Guidebook
- ☐ Hotel (please specify)
- ☐ Internet/website
- ☐ Local knowledge
- ☐ Newspaper/magazine article
- ☐ Sign
- ☐ Word of mouth
- ☐ Don't know
- ☐ Other (please specify)

Q9. Which of the following are allowed in a Ra'ui?
Please tick ALL that apply

- ☐ Commercial fishing
- ☐ Fish feeding
- ☐ Recreational fishing
- ☐ SCUBA Diving
- ☐ Shell collecting
- ☐ Snorkelling
- ☐ Spearfishing
- ☐ Swimming
- ☐ Walking on the reef
- ☐ Don't know

C. YOUR VIEWS ON THE BEACHES AND LAGOON

Q10. Overall, how satisfied or dissatisfied are you with the beaches you visited?

- ☐ Very Satisfied
- ☐ Satisfied
- ☐ Neither satisfied nor dissatisfied
- ☐ Dissatisfied
- ☐ Very dissatisfied

PLEASE TURN OVER >>

Q11. Thinking about the beaches you visited, how important are each of the following features to you?

	Extremely important	Very important	Somewhat important	Not very important	Not at all important
Access to toilet facilities	☐	☐	☐	☐	☐
Availability of a range of activities	☐	☐	☐	☐	☐
Water quality in lagoon	☐	☐	☐	☐	☐
Amount of litter in lagoon	☐	☐	☐	☐	☐
Amount of litter on beach	☐	☐	☐	☐	☐
Number of people on the beach	☐	☐	☐	☐	☐
Number of people in the lagoon	☐	☐	☐	☐	☐
Amount of coral	☐	☐	☐	☐	☐
Different types of coral	☐	☐	☐	☐	☐
Number of fish in lagoon	☐	☐	☐	☐	☐
Number of big fish in lagoon	☐	☐	☐	☐	☐
Different types of fish in lagoon	☐	☐	☐	☐	☐

Q12. Similarly, how satisfied are you with each of the following features?

	Extremely satisfied	Very satisfied	Somewhat satisfied	Not very satisfied	Not at all satisfied	No experience
Access to toilet facilities	☐	☐	☐	☐	☐	☐
Availability of a range of activities	☐	☐	☐	☐	☐	☐
Water quality in lagoon	☐	☐	☐	☐	☐	☐
Amount of litter in lagoon	☐	☐	☐	☐	☐	☐
Amount of litter on beach	☐	☐	☐	☐	☐	☐
Number of people on the beach	☐	☐	☐	☐	☐	☐
Number of people in the lagoon	☐	☐	☐	☐	☐	☐
Amount of coral	☐	☐	☐	☐	☐	☐
Different types of coral	☐	☐	☐	☐	☐	☐
Number of fish in lagoon	☐	☐	☐	☐	☐	☐
Number of big fish in lagoon	☐	☐	☐	☐	☐	☐
Different types of fish in lagoon	☐	☐	☐	☐	☐	☐

D. YOUR VIEWS ON PROTECTING THE OCEAN

Q13. How important do you consider the following aspects of ocean protection to be?

	Extremely important	Very important	Somewhat important	Not very important	Not at all important	Don't know
Areas of the sea that are protected from fishing (marine reserves)	☐	☐	☐	☐	☐	☐
Local community involvement in marine reserves	☐	☐	☐	☐	☐	☐
Policing of marine reserves	☐	☐	☐	☐	☐	☐
Education about marine reserves	☐	☐	☐	☐	☐	☐

DATE ______ TIME ______ SITE ______ INTV ______

Q14. What percentage of Cook Islands waters IS within fully protected marine reserves where fishing is not allowed?

- ☐ Less than 10%
- ☐ 10-19%
- ☐ 20-29%
- ☐ 30-39%
- ☐ 40-49%
- ☐ 50% and above
- ☐ Don't know

Q15. What percentage of Cook Islands waters DO YOU THINK SHOULD BE within fully protected marine reserves where fishing is not allowed?

- ☐ Less than 10%
- ☐ 10-19%
- ☐ 20-29%
- ☐ 30-39%
- ☐ 40-49%
- ☐ 50% and above
- ☐ Don't know

E. ABOUT YOU

Q16. Are you...

- ☐ Male?
- ☐ Female?

Q17. How old are you?

- ☐ 18-25
- ☐ 26-35
- ☐ 36-45
- ☐ 46-55
- ☐ 56-65
- ☐ 65+

Q18. What is the highest level of education you have achieved?

- ☐ Primary school
- ☐ Secondary school (high school)
- ☐ Undergraduate degree (Such as BSc, BA)
- ☐ Postgraduate degree (Such as MSc, MA, PhD)

Q19. What is your nationality?
Please write in

Q20. Where do you usually live?
Please write in your country

Q21. How would you rate your knowledge about coral reefs?

- ☐ Very good
- ☐ Good
- ☐ Fair
- ☐ Poor
- ☐ Very poor
- ☐ Don't know

THANK YOU FOR COMPLETING THIS QUESTIONNAIRE! YOUR HELP IS APPRECIATED. PLEASE HAND IT BACK TO THE INTERVIEWER.

NUMBER ☐☐☐☐☐ CHECKED ☐

APPENDIX C: DATA CLASSIFICATION TABLE SHOWING VARIABLES COLLECTED FROM QUESTIONNAIRE RESPONDENTS

1. SOCIO-DEMOGRAPHICS

Question	Description	Data classification
Sex	Asks respondents to state their gender	Male / Female
Age category	Age of the respondent at last birthday	18-25 / 26-35 / 36-45 / 46-55 / 56-65 / 65+
Education	Highest level of education achieved	Primary school Secondary school (high school) Undergraduate degree (Such as BSc, BA) Postgraduate degree (Such as MSc, MA, PhD)
Nationality?	Determines the respondent's nationality	Open-ended text
Where do you usually live?	Determines which country the respondent usually resides in	Open-ended text
How would you rate your knowledge about coral reefs?	Self-rated level of knowledge about coral reefs	1=very poor 2=poor 3=fair 4=good 5=very good DK=don't know

2. VISIT ATTRIBUTES

Question	Description	Response categories
During this visit to Rarotonga, on how many days did you go to the beach?	Frequency of beach/lagoon visitation	1=1-3 days 2=4-6 days 3=7-9 days 4=9+ days DK=don't know
What beach-based activities have you participated in during this visit to Rarotonga??	Types of activity undertaken by respondents at the beach/lagoon	Yes / No for each of the following options. Multiple responses permitted Beach walking Canoeing/kayaking Fishing Kitesurfing Lagoon Cruise Picnicking/Barbecuing Photography Sailing Stand-up paddle boarding (SUP) SCUBA diving Snorkelling Sunbathing/Relaxing Surfing/Bodyboarding Swimming Windsurfing Other (please specify)
How often do you participate in the following activities?	Frequency of SCUBA diving and snorkelling	Two activities: a. Snorkel b. SCUBA dive Options for each activity: Weekly Every two weeks Monthly Several times a year Once a year Less often

		First time Never (skip next question)
If you snorkel or SCUBA dive, how would you rate your ability for each	Self-rated level of SCUBA diving and snorkelling ability	Two activities: a. Snorkel b. SCUBA dive Rating for each activity 1=very poor 2=poor 3=fair 4=good 5=very good DK=don't know
Are corals…	Used to test respondent's basic knowledge about coral reefs	Yes / No / Don't know for each of the following options a. Animals b. Plants c. Alive

3. VISITOR AWARENESS OF RA'UI SYSTEM

Question	Description	Response categories
Have you heard of any community-run marine reserves (also called Ra'ui) around Rarotonga?	Overall respondent awareness of the Ra'ui	Yes/No
If YES, which of the following beaches are in Ra'ui?	Do respondents know where the Ra'ui are located?	*Please tick all that apply* Black Rock Edgewater Hotel Fruits of Rarotonga/Tikioki Muri Rarotongan Hotel/Aroa Club Raro Asunto Don't know
If YES, how did you find out about the Ra'ui?	Major information sources that respondents used to learn more about the Ra'ui	*Please tick all that apply* Brochure/leaflet Buoy markers Guidebook Hotel (please specify) Internet/website Local knowledge Newspaper/magazine article Sign Word of mouth Don't know Other (please specify)
If YES, which of the following are allowed in a Ra'ui?	Testing awareness of Ra'ui regulations	*Please tick all that apply* Commercial fishing Fish feeding Recreational fishing SCUBA Diving Shell collecting Snorkelling Spearfishing Swimming Walking on the reef Don't know

4. VISITOR PERCEPTIONS OF AND SATISFACTION WITH NATURAL RESOURCE MANAGEMENT

Question	Description	Response categories
Overall, how satisfied or dissatisfied are you with the beaches you visited?	Overall respondent satisfaction	1=very dissatisfied 2=dissatisfied 3=neither satisfied nor dissatisfied, 4=satisfied 5=very satisfied
Thinking about the beaches you visited, how important are each of the following features to you?	Gauges respondent perceptions of the beaches and lagoon and the importance they attach to respective aspects	This is a matrix format 1=not at all important 2=not very important 3=somewhat important 4=very important 5=extremely important Options: Access to toilet facilities Availability of a broad range of activities Water quality in lagoon Amount of litter in lagoon Amount of litter on beach Number of people on the beach Number of people in the lagoon Amount of coral Different types of coral Number of fish in lagoon Number of big fish in lagoon Different types of fish in lagoon
Similarly, how satisfied are you with each of the following features?	Gauges respondent satisfaction with the beaches and lagoon	This is a matrix format 1=not at all satisfied 2=not very satisfied 3=somewhat satisfied 4=very satisfied 5=extremely satisfied 0=no experience Response categories are as above

5. VISITOR ATTITUDES TOWARDS MARINE CONSERVATION

Question	Description	Response categories
How important do you consider the following aspects of ocean protection to be?	What importance to respondents assign to different aspects of marine protected areas?	1=not at all important 2=not very important 3=somewhat important 4=very important 5=extremely important DK=Don't know Options Areas of the sea that are protected from fishing (marine reserves) Local community involvement in marine reserves? Policing of marine reserves? Education about marine reserves?
What percentage of Cook Islands waters IS within fully protected marine reserves where fishing is not allowed?	Respondent awareness of what percentage of Cook Islands waters is free of fishing	1=less than 10% 2=10-19% 3=30-39% 4=40-49% 5=50% and above 0=Don't know
What percentage of Cook Islands waters DO YOU THINK SHOULD BE within fully protected marine reserves where fishing is not allowed?	Respondent attitudes about marine protection	1=less than 10% 2=10-19% 3=30-39%, 4=40-49% 5=50% and above 0=Don't know

APPENDIX D: DEBRIEFING FORM

Debriefing form

Thank you for your participation in our study! Your participation is greatly appreciated.

Purpose of the Study

Earlier, we told you that the study was about assessing the effectiveness of a new HD video technique for estimating fish populations. In actuality, our study is about observer bias in assessments of marine protected areas. We wanted to see whether you unconsciously counted more fish on the marine reserve transects than on the control site transects, because you expected there to be more fish in the marine reserve. To test this, we randomly split you into two groups. One group were told which transects were in reserves and which were in controls. One group weren't.

Unfortunately, in order to properly test our hypothesis, we could not provide you with all of these details prior to your participation. This ensures that your reactions in this study were spontaneous and not influenced by prior knowledge about the purpose of the study. If we had told you the actual purposes of our study, your ability to estimate fish counts could have been affected. We apologise for misleading you, but we believe this was the only way to examine the processes that are the object of our research. In designing this study, we took care to minimise any possible risks or discomforts that might be related to the deception.

Confidentiality

Now that you understand the true nature of our study and are fully informed, you have the chance to refuse the use of the data we collected from you. You are free to ask us not to use your data in our study analysis. As explained earlier, the information you've provided is completely confidential and will only be used for research purposes. It will not be used in a manner which would allow identification of your individual responses.

Because this experiment is ongoing, we request that you not share the true nature and purpose of this experiment with others who might potentially participate in our study.

Consent

If you have any questions about this research you may ask them now, or contact me, Steve Rocliffe on 07843 245 701 or sr588@york.ac.uk.

If you agree to allow us to use the data, please sign this form below. You may keep the other copy of this form for your future reference.

I have read this debriefing form and I agree to allow the use of my data for research purposes.

____

Name (printed)

____

Signature

____

Date

APPENDIX E: MARINE MANAGED AREAS IN THE WESTERN INDIAN OCEAN

Table B1: Marine Protected Areas (level 1 and 2) in the Western Indian Ocean

Country	Marine protected area	IUCN Category	Date established	Area (km^2)
Comoros	Mohéli [6]	II	2001	404.0
Îles Éparses	Glorieuses [6]	IV	2012	43,000.0
	Ile Tromelin Réserve Naturelle [6]	IV	1975	
	Ilot d'Europa Réserve Naturelle [6]	IV	1975	
	Ilot de Bassas da India Réserve Naturelle [6]	IV	1975	
Kenya	Diani	VI	1995	75.0
	Kisite	II	1978	28.0
	Kiunga	VI	1979	250.0
	Malindi	II	1968	6.3
	Malindi-Watumu	VI	1968	245.0
	Mombasa Marine National Park	II	1986	10.0
	Mombasa Marine National Reserve	VI	1986	200.0
	Mpunguti	VI	1978	11.0
	Watamu	II	1968	10.0
Madagascar	Kirindy Mitea Marine	II	2010	288.4
	Masoala [1]	II	1997	100.0
	Nosy Antafana	II	1989	10.0
	Nosy Hara	Unknown	2001	1,254.7
	Nosy Tanikely	Unknown	1966	1.8
	Nosy Ve/Androka	Unknown	2009	820.8
	Sahamalaza-Nosy Radama	Unknown	2001	127.6
	Réserve de la Biosphère du Tulear	NA	2003	

Mauritius	Anse aux Anglais	Unknown	2007	1.5
	Balaclava	II	2000	4.9
	Black River	IV	2000	8.0
	Blue Bay	Unknown	2000	3.5
	Grand Bassin	Unknown	2007	14.1
	Grand Port	IV	2000	18.3
	Passe Demi	Unknown	2007	7.2
	Port Louis	IV	2000	3.3
	Poste Lafayette	IV	2000	2.8
	Poudre d'Or	IV	2000	25.4
	Riviere Banane	Unknown	2007	1.5
	SEMPA	Unknown	2009	43.0
	Trou d'Eau Douce	IV	2000	5.7
Mayotte	Mayotte [2]	Unknown	2010	68,381.0
Mozambique	Bazaruto [3]	II	2001	1,430.0
	North Quirimbas	Unknown	2008	212.0
	Ponta do Ouro [4]	Unknown	2009	678.0
	Primeiras and Segundas	Unknown	2012	10,409.3
	Quirimbas	Unknown	2002	1,522.0
	Vilanculos	Unknown	2000	300.0
Réunion	Réunion	IV	2007	35.0
Seychelles	African Banks	Ib	1987	8.3
	Aldabra	Ia	1981	142.0
	Anse Faure Shell Reserve	NA	1987	1.1
	Aride Island	Ia	1973	0.7
	Baie Ternay	II	1979	0.9
	Cousin Island	Ia	1975	0.0
	Curieuse	II	1979	12.8
	Ile Cocos, Ile La Fouche, Ilot Platte	Unknown	1997	1.7
	La Digue Shell Reserve	NA	1987	1.6
	North East Point Shell Reserve	NA	1987	3.0
	Port Launay	II	1979	1.5
	Praslin Shell Reserve	NA	1987	1.7
	Silhouette	II	1987	16.6
	Ste. Anne	II	1973	10.0

South Africa	Aliwal Shoal	IV	2004	124.7
	Maputaland [5]	IV	2000	385.2
	St. Lucia [5]	IV	2000	442.7
	Trafalgar	IV	2000	8.3
Tanzania	Bongoyo Island	II	1975	7.3
	Fungu Yasini	II	1975	7.5
	Kiwengwa	Unknown	2000	17.5
	Mafia Island	VI	1995	615.0
	Maziwe Island	II	1981	2.6
	Mbudya Island	II	1975	8.9
	Mnazi Bay-Ruvuma Estuary	VI	2000	430.0
	Nyororo, Shungumbili and Mbarakuli	Unknown	2007	
	Pangavini Island	II	1975	2.0
	Saadani	Unknown	1969	70.0
Tanzania – Zanzibar	Chumbe Island Coral Park (CHICOP)	Ia	1994	0.3
	Pemba Channel (PECCA) [6]	VI	2005	1,000.0
	Mnemba Island-Chwaka Bay (MIMCA)	VI	2002	0.2
Total	74 MPAs			133,273

1. Includes 4 MPAs: Tampolo, Masoala-Ambodilaitry, Tanjona and Nose Mangabe
2. Includes Saziley, Passe and N'Gouja
3. Originally gazetted 1971 with 600km^2 marine area; extended in 2001
4. Includes Ilhas da Inhaca e dos Portugueses
5. Part of iSimangaliso Wetland Park
6. SP and MS had in-depth and more recent experience of these 6 sites and updated the effectiveness ratings from Burke et al [1] accordingly. Mohéli was downgraded from “Effective” to “Partly Effective”; Pemba channel was moved from “Unrated” to “Not Effective”; Glorieuses was moved from “Unrated” to “Partly Effective”; and the other three Îles Éparses reserves (Ile Tromelin, Ilot d’Europa and Ilot de Bassas da India) were moved from “Unrated” to “Effective”.

Table B2: Locally Managed Marine Areas (level 3 and 4) in the Western Indian Ocean

Country	Marine protected area	Date established	Area (km^2)
Kenya	Bureni	2010	0.52
	Jimbo	2011	7.8
	Kanamai	2011	0.22
	Kibuyuni	2010	25.1
	Kiweni	2010	3
	Majoreni	2011	23.5
	Mkwakwani/Tradewinds	2009	0.118
	Mkwiro	2011	8.1
	Msambweni	2011	0.46
	Shimoni	2008	3.5
	Tiwi (Nyari)	2009	0.125
	Vanga	2008	28.3
	Wasini	2008	8.6
	Kuruwitu	2006	0.29
Madagascar	Ambodiforaha	2011	3.1
	Ambodimangamaro	2012	1.5
	Amboditangena	2009	1.91
	Ambodivae bae	2009	200
	Ambohibola	2008	25
	Ambondrolava Mangroves	2008	6
	Analanjahana	2011	2.2
	Anandrivola	2012	1.5
	Andakatombaka	2012	1.5
	Aniribe	2011	2.28
	Ankarea (Mistio - Tsarabanjina)	2011	1736.9
	Ankivonjy (Bay de Russes)	2010	1966.66
	Antsirakivolo	2009	1.86
	Beheloke	2008	25
	Belo Sur Mer	2009	200
	Hoalampano	2012	1.5
	Imorona	2009	2.09
	Itampolo	2008	25

	Mahasoa	2011	2.48
	Maintimbato	2009	1.91
	Manjaboaka	2009	240
	Maromena/Befasy	2008	25
	Nosy Ve	2006	200
	Ranobe	2007	243
	Rantohely	2009	7.24
	Seranambe	2012	1.5
	Soariake	2008	750
	Tahosoa	2008	50
	Tampolo (Region Cap Est)	2011	3.43
	Tanandava	2011	2.39
	Teariake	2011	260
	Vatolava	2011	1.14
	Velondriake	2006	640
	Vohitralanana	2009	3.18
Mozambique	Vamizi Marine Sanctuary [1]	2008	18
Tanzania	Boma-Mahandakini [2]	2001	145
	Boza-Sangea [2]	1997	396
	Deepsea-Boma [2]	2000	377
	Mkwaja-Sange [2]	2000	388.5
	Mtang'ata [2]	1997	96.5
	Mwarongo-Sahare [2]	2000	195.5
	Njisopoja [3]	2011	884
	Mbwekieki [3]	2011	208
	Kimsa [3]	2011	306
	Mchimchnumya [3]	2011	356
	Dokichunda [3]	2011	478
	Jojibaki [3]	2011	266
Tanzania --Zanzibar	Menai Bay (MBCA)	1997	470
Total	62 LMMAs		11,329

1. Part of North Quirimbas
2. Tanga collaborative Management Areas
3. RuMaKi project

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