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Assessing the impact of coral reef community management in the Kingdom of Tonga

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Thesis submitted by

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Acknowledgments

This PhD began as a voyage of discovery while sailing through the islands of Tonga in 2016. During my first year in the Kingdom I was surprised to learn that very little information was available on the health status of the country's coral reef ecosystem. Furthermore, during the early period of my partnership with the Tongan Ministry of Fisheries, I discovered that despite this lack of critical baseline data, there was nevertheless a group of incredibly dedicated Tongans working towards improving management at the local level. The aim of this thesis from the start has been to support their work.

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Sketch by Dominique Serafini, former comic artist for Equipe Cousteau of
S.V. Chaveta while conducting surveys in Vava'u

Statement of contribution of others

This thesis was supervised by **Prof. Geoffrey Jones**, **Prof. Bob Pressey, Dr. Tom Bridge and Dr. Georgina Gurney**. All supervisors contributed to the development of ideas and provided guidance, intellectual input and editorial assistance. I developed the research questions, collected or collated and analysed all the data, and wrote all the chapters. All research activities completed in this thesis were conducted in accordance with James Cook University Animal Ethic Guidelines (permit approval A2454).

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Authorship of chapters for publications is shared with members of my thesis committee (Geoffrey Jones, Bob Pressey, Tom Bridge and Georgina Gurney), key staff at the Tongan Ministry of Fisheries (Tuikolongahau Halafihi and Siola'a Malimali) as well as various contributing authors (Daniela Ceccarelli, Alexandra Dempsey, Sophie Gordon, Sam Purkis, Jason Sheehan, Paul Southgate, Karen Stone, Rebecca Weeks and Mathew Wyatt). Specifically, in Chapter 2 R. Pressey, G. Gurney and R. Weeks contributed to study design and editorial assistance. In Chapter 3, A. Dempsey and S. Purkis contributed to data layers and editorial assistance. S. Malimali and T. Halafihi contributed to in country support and editorial assistance. S. Gordon and P. Southgate contributed to development of the ideas, formatting and editorial assistance. T. Bridge, R., Pressey and G. Jone provided intellectual input and editorial assistance. In Chapter 4, K. Stone and D. Ceccarelli provided data and editorial assistance. M. Wyatt provided assistance with benthic annotations. S. Gordon contributed to field work, idea development and editorial assistance. S. Malimali and T. Halafihi contributed to in country support and editorial assistance. T. Bridge, R., Pressey and G. Jone provided intellectual input and editorial assistance. In Chapter 5, J. Sheehan provided interllectual input and editorial assistance. S. Malimali and T. Halafihi contributed to in country support and editorial assistance. T. Bridge, R., Pressey and G. Jone provided intellectual input and editorial assistance. In Chapter 6, K. Stone and D. Ceccarelli provided data and editorial assistance. S. Malimali and T. Halafihi contributed to in country support and editorial assistance. T. Bridge, R., Pressey and G. Jone provided intellectual input and editorial assistance. In Chapter 7, S. Malimali contributed to in country support and editorial assistance. T. Bridge, R., Pressey and G. Jone provided intellectual input and editorial assistance.

General Abstract

Humanity's increasing environmental footprint is currently driving a period of unprecedented change for both natural ecosystems and human societies. This is particularly true for coral reefs, which in recent decades have experienced widespread devastation from extreme temperatures associated with climate change, as well as local stressors such as overfishing and pollution. These stressors are only predicted to increase through time, with ramifications for the millions of people who depend on coral reef systems for food security and livelihoods. Finding solutions that can balance human needs with the conservation of nature is critical to maintaining coral reef social ecological systems (SES) in the future. Managing coral reef ecosystems for the future will include a range of tools for mitigating local stressors in order to improve ecosystem resilience and food security.

In recent decades, local approaches to marine management, known variously as *community-based management*, *co-management* or *locally managed marine areas*, have become increasingly popular. These approaches are based on the premise that coral reef management primarily focuses on changing patterns of behaviour at the local level, and therefore will be most effective if resource users are actively involved in the process. Furthermore, determining the best balance of incentives to achieve engagement in conservation is critical, as resource users may have more immediate concerns such as finding food or making a living. Local management can also involve multiple strategies, including no-take marine protected areas (MPAs), periodically harvested areas, and access restrictions. Determining the best approach, or combination of approaches to use on coral reef SESs is a fundamental role of scientists, managers and conservationists.

Assessing the efficacy of management involves being able to determine its impact, or the difference an action makes over and above the counterfactual condition of no action. While many studies have examined the effectiveness of marine management, few have been able to quantify impact, instead focusing on measures such as inputs (e.g. cost), outputs (e.g. number or extent of protected areas) or outcomes (e.g. habitat representation). The widespread reporting of targets not directly associated with impact has led to the now global trend of residual conservation, where conservation or management actions are situated so as to minimize overlap with areas of high resource value.

The overarching aim of this thesis was to understand the ecological impacts of community-based marine management on coral reef social-ecological systems. I use Tonga and its Special Management Area (SMA) program as a case study by which to address this objective. The SMA program is a dual approach to marine management, whereby communities are granted exclusive access to the marine environment adjacent to their village in exchange for making part of it a no-take

MPA. Here I show that a dual approach to marine management can yield positive impacts to natural resource management and biodiversity conservation once incentives have been provided that promote non-residual conservation.

Prior to addressing my main research objectives, I set out to answer a series of preliminary questions designed to either address specific theoretical knowledge gaps in the literature or fill in geographical knowledge gaps. **Chapter 2** addresses the question: *what is currently known about the impacts of MPAs in the South Pacific?* Here, I conducted a semi-structured review of the literature from the South Pacific to examine the ecological and socioeconomic impacts of MPAs. Specifically I asked: *i) what are the overall ecological and socioeconomic impacts of MPAs in the regions? ii) what factors are associated with positive, neutral or negative impacts? and iii) to what extent has the MPA evaluation literature from the region incorporated robust impact evaluation techniques?* From 52 identified ecological studies and eight socioeconomic studies, 42% and 72% of measured impacts were positive, respectively. The proportion of positive impacts was comparable between community-based and more centralized approaches to management, although no-take MPAs had more positive impacts than periodically harvested closures. However, few studies provided any clear consideration of factors beyond the presence of the MPA that may have confounded their results, leading to the conclusion that robust impact evaluation techniques have yet to be fully-embraced in the region.

Chapter 3, 4 and 5 then develop the framework needed to describe Tonga's coral reef social ecological system, including its socio-environmental context (**Chapter 3**), ecological status (**Chapter 4**) and approach to marine management (**Chapter 5**). For **Chapter 3** I asked *what socio-environmental spatial data is available for Tonga's coral reefs?* I compiled a marine socio-environmental spatial dataset covering Tonga's coral reef ecosystem from various global layers, remote sensing projects, local ministries and the 2016 national census. The dataset consists of eleven environmental and six anthropogenic variables spatially overlaid across the near-shore ecosystem of Tonga. The environmental variables selected include: bathymetry, coral reef density, distance from deep water, distance from land, distance from major terrestrial inputs, habitat, land area, net primary productivity, salinity, sea surface temperature, and wave energy. The anthropogenic variables selected include: fishing pressure, management status, distance to fish markets, distance from villages, population pressure, and a socioeconomic development index based on population density, growth, mean age, mean education level, and unemployment.

Chapter 4 addresses the question - *what is the present ecological status of Tonga's coral reef ecosystem and reef fish fishery?* Here I present the results from Tonga's first national coral reef monitoring expedition, in which 375 sites were surveyed across the three main island groups, Tongatapu, Ha'apai and Vava'u, to describe broad trends in the status of the countries coral reefs and reef fish fishery. I then combine this data with the spatial layers from Chapter 3 to describe the relative importance of various socio-environmental variables on key metrics of reef state. While mean live coral cover across Tonga was 18%, this showed a strong decrease with increasing latitude and was associated primarily with sea surface temperature. Reef fish species richness and density were comparable throughout Tongatapu and Ha'apai, but lower in the northern island group Vava'u, and influenced primarily by habitat-associated variables (slope and structural complexity). Target adult reef fish biomass was greatest in the central island group, and lower both near the capital in the south (Tongatapu) and in the north (Vava'u), which was negatively correlated to both increased fishing activity and population pressure. Both Chapters 3 and 4 provide crucial context and data for addressing the research aims of this thesis in subsequent chapters.

Chapter 5 introduces Tonga's SMA program and focusses on 2 questions – *i) what is Tonga's approach to marine management* and *ii) why is it broadly relevant to marine conservation?* This chapter discusses and analyzes key characteristics of the program and its background, including both the mechanisms that have motivated its successful national expansion and its ability to configure no-take reserves in areas that are considered to have high value to resource users. It demonstrates that granting communities exclusive access zones in exchange for implementing no-take reserves has encouraged conservation actions while fostering long-term relationships with resources. Ensuring that no-take reserves occur within the boundaries of exclusive access zones also entices communities to protect areas of greater extractive values than they would have otherwise. This chapter concludes that the success of this program offers a way forward in achieving national and international targets for conservation and sustainable fisheries management.

Chapters 6 and 7 use a counterfactual framework, where outcomes from management are compared to predicted outcomes if management had never occurred, to examine the ecological impacts, and potential future ecological impacts, of Tonga's SMA program. **Chapter 6** focusses on the question - *what are the ecological impacts of Tonga's oldest Special Management Areas?* Here, I conduct a rigorous impact evaluation of Tonga's dual management system by comparing the ecological status of the 14 oldest management areas to their predicted counterfactual conditions. I use statistical matching of ecological survey sites from Chapter 4 to pair managed areas with those open to fishing across 11 of the socio-environmental spatial variables developed in Chapter 3. No-take areas generally had positive impacts on the species richness, biomass, density and size of target reef fish, while exclusive access areas were similar to predicted counterfactual conditions. The latter is likely because overall fishing pressure in these areas does not change, although more fish are exploited by communities with access rights. Our findings suggest that dual management is effective at incentivizing community-based no-take areas for biodiversity conservation and resource management.

Chapter 7 expands the approach in Chapter 6, using predictive techniques to ask two questions: *i) what is the potential future impact of marine management in Tonga* and *ii) how much potential impact is lost if communities design their own no-take MPAs?* This approach compares recently implemented community-based no-take reserves to various systematic configurations aimed at maximizing impact. I estimate that the community designed MPAs provide 84% of the recovery potential of a systematic configuration with the greatest potential impact. This high potential impact results from community-based reserves being located close to villages, where fishing pressure is historically greatest, which provides strong support for local management when there is little scope for more systematic approaches.

Collectively, the different components of this thesis represent a practical and methodical approach by which to examine existing and potential future ecological impacts of local marine management. The work demonstrates that Tonga's community-based approach has yielded positive ecological impacts, and has been able to avoid issues that plague many MPAs. Furthermore, it also identifies the mechanisms by which it has done so. I show that the implementation of no-take MPAs directly adjacent to communities is preventing residual conservation from local management in Tonga, and thereby maximizing the differences between reefs within and beyond MPA boundaries. By providing exclusive access zones in exchange for implementing no-take MPAs, the SMA program has successfully identified a mechanism by which to incentivize communities and rapidly expand its reach. Developing these techniques, and discovering new solutions that are also able to balance conservation and human needs, will remain critical to improving the outlook of coral reef social ecological systems.

Publications associated with this thesis

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Smallhorn-West P, Stone K, Ceccarelli D, Malimali S, Halafihi H, Bridge T, Pressey B, Jones G (In press) Community-based management yields positive impacts for coastal fisheries resources and biodiversity conservation. *Conservation Letters* (Chapter 6)

Smallhorn-West P, Bridge T, Malimali S, Pressey R, Jones G. (2019) Predicting impact to determine the efficacy of community-based marine reserve design. *Conservation Letters.* DOI: 10.1111/conl.12602 (Chapter 7)

Smallhorn-West P, Govan H (2018) Towards reducing misrepresentation of national achievements in marine protected area targets. *Marine policy* 97, 127-129 (Appendix)

Smallhorn-West P, Garvin J, Slayback D, DeCarlo T, Gordon S, Fitzgerald S, Halafihi T, Jones G, Bridge T (2019) Coral reef annihilation, persistence and recovery at Earth's youngest volcanic island. *Coral reefs* (Appendix)

Smallhorn-West P (2020) Progress towards conserving Tonga's coral reefs. *SPC Fisheries newsletter #160* (Appendix)

Reports

Smallhorn-West P, Sheehan J, Stone K, Malimali S, Halafihi H, Matoto L, Rodriguez-Troncoso A, Bridge T, Pressey B, Jones G (2020) Kingdom of Tonga Special Management Area Report 2020. (Appendix)

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Chapter 1: General introduction

Both natural ecosystems and human societies are currently experiencing a global period of rapid change and uncertainty due to increasing human pressures (Diaz et al. 2019). Coral reefs, which are among the most diverse ecosystems on earth, have now also become one of the most threatened (Hughes et al., 2017a; Pandolfi et al., 2011). Globally, coral reef ecosystems are in decline due to large-scale impacts such as environmental extremes associated with human-induced climate change (Hughes et al., 2017b; Hughes et al., 2019), compounded by perennial disturbances such as overfishing and pollution (Brodie and Pearson, 2016). Not only are these disturbances a concern for the ecosystems themselves, but also for the millions of people worldwide who depend on them for food security and to support their livelihoods (Mills et al., 2011; Pauly, 2006). Furthermore, projected increases in the human population suggests that, not only will these disturbances increase in decades to come, but more and more people will be dependent on degrading coral reef ecosystems, further exacerbating the problem. Determining how to balance human needs with the conservation of nature on a changing planet is a fundamental role of scientists, managers and conservationists in the 21st century.

Achieving sustainable futures for the world's coral reefs and the people who depend on them requires managing coral reefs as linked social ecological systems (SESs) (Cinner et al., 2012; Gurney et al., 2019). This approach emphasizes the fundamental interdependency of social and environmental change. It recognizes the importance of documenting and understanding interactions and feedbacks between social and ecological components of the systems (Cumming et al. 2020; Fischer et al., 2015). It also involves recognizing the importance of achieving multiple objectives spanning both conservation outcomes (e.g. biodiversity or ecosystem health) and socio-economic well-being (e.g. food security or community empowerment). Indeed, in many parts of the developing world the concepts of conservation and sustainable use are inseparable (Govan et al., 2009). Numerous studies have now called for an SES approach to conservation and management of coral reefs (Hughes et al., 2017a; Bellwood et al. 2019), which is also increasingly reflected in international environmental policy, such as the Convention on Biological Diversity and the United Nations Sustainable Development Goals (Gurney et al., 2019).

1.1 Marine Management

Acknowledging the international scale of many threats to coral reefs (i.e. climate change), management approaches encompass a suite of tools for mitigating local stressors (e.g. overfishing and pollution) in order to boost ecosystem resilience (Mellin et al., 2016) and food security (Mascia, et al., 2010). In many developed countries, management often takes the form of a centralized approach, with natural resource management or biodiversity protection undertaken by a central governing authority. This style of management was notably espoused in 1968 by Garret Hardin, who asserted that state control of natural resources was necessary to prevent the tragedy of the commons, whereby individual users sharing a common resource (i.e. fishery) and acting in their own self-interest behave contrary to the common good (Hardin, 1968). A centralized approach in theory could facilite systematic conservation planning (Margules and Pressey, 2000) given jurisdictional mandates to plan or coordinate actions to maximize conservation impact. However, in practice, particularly in developing countries, centralized management has often failed because it does not incorporate local stakeholder objectives, resulting in low support, compliance and ultimately "paper parks" (Gaymer et al., 2014).

In contrast to centralized management, community-based approaches are becoming increasingly popular in developing countries with limited governance resources or in areas with strong local tenure rights (Govan et al., 2006; Govan, 2009). There has been a growing realization that, since the management of coral reefs primarily focuses on changing patterns of behaviour at the local level, resources will be more effectively managed if resource users are actively involved in the process (Cohen, 2013; Pomeroy, 1995). This has been particularly the case for coral reef management in South-East Asia and the South Pacific (Johannes, 2002; Govan, 2009). Community-based management ensures that those who are most affected by management (e.g. fishers and other resource users) will be involved in making management decisions, which hence generally have greater local support than centralized approaches. This was first highlighted by Elinor Ostrom, who identified that solutions to managing common pool resources with multiple stakeholders can be successfully developed on a local scale (Cox and Arnold, 2010; Ostrom, 1990). Here, I define *community-based marine management* as natural resource or biodiversity management by, for and with the local community (Western and Wright 1994). Importantly, community-based and centralized approaches are not mutually exclusive, and in practice co-management arrangements are more common than those relying exclusively on local input (Cinner and Huchery, 2014; Wamukota et al. 2012). *Co-management* refers to situations where communities share responsibilities for making and enforcing natural resource management rules with governments, civil society, and/or academia (Cinner et al., 2012). *Community-based marine management* and *co-management* are also both often synonymous with *locally managed marine areas* (LMMAs), which, particularly in the South Pacific, are defined as areas of nearshore waters being actively managed by local communities or in collaboration with local government/partner organizations to achieve local objectives (Govan et al. 2006; Govan et al. 2009; Jupiter et al., 2014). Notably, given that local management actions generally prioritize local goals, such as maintaining target fisheries, and might not target broad conservation objectives, co-management is seen as a way to reconcile the two kinds of goals and support ecosystem-based approaches to management (Berkes, 2012; Cohen, 2013).

Marine protected areas (MPAs) represent a key strategy by which coral reefs are managed, focusing on objectives for both conservation (e.g. increasing biodiversity) and socioeconomic (e.g. improving fish stocks) concerns (Govan and Jupiter, 2014; Jupiter et al., 2014). Permanent closures, or no-take MPAs, are employed worldwide in order to protect species and habitats within their boundaries (Lester et al., 2009), enhance fisheries through adult spillover (Halpern et al. 2010; Russ and Alcala 1996) and larval export (Harrison et al 2012; Almany et al., 2017), and increase overall ecosystem resilience (Mellin et al. 2016). Global commitments to conservation have resulted in large MPA spatial targets, such as Aichi target 11, which calls for 10% of marine areas to be effectively conserved through protected areas by 2020 (Anon, 2013). While no-take MPAs have increased in extent, in certain instances they might not be appropriate due to conflicts with resources users (Christie, 2004; James and Dearden, 2014). Managers and conservationists often argue that biodiversity values and long-term food security outweigh immediate requirements for resource use (Hutton and Leader-Williams, 2014). However, offering long-term assurances of increased food security and ecosystem health might not always be important for people for whom finding food or making a living are immediate concerns. In these circumstances, two options are available to potential managers: either identify incentives by which to promote the expansion of no-take MPAs that adequately compensate loss to local stakeholders; or implement alternate management strategies other than no-take MPAs.

Jupiter et al. (2014) outlined six strategies that could be used in the community context to achieve a range of ecological or socioeconomic outcomes typically expected from coral reef management. While these included no-take MPAs, they also included periodically harvested closures, catch restrictions on species, gear or access, and alternative livelihood strategies. Periodically-harvested closures are a common but highly variable management tool that is used to protect target species between pulse harvest events (Cohen & Foale, 2013; Cohen & Alexander, 2013). They have been used extensively in the South Pacific and are often seen as a more feasible option than no-take reserves as they minimize the cost incurred by fishers from lost access to resources (Carvalho et al., 2019; Goetze et al., 2017). However, due to often short closing cycles compared to recovery times of target species, they might not achieve meaningful ecological outcomes, particularly for longer-lived species (Goetze et al., 2016). Likewise, access restrictions are a management strategy that has been employed with varying degrees of success in the local context (Gelcich et al., 2017). For example, in Chile territorial use rights for fisheries have been implemented for decades as a co-management approach that has achieved add-on conservation benefits (Gelcich et al., 2008; Gelcich et al., 2012). However, while the ability to restrict access is a foundation of effective resource governance (Ostrom, 1990, Jupiter et al., 2014), it might not change the volume harvested, just who harvests it (Polunin 1984). Importantly these strategies are not mutually exclusive and different combinations can be applied concurrently to either maximize their effectiveness or to act as incentives when management may not be supported at the local level. Relatively little research has gone into examining the efficacy of different combinations of approaches (but see Villasenor-Derbez et al., 2019 for a description of TURF-reserves).

In recent decades, there has been a global movement to expand both MPAs and other forms of coastal marine management, with co-management approaches proliferating in the developing world (Mills et al., 2019). The expansion of local approaches is particularly evident in the South Pacific, where the rate of uptake has been unprecedented (Govan et al., 2009; Govan, 2009; Johannes, 2002), although non-independent states (e.g. French Polynesia and New Caledonia) often use centralized management. The main drivers for this rapid uptake are generally perceived threats to food security and community desire to improve livelihoods, as well as a strong history of customary tenure (Johannes, 2002; Jupiter, 2017). The South Pacific is also a region where people are highly reliant on their marine ecosystems, and thus particularly vulnerable to both large-scale and local stressors. Given its large geographic area and strong dependence on marine resources, this region is ideally suited for research into the efficacy of various approaches to coral reef management. In parallel, the number of studies examining the processes, design principles and outcomes of marine management have also grown (e.g. Edgar et al., 2014; Lester et al., 2009). However, the ultimate success of any strategy should not be based on its rate of expansion or support, but on whether it achieves explicit ecological and/or socio-economic objectives. A key question is therefore *how do we determine the efficacy of various approaches to marine management?* or, more broadly, *how do we measure success in marine conservation?*

1.2 Impact

The fundamental premise of both conservation and management is to make a difference. Retrospectively this means being able to determine accurately the difference an action has made, and during the planning phase, ensuring potential actions will make as much positive difference as possible. However, despite vast efforts by both researchers and managers at minimizing the loss of biodiversity and ecosystem services, problems persist. While acknowledging the existence of large-scale institutional barriers, many problems in conservation are also partly the result of misaligned targets (Ferraro and Pressey, 2015; Pressey et al., 2015). This is a systemic problem in conservation science whereby managers and researchers continually measure progress with criteria that are unrelated to making a difference. Broadly, there are three types of conservation measures – inputs, outputs and outcomes, which are frequently quantified but often misused (Mascia et al., 2017; Pressey et al., 2017). Inputs are the investments in management or conservation, such as raw materials, money, staff or time. Outputs are the concrete, countable products of one or more conservation actions, such as the number or total extent of protected areas (e.g. CBD Aichi target 11). Outcomes are the assumed short- and medium-term effects of an intervention's output, such as those related to representation of species or ecosystems, or the conditions inside a management area either at one or multiple points in time. However, these three measures are unable to quantify differences that have arisen from conservation or management actions. Pressey et al. (2017) suggested that this mismeasure in conservation science is analogous to medical research achieving stated targets but failing to reduce human suffering and death.

The large-scale mismeasure of conservation progress has accompanied, and to some extent resulted in, the well-established trend of residual conservation, whereby management areas are often located and configured to minimize costs and conflicts with resource users. For example, two global reviews of terrestrial protected areas clearly demonstrated that most protected areas are biased towards areas unlikely to face land conversion pressures even in the absence of protection (Joppa and Pfaff, 2009, 2011). This bias was also confirmed in many parts of the world, including Costa Rica (Andam et al., 2008) and Brazil (Vieira et al. 2019), where the potential impacts of protected areas on deforestation were reduced substantially by protected areas being far from roads and on land unsuitable for extractive use. Likewise, in the ocean many MPA targets are based on perverse measures that drive residual conservation (Devillers et al., 2015). For example, Failler et al., (2019) stated that "although the shoreline is an extremely important area to conserve, it is essential to develop large offshore MPAs in order to ensure the achievement of the Aichi Target 11 [for West African countries]".

While these trends are partly driven by strong political motivations and international targets, often conservation planners equally support them. For example, Devillers et al. (2015) found that the rezoning of Great Barrier Reef, arguably the most managed reef ecosystem on earth and which in 2004 had an increase in no-take MPAs from 4.6% to 33.3%, represented a less than 5% change in business as usual for trawling activities. In planning various MPA configurations in the western Indian Ocean, McClanahan et al. (2016) prioritized areas that minimized recovery time to carrying capacity, thereby also explicitly minimizing any difference protection was likely to make. Underlying these findings is the imperative to minimize opportunity cost, which is partly driven by conservation software such as Marxan (Watts et al., 2009). However, given that conservation and management are fundamentally about changing human behaviour, focusing on minimizing cost is likely to perpetuate residual conservation. Instead, conservation science should aim to frame strategies based on differences made to current or potential future human actions while acknowledging the need to acheive broader societal goals.

Impact is defined as the effects of an action on one or more intended (or unintended) outcomes, over and above the counterfactual condition (Ferraro, 2009) of no action or a different action (Mascia et al. 2017; Pressey et al., 2015). Importantly, determining impact involves a counterfactual framework that estimates what the outcomes would have looked like in the absence of an intervention (Adams et al., 2019). Impact evaluations require a clear understanding of contextual variables that can confound results, leading to over- or underestimations of impact. When applied in the conservation sector, impact evaluation focuses on disentangling the effects of management or policy interventions (e.g. protected areas) on variables of interest (e.g. biodiversity) from broader changes in the environment (Ahmadia et al., 2015). While acknowledging the vast literature on MPA outcomes (e.g. Edgar et al., 2014; Lester et al., 2009), few have incorporated counterfactual thinking into their analyses (but see Ahmadia et al., 2015; Gill et al., 2017). For example, commonly used Before-After-Control-Intervention (BACI) and Control-Intervention (CI) methods for evaluating existing management areas typically do not account for confounders (or do so haphazardly). Instead, these sampling designs assume that there is no difference in biophysical or social conditions in treated (protected) and untreated (control) areas or that any differences in conditions do not influence whether an area is selected for treatment or what outcomes are observed and that this holds through time (Adams et al. 2019; Kerr et al., 2019). By evaluating the impacts of marine management on conservation and/or socioeconomic outcomes, we can inform evidence-based policy and improve predictions of future impacts (Ferraro and Pattanayak, 2006; Fraser et al., 2019)

1.3 Research gaps and thesis aims

At the outset, it is important to acknowledge the large body of literature on co-management, community-based management and LMMAs, as well as outcomes associated with MPAs in general. Community management is often promoted, particularly by NGOs in the Pacific, as critical for addressing ecological and socioeconomic issues (Cohen, 2013; Leisher et al. 2007). However, approaches to local marine management have yet to be viewed through the lens of impact and counterfactual analysis, which is necessary to accurately gauge their efficacy. While community engagement in designing and implementing management is considered critical to successful co-management, it does not by right imply positive conservation or socioeconomic impacts. A number of key research questions therefore exist in this context:

- To what extent has the current MPA evaluation literature, in particular the community management literature, embraced impact evaluation techniques and counterfactual framing?
- Are local management initiatives able to achieve positive impacts for both biodiversity conservation and natural resource management?
- Given that the scaling of community-based approaches to marine management are generally more ad hoc than those that are centrally managed, does local management fall into the same traps associated with residual conservation and minimizing opportunity costs?

- Given that centrally managed MPAs may be able to more actively incorporate systematic conservation planning into their design, how much impact is lost by implementing community based approaches?
- While there are well-established approaches for evaluating existing management areas (e.g. Ahmadia et al., 2015; Ferraro, 2009; Ferraro and Pattanayak, 2006; Gill et al., 2017; Joppa and Pfaff, 2011), is it possible to use predictive techniques to quantify the potential future impact of management?

The overarching aim of this thesis is *to understand the ecological impacts of community-based marine management on coral reef social-ecological systems*. I use Tonga as a case study by which to address this objective. Tonga is a small island nation in the South Pacific that is heavily reliant on the health of its marine ecosystem. Over the past decade, the Tongan Ministry of Fisheries has scaled up a community management approach, termed the Special Management Area (SMA) program, to a national level. The SMA program is a dual approach to local marine management whereby communities are granted exclusive access to the marine environment adjacent to their villages in exchange for making part of these areas no-take MPAs. Tonga was selected as a case study because: i) its small geographic size made it feasible to conduct research at a national level, ii) the SMA program is currently the only form of marine spatial management actively implemented in the country, and iii) I had an existing working relationship with the Tongan Ministry of Fisheries, who was able to support and facilitate a critical appraisal of its local management program. In addition to the outputs of this thesis, this work also provides baseline ecological and socio-environmental data critical to the effective management of Tonga's coral reefs. The Tongan Government currently lacks both the capacity and resources to monitor its coral reef system, as well as to determine the efficacy of its approach to management. For this reason, the results of this thesis are also being shared directly with the Tongan government, Ministry of Fisheries and the general public, through publications, meetings and various reports.

This thesis is organised as a series of chapters written for publication in peer-reviewed journals, but reformatted to fit a thesis structure. I developed the research questions, collected or collated and analysed all the data, and wrote all the chapters. Authorship of chapters for publications is shared with members of my thesis committee (Geoffrey Jones, Bob Pressey, Tom Bridge and Georgina Gurney), key staff at the Tongan Ministry of Fisheries (Tuikolongahau Halafihi and Siola'a Malimali) as well as various contributing authors (Daniela Ceccarelli, Alexandra Dempsey, Sophie Gordon, Sam Purkis, Jason Sheehan, Paul Southgate, Karen Stone, Rebecca Weeks and Mathew Wyatt). All references and additional supporting information are provided in the appendices. I have also written three additional journal articles and a national report that are directly relevant to this thesis and also provided in the appendices.

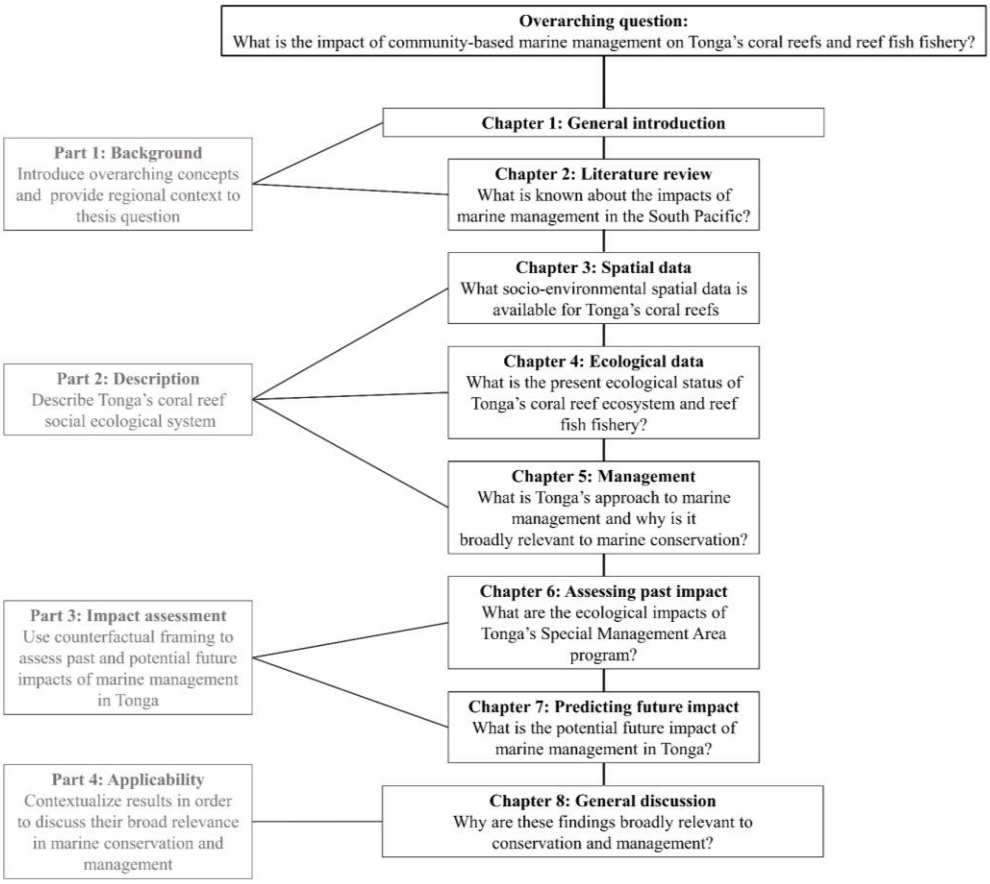


Figure 1.1. Conceptualization of thesis

Prior to addressing my overarching research aim, it was important to answer a number of preliminary questions designed to address either specific theoretical gaps in the literature or fill in geographical knowledge gaps (Figure 1.1). **Chapter 2** describes what is currently known about the ecological and socioeconomic impacts of MPAs in the South Pacific. **Chapters 3, 4 and 5** describe the context of Tonga's coral reef social ecological system, including its socio-environmental context (**Chapter 3**), ecological status (**Chapter 4**) and approach to marine management (**Chapter 5**). Within these chapters, I present results collected from a three-year collaboration with the Tongan Government and Ministry of Fisheries, in which I developed and completed the first time point of a national coral reef monitoring program. **Chapters 6 and 7** then use counterfactual framing, with which I compare outcomes from management to predicted outcomes if management had never occurred, to examine the past and potential future ecological impacts of Tonga's SMA program.

Chapter 2 addressed the question: *what is currently known about the impacts of marine protected areas in the South Pacific?* Here, I conducted a semi-structured review of the literature from the South Pacific examining both ecological and socioeconomic impacts of marine protected areas. The review asked *i) what are the overall ecological and socioeconomic impacts of MPAs in the* *region? ii) what factors are associated with positive, neutral or negative impacts?* and, *iii) to what extent has the MPA evaluation literature from the region incorporated robust impact evaluation techniques?* I developed the concept of the review, conducted the literature search, analysed the data and wrote the chapter. Georgina Gurney, Bob Pressey and Rebecca Weeks assisted with conceptualizing the review and editing the chapter.

Chapter 3 asked *what socio-environmental spatial data are available for Tonga's coral reefs?* This chapter compiles a marine socio-environmental dataset covering Tonga's near-shore marine ecosystem from various global layers, remote sensing projects, local ministries and the 2016 national census. It provides eleven environmental and six anthropogenic variables summarized in ecologically relevant ways, spatially overlaid across the near-shore marine ecosystem of Tonga. I developed the concept for this chapter, collated existing layers or used available data to create new layers and wrote the chapter. Alexandra Dempsey and Sam Purkis provided data layers. All authors provided advice on concept development and assisted with editing the manuscript.

Chapter 4 sought to answer the overarching question: *what is the present ecological status of Tonga's coral reef ecosystem and reef fish fishery?* I present the results of Tonga's first national coral reef monitoring expedition, in which 375 sites were surveyed across the three main island groups, to describe broad trends in the status of the country's coral reefs and reef fish fishery. I then combine these data with the spatial layers from Chapter 3 to describe the relative importance of various socio-environmental variables on key metrics of reef condition. I developed the concept for this chapter, designed the data collection, conducted most ecological surveys (n= 279/375), analysed the data and wrote the chapter. Daniela Ceccarelli and Karen Stone conducted the remaining ecological surveys. Mathew Wyatt assisted with machine annotations of benthic photoquadrats. All authors provided advice on concept development and assisted with editing the manuscript.

Chapter 5 introduced Tonga's Special Management Area (SMA) program by asking the question: *What is Tonga's approach to marine management and why is it broadly relevant to marine conservation?* It provides key characteristics of the program and its background, including the mechanisms that have motivated its successful national expansion. It also uses spatial data drawn from Chapter 3 to demonstrate how the SMA program has configured no-take MPAs in areas considered to have high value to resources users, and thereby avoiding residual conservation. I developed the concept for this chapter, analysed the data and wrote the chapter. All other authors provided advice on concept development and assisted with editing the manuscript.

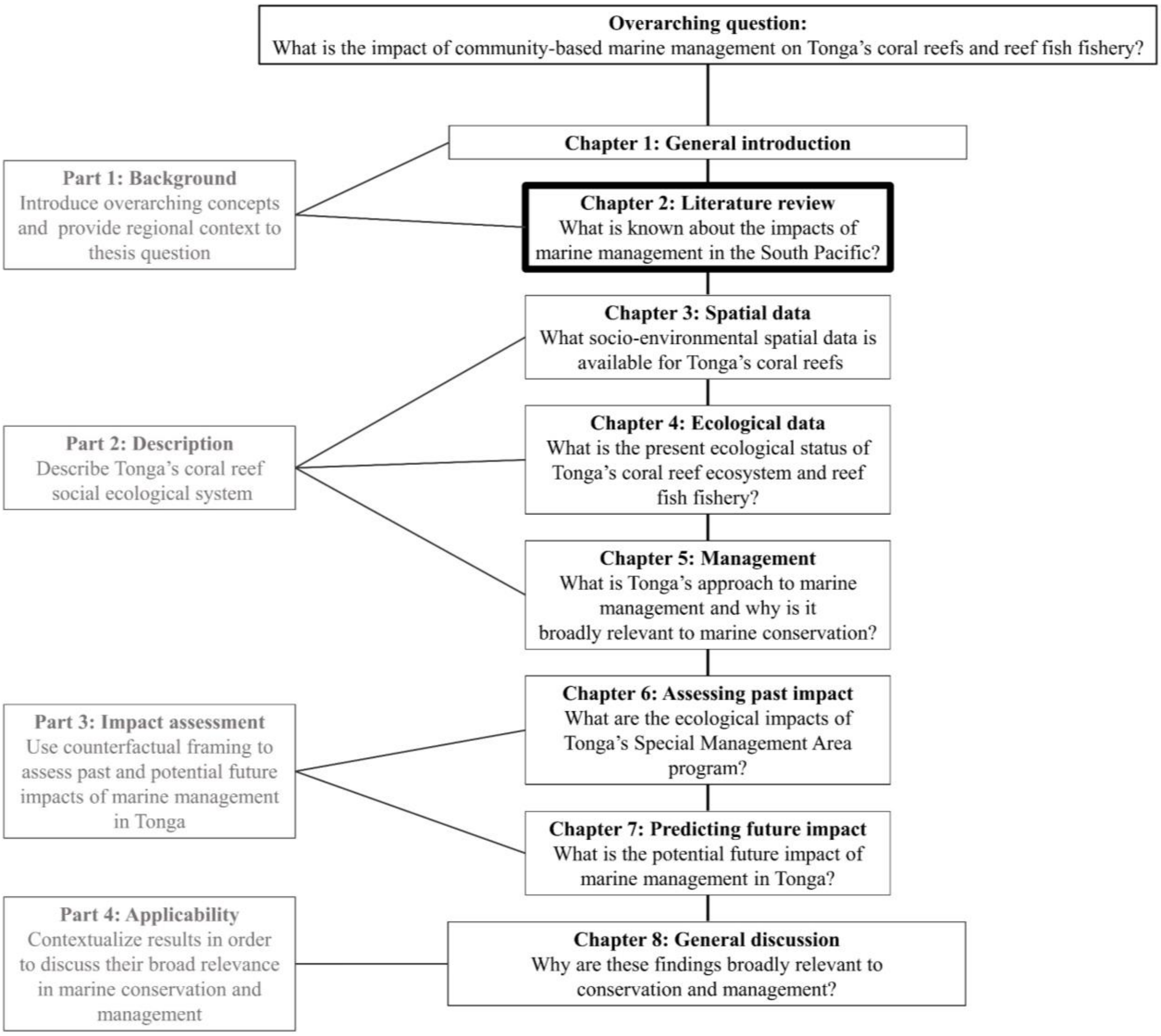
Chapter 6 looked at the past to address the question: *What are the ecological impacts of Tonga's Special Management Area program?* I compare the current ecological status of the 14 oldest management areas in Tonga to their predicted counterfactual conditions. This chapter uses statistical matching of ecological survey sites from Chapter 4 to pair managed areas with those open to fishing across 11 of the socio-environmental spatial variables developed in Chapter 3. I developed the concept for this chapter, designed, coordinated and conducted data collection, analysed the data and wrote the chapter. Karen Stone and Daniela Ceccarelli collected some data. All authors provided advice on concept development and assisted with editing the manuscript.

Chapter 7 followed from Chapter 6, but instead uses predictive techniques to look forwards and ask: *What is the potential future impact of marine management in Tonga?* Here I predict conservation impact to compare recently implemented community-based no-take reserves to various systematic configurations aimed at maximizing impact. Specifically I asked: *How much potential impact is lost if communities design their own no-MPAs?* This chapter uses socio-environmental variables from Chapter 3 and ecological surveys from Chapter 4. I developed the concept for this chapter with assistance from Bob Pressey. I analysed the data and wrote the chapter. All authors assisted with editing the manuscript.

Chapter 8 provides a synthesis of the previous chapters and contextualizes the results with concepts introduced in Chapter 1. In particular, it reflects on the importance of impact, impact evaluation and residual conservation in marine management. I also discuss limitations of my research and suggest avenues for future research.

Chapter 2: Ecological and socioeconomic impacts of marine protected areas in the South Pacific: assessing the evidence base

P. Smallhorn-West, R. Weeks, G. Gurney and R. Pressey



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2.1 Abstract

Marine protected areas (MPAs) in the South Pacific have a unique history that calls for a regional-scale synthesis of MPA impacts and the factors related to positive ecological and socioeconomic change. However, recommendations of best approaches to MPA implementation can be made only when evaluation techniques are sound. Impact evaluation involves quantifying the effects of an intervention over and above the counterfactual of no intervention or a different intervention. Determining the true impact of an MPA can be challenging because additional factors beyond the presence of an MPA can confound the observed results (e.g. differences in ecological or socioeconomic conditions between MPA and control sites). While impact evaluation techniques employing counterfactual thinking have been well developed in other fields, they have been embraced only slowly in the MPA evaluation literature. We conducted a structured literature search and synthesis of MPA evaluation studies from the South Pacific to determine: i) the overall ecological and socioeconomic impacts of MPAs in the region, ii) what factors were associated with positive, neutral, or negative impacts, and iii) to what extent the MPA evaluation literature from the region has incorporated counterfactual thinking and robust impact evaluation techniques. Based on 52 identified studies, 42% of measured ecological impacts were positive. While 72% of socioeconomic impacts were positive, these were from only eight studies. The proportion of positive impacts was comparable between community-based and centrally governed MPAs, suggesting that both governance approaches are viable options in the region. No-take MPAs had a greater number of positive ecological impacts than periodic closures and there was little evidence of any long-term ecological recovery within periodic closures following harvesting. Importantly, more than half of the studies examined (59%) did not provide any clear consideration of factors beyond the presence of the MPA that might have confounded their results. We conclude that counterfactual thinking has yet to be fully embraced in impact evaluation studies in the region and recommend pathways by which progress can be made.

2.2 Introduction

Natural ecosystems are under increasing anthropogenic pressures, some of which can be mitigated by protected areas (Chape et al. 2005; Mora et al. 2006). Marine protected areas (MPAs) have a diversity of objectives that can include enhancing ecosystem resilience, protecting biodiversity, and benefiting fisheries livelihoods by fostering sustainable harvesting (Halpern and Warner 2002; Gaines et al. 2010; Mellin et al. 2016). As a result of international targets (i.e. Convention on Biological Diversity Aichi 11) calling for nations to protect 10% of coastal and marine areas by 2020 (Toonen et al. 2013; Thomas et al. 2014), MPAs are expanding globally. Because MPAs alter human behaviour in ecosystems, their impacts have both ecological and socioeconomic dimensions. The impacts of MPAs can therefore be broad and multifaceted, with perceived success relating to the specific objectives for which MPAs are established (Jupiter et al. 2014). These objectives can be large-scale, such as national commitments to protect biodiversity, or local, such as enhancing the food security of communities and ecosystem resilience, and a single MPA can be established to achieve multiple objectives.

At the outset, the term "impacts" of MPAs is problematic. Studies vary in the rigour with which they determine impact, and therefore in their reliability. The most rigorous technique involves formal impact evaluation (Ferraro and Hanauer 2014, Ferraro and Pressey 2015), with impact defined as the intended or unintended consequences that are directly or indirectly caused by an intervention (e.g. MPA implementation) (Table 2.1) (Mascia et al. 2014). Importantly, this definition of impact involves counterfactual analysis (Table 2.1), which supports causal inference by asking: what would have happened in the absence of the intervention (Pressey et al. 2015, 2017; Adams et al. 2019)? Determining impact in this rigorous way can be challenging because it involves identifying how much observed conditions are due to the intervention, and how much to confounding factors (Table 2.1) that can mask intervention failure or exaggerate success (Adams et al. 2019). For example, Andam et al (2008) demonstrated that the actual impact of protected areas on deforestation in Costa Rica was confounded by most protected areas being located far from roads and in places that were unlikely to be deforested regardless of their status. Impact evaluation techniques employing counterfactual thinking have been well developed in fields other than conservation science, and although several studies have outlined quasi-experimental approaches for impact evaluation of protected areas (Ferraro and Hanauer 2014; Ahmadia et al. 2015), they have been embraced only slowly in the MPA evaluation literature (Pressey et al. 2017). Consequently, many studies aiming to estimate the impacts of MPAs have been limited by choice of counterfactual sites, often associated with little consideration of confounding factors. Despite this caveat, we refer throughout this paper to 'impacts' as estimates of MPA performance, acknowledging that these estimates vary in rigour and do not all constitute actual impact evaluationss. Part of our review assesses this variation in rigour.

Table 2.1. Key terms and definitions

Term	Definition
Before-after (BA)[a]	An evaluation technique that measures outcome variable(s) prior to and following MPA(s) implementation. It assumes that there are no concurrent factors that may influence outcome variable(s) and therefore any changes are attributable to the MPA.
Control-intervention (CI)[a]	An evaluation technique that measures outcome variable(s) at a single point in time at sites inside and outside MPA(s). It assumes that the outside site (control) accurately reflects the counterfactual condition of the MPA site. Specifically, it assumes that there are no differences between the control and MPA sites with respect to the outcome variables prior to the MPA being implemented and that the only factor that may influence outcome variables is the MPA.
Before-after-control-intervention (BACI)[b]	An evaluation technique that measures outcome variable(s) at sites inside and outside of MPA(s) prior to and following MPA implementation. This technique relies on the parallel trends assumption, that, in the absence of management, changes in outcome variables in MPA sites would be the same as those in control sites.
Before-after-control-intervention-paired-series (BACIPS) [c]	An evaluation technique that measures outcome variable(s) at paired sites inside and outside of MPA(s) prior to and following MPA implementation. This technique uses the average difference in the before period as a null hypothesis for the difference that would exist in the after period in the absence of an intervention. An addition to this approach is the progressive change BACIPS, which incorporates recovery rates into measurements of difference instead of assuming step-wise change following management.
Matching [d]	Grouping MPA sites with one or more control sites based on statistical measurements of similarity across multiple ecological or socioeconomic factors. Matching can be incorporated into CI, BACI, and BACIPS experimental designs.
Confounding factor	A known or unknown factor that can mask the true impact of an intervention, resulting in over or under-estimations of impact (see table 3).
Counterfactual [e]	The outcome that would have occurred in the absence of the intervention considered.
Factor	An element predicted to influence one or more reported variables. Factors can be those predicted to drive differences in MPA impacts (e.g. governance and management strategies), or those controlled for when selecting MPA and control sites (e.g. habitat and education level).
Impact [e]	The intended and unintended consequences (e.g. changes in knowledge and attitudes, behaviours, and/or social and environmental conditions) that are directly or indirectly caused by an intervention.
Outcome [e]	The desired ends that interventions are intended to induce (e.g. changes in knowledge and attitudes, behaviours, achieved targets of fish abundance or coral cover).
Reflexive counterfactual (RC) [f]	Framing social perception questions in a way that attributes causality to the protected area (e.g. *Are there more fish because of the MPA?*) and uses the surveyed individuals' perceptions of pre-existing conditions as the comparator.
Variable	An indicator for which change is measured within a study (e.g. target species biomass, income, catch).

[a] Adams et al. 2019

[b] Gertler et al. 2011

[c] Thiault et al. 2017b

[d] Ahmadia et al. 2015

[e] Mascia et al. 2016

[f] Franks et al. 2014

The direct ecological impacts of MPAs are generally changes in biomass, abundance, and diversity of target species (Alcala 1988; Russ and Alcala 1996; 2004; Halpern and Warner 2002), which are associated with limiting acute disturbances such as fishing, destructive anchoring, or development. Indirect flow-on effects, such as changes in total biodiversity, coral cover, or rates of herbivory, depend largely on changes in ecosystem dynamics based on the responses of target species (Mellin et al. 2016). Changes in ecological parameters can, in turn, influence socioeconomic impacts, such as fish catch and related income (Bartlett et al. 2009a). While some of the socioeconomic impacts of MPA implementation derive from ecological impacts, others do not depend on changes in marine ecosystems. For example, direct socioeconomic impacts can include community empowerment (Egli et al. 2010) or conflict over unfairness in regard to management-related decision-making (Gurney et al. 2014).

An extensive body of literature has sought to understand factors (Table 2.1) related to MPA impact (e.g. Halpern 2003; Claudet et al. 2008; Lester et al. 2009; Vandeperre et al. 2011). Larger and older MPAs generally have more impact (Edgar et al. 2014), as well as those with adequate staff and budget capacity (Gill et al. 2016). Management practices, here defined as the rules by which access to a reserve is administered, can also differ, with potential benefits and limitations of different practices. For example, whilst is has been established that permanent no-take MPAs often have greater ecological impact than periodic closures (Edgar et al. 2014), in some instances conflicting interests between users have resulted in periodic closures being more effective at achieving direct ecological impact (Giakoumi et al. 2017; Goetze et al. 2017a). Likewise, different governance strategies, defined here as how authority for administration is allocated, also have their own strengths and weaknesses. Centralized governance of MPAs is common in high-income countries and typically focused on biodiversity conservation objectives, but might not incorporate local stakeholders' objectives, resulting in low support and compliance and, therefore, a reliance on enforcement (Gaymer et al. 2014). In contrast, community-based governance, which is more prevalent in countries with strong local tenure rights or where government resources are limited (Govan 2009b), often focuses on local objectives and can therefore have greater local support (Ostrom 1990; Cox 2010), although broader conservation priorities might be achieved only incidentally (Ban et al. 2011). Importantly these strategies are not mutually exclusive and some countries have 'scaled-up' locally managed MPAs into broader networks (e.g. Fiji FLMMA) (Ban et al. 2011).

The body of work on MPA impacts varies widely in geographic scope, with both narrow and very broad scopes having limitations in identifying factors associated with positive impacts. Many studies have identified the impacts of individual or small groups of MPAs, and while these studies can demonstrate isolated successes and failures, they are unable to draw conclusions about different strategies within the same socio-economic and political contexts. In contrast, global reviews of MPA impacts have often compared various approaches to MPA implementation (Lester et al. 2009; Selig and Bruno 2010; Edgar et al. 2014; Gill et al. 2016). However, the high inherent differences between MPAs in such a broad-scale approach (e.g. habitats, species, governance, funding, and enforcement) likely misses many regionally relevant factors associated with impact. Regional-scale analyses (e.g. Giakoumi et al. 2017; Kamil et al. 2017) are therefore useful because they are able to highlight factors that confer positive impacts from MPA implementation that have particular importance in those contexts and that might differ from global generalisations. In regional studies, perspectives on MPA impacts can, for example, be compared between countries with very different management strategies while controlling for similarities in habitat and governance.

Protected areas in the South Pacific have a unique history that calls for a region-specific and regional-scale synthesis of MPA impacts. The region has a long tradition of local marine management, arising from high population densities on small land areas, with a large dependence on marine resources (Johannes 1978; Govan 2009). Western colonialism undermined traditional management with the imposition of new, centrally based laws by colonial powers and a breakdown of traditional authority (Johannes 1978). However, in the following 25 years, this process was sufficiently reversed for Johannes (2002) to retract his earlier appraisal, describing a renaissance of traditional marine management in Oceania. Even more recently, the commitment of Pacific Island nations to the CBD Aichi target 11 of 10% protected-area coverage for their marine and coastal waters has resulted in some countries creating additional large, centrally governed MPAs (e.g. Marae Moana – Cook Islands; Le Parc Naturel de la Mer de Corail – New Caledonia). This long and variable history, combined with strong local support and a rapid expansion of MPAs, has resulted in multiple management and governance strategies across a large area. The South Pacific is therefore ideally suited to examine different factors associated with MPA impacts. However, despite the extensive MPA literature in the region, the extent to which MPA evaluators have embraced counterfactual thinking and robust impact evaluation techniques, including consideration of confounders, is still unclear.

In this paper, we conducted a structured literature search and synthesis of studies that have set out to estimate MPA impact from the South Pacific. Studies not able to demonstrate impact in the counterfactual sense often instead measure outcomes, defined as the desired ends that interventions are intended to induce (Mascia et al. 2014) (Table 2.1), and these studies are also included in this review. Our questions are divided into two sections. Part 1 asks: i) what have been the overall ecological and socioeconomic impacts of MPAs in the South Pacific?, and ii) what are the factors that have been associated with positive, neutral, or negative ecological or socioeconomic impacts? Part 2 questions to what extent the MPA impact literature from the South Pacific has embraced counterfactual thinking and robust impact evaluation techniques, including consideration of confounding factors. We conclude that there is room for improvement in how MPA evaluation studies are being conducted in the South Pacific.

2.3 Methods

Study region

This study chose a predefined search area based on what has been traditionally termed the South Pacific (Figure 2.1). This is the region which most strongly identifies with the shifts from traditional marine tenure to central colonialist management, back to the renaissance of community-based management, and to the current paradigm of large-scale MPA implementation aimed at achieving international CBD targets.

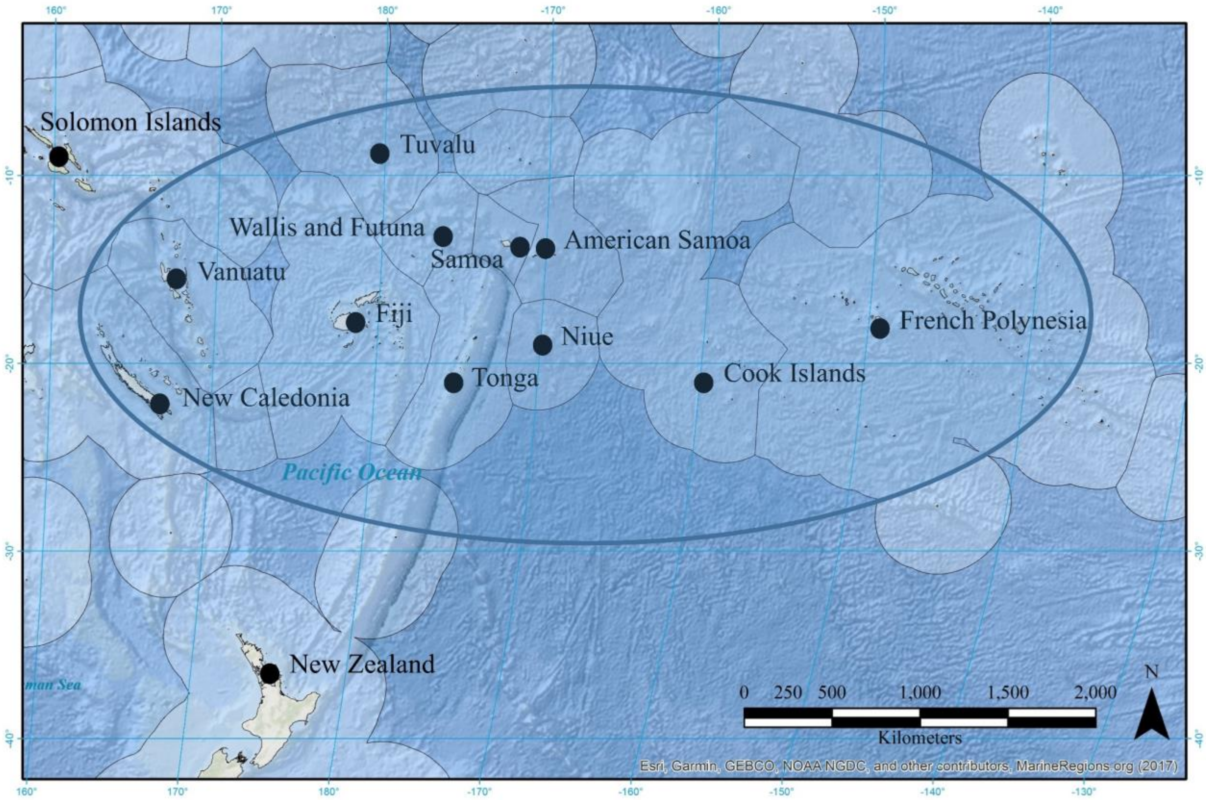

Figure 2.1. The South Pacific, as defined as the study region for this review.

Literature search

A structured search and review of the MPA literature from the South Pacific was conducted in Google Scholar and Web of Science during January and February 2018, and in February 2019. The identification of studies, inclusion criteria for the review, and data extracted for analysis are summarized in Figure 2.2. The search string was developed to include both locations and management terms specific to the region. Articles were screened by their title and abstract prior to full-text viewing based on pre-determined criteria. Articles were included for full-text viewing if they contained some sort of measurement of ecological or socioeconomic variables associated with the implementation, existence, or removal of an MPA in the South Pacific.

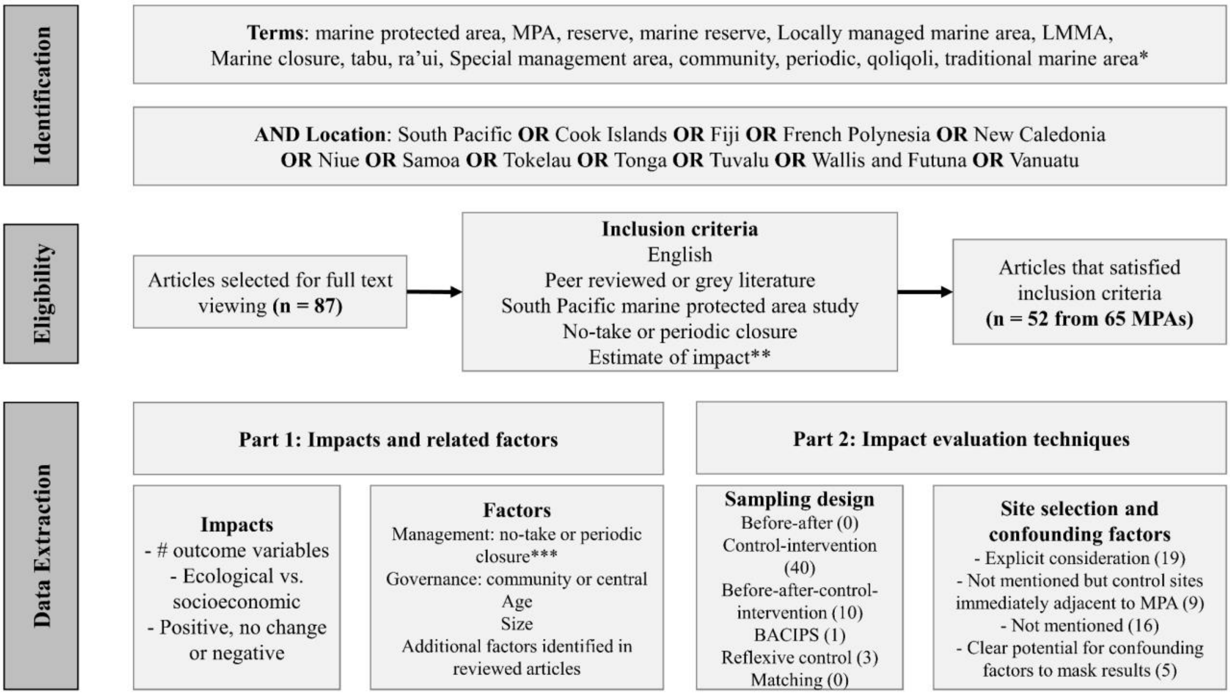

Figure 2.2. Flow diagram for article screening and inclusion in the review. For part 2 of data extraction (Impact evaluation techniques), the number of studies utilizing different sampling methods and site selection criteria are included in brackets. *While not all the marine conservation interventions listed in the terms box are necessarily MPAs, these terms were nonetheless used for initial screening purposes. Once articles were selected for full text viewing, only interventions that incorporated fully closed or periodically harvested closures were included in the analysis. ** Estimates of impact included any study using BA, CI, BACI, BACIPS, reflexive counterfactual or matching techniques (see Table 1 for definitions). While not all of these techniques necessarily quantify impact reliably, they were nonetheless included to assess how well studies in the region incorporated impact evaluation techniques. *** Impacts for periodic closures were measured at multiple time points: pre-harvest, immediately post-harvest, and following a recovery period.

Impacts and related factors

For each study that satisfied the inclusion criteria (Figure 2.2), the number and type (ecological or socioeconomic) of measured variables (Table 2.1) were recorded. All variables that were measured against a temporal or spatial control were examined and their difference (i.e. negative, no change, positive) from the control was noted. Rather than a traditional meta-analysis, which considers the relative effectiveness of each study at achieving a specific objective, we examined all outcome variables reported in each study to provide a comprehensive assessment of the impacts of MPA implementation relative to various factors. MPAs then were categorised according to the factors of governance approach (i.e. central or community governed) and management strategy (i.e. no-take or periodically harvested), and size and age if data were available. Not all MPAs fitted neatly within these categories. For example, management within an MPA might be to implement catch or gear restrictions, but we selected these categories because they were the most frequently cited in the literature. If multiple studies reported the same impact for a variable from the same MPA, only the most recent was used. Likewise, for no-take MPAs, if a study reported the same impact for the same variable at multiple points in time, only the most recent was included. However, if the impact of the variable differed between times for the same MPA, then both times were used. To assess the ecological recovery potential of periodic closures, pre-harvest, post-harvest, and recovery time points were all recorded separately. Finally, any additional factors suggested by authors in the studies as influencing MPA impacts were also recorded.

Impact evaluation techniques

To examine the extent to which the literature on impacts of South Pacific MPAs employs robust evaluation techniques, we examined each study's: i) sampling design protocols, and ii) justification of site selection and degree to which potential confounding factors were considered explicitly.

Most studies that intend to estimate MPA impact employ control-intervention (CI) or before-after-control-intervention (BACI) sampling protocols (Table 2.1), which might only quantify outcomes, and not impact, if the underlying assumptions are not verified (Adams et al. 2019). A control-intervention approach assumes that there were no differences in the outcomes of interest between the control and intervention sites prior to the implementation of the intervention. The need for this assumption is avoided by a before-after-control-intervention approach which assumes, however, parallel trends in variables of interest, that is, in the absence of the intervention the difference between the intervention and the control groups with respect to the outcome of interest is constant over time. An extension of the BACI approach is the paired series BACI (BACIPS), with which individual MPA sites are paired with control sites, which also assumes parallel trends in variables of interest. The degree to which these assumptions hold depends on how well potential confounders are accounted for. For example, matching methods (Table 2.1) provide the most rigorous approach to ensuring that confounders are accounted for, and can be applied to CI, BACI, and BACIPS sampling designs, although this approach has emerged only slowly in the MPA evaluation literature. Perception data from socioeconomic studies were also included in our review if the questions were framed so as to contain a reflexive counterfactual (Table 2.1), which involves framing survey questions in a way that attributes causality to an intervention. While reflexive counterfactuals can avoid the potential pitfalls of confounding factors, they also assume that each individual questioned has an accurate knowledge of the system both before and after intervention, as well as a strong understanding of attributing causality.

How well studies considered potential confounding factors was then assessed by searching publications for justification of selecting both MPA and control sites, as well as explicit recognition that additional factors could be masking actual impact. Many MPA evaluation studies often situate control sites immediately adjacent to MPAs without justification, assuming that this accounts for most potential confounders. We therefore included this approach as an additional category within our analysis. Studies were therefore categorized based on whether there was: i) explicit discussion of site selection criteria and potential confounding factors; ii) selection of spatial control sites immediately adjacent to MPA sites but with no mention of reasons; iii) no discussion of site selection criteria or the potential for confounding factors to affect results; or iv) clear evidence that the authors had selected biased control sites or the presence of additional confounding factors likely masking the true impact of MPAs (with or without discussion of site selection criteria).

2.4 Results

Of the 87 articles that were selected for full text viewing, 52 studies examining the impacts of 65 MPAs satisfied the selection criteria and were analysed further (Figure 2.2, Table 2.2). There was a large disparity in the number of cases assessing MPA impact between countries, with two countries - Fiji and New Caledonia - accounting for 75% of all studies. There was also a large disparity between the number of studies assessing ecological and socioeconomic variables. Of the 52 studies examined, only eight assessed socioeconomic data with methods that aimed to quantify impact.

Impacts and related factors

Impacts

Six hundred and sixty-two instances of 151 ecological impact variables were recorded. Overall, 42% of instances reported positive ecological impacts. The most frequently measured variables were total fish diversity and target fish biomass, and the nine most frequently measured variables accounted for 41% of the measured impacts (Figure 2.3). While 50% of studies that reported total fish biomass indicated positive impacts, only 38% were positive in the case of target fish biomass. The inverse was true for fish density (total 41%; target 48%). Positive impacts for invertebrates (67%) were almost twice as numerous as those measured for fish (38%). Positive impacts on coral cover were recorded in only 13% of cases.

Table 2. Key characteristics of the 52 studies evaluating MPA impact in the South Pacific. Governance type CB refers to community-based and C to central; management strategy NT refers to no-take and PC to periodic closures. See Table 1 for definitions.

Citation	*Country*	*Protected area name(s)*	*Data Type*	*Governance type*	*Management type*	*MPA age (years)*	*MPA size (km^2)*	*Experimental design*	*Site selection and confounding factors*
Albert et al. 2016	Fiji	Dravuni, Muaivuso and Namada	Ecological	CB	NT	2, 2, 3	Not listed, 3, 0.5	CI	Not discussed but adjacent control sites
Bartlett et al. 2009a	Vanuatu	Nguna, Pele and Emao	Socioeconomic	CB	NT & PC	4 to 6	Not listed	CI	Explicit discussion of site selection
Bartlett et al. 2009b	Vanuatu	Not Listed	Ecological	CB	NT & PC	4 to 6	0.07 to 0.21	CI	Explicit discussion of site selection
Berdach 2003	Tuvalu	Funafuti protected area	Ecological and Socioeconomic	C	NT	6	33	CI & RC	Not discussed
Beukering et al. 2014	Fiji	Navakavu	Socioeconomic	CB	NT	5	37.1	CI	Explicit discussion of site selection
Bonaldo and Hay. 2014	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	10, 10, 10	Not listed	CI	Explicit discussion of site selection
Bonaldo et al. 2017	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	10, 10, 10	Not listed	CI	Explicit discussion of site selection
Carassou et al. 2013	New Caledonia	Abore, Merlet and Bourail	Ecological	C	NT	8, 16 and 34	Not listed	CI with different fishing pressures	Not discussed
Clements and Hay 2017	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	Not listed	0.45 to 0.78	CI	Not discussed

Citation	*Country*	*Protected area name(s)*	*Data Type*	*Governance type*	*Management type*	*MPA age (years)*	*MPA size (km^2)*	*Experimental design*	*Site selection and confounding factors*
Clements and Hay 2018	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	12 to 13	0.45, 0.48, 0.78	CI	Explicit discussion of site selection
Clements et al. 2012	Fiji	Komave, Namada,Namatakula, Votua	Ecological	CB	NT	4 to 6	0.5 to 0.8	CI	Not discussed but adjacent control sites
D'agata et al. 2016	New Caledonia	Beautemps-Beaupre, Ouvea, Borendy, Pouebo, Hienghene, Southern Lagoons	Ecological	C & CB	NT & PC	Not listed, 21, 38	Not listed, 175	CI with gradient of controls	Explicit discussion of site selection
Dell et al. 2015	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	Not listed	0.5 to 0.8	CI	Explicit discussion of site selection
Dell et al. 2016	Fiji	Vatu-o-lalai and Votua	Ecological	CB	NT	Not listed	0.5 to 0.8	CI	Explicit discussion of site selection
Dixson et al. 2014	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	Not listed	Not listed	CI	Explicit discussion of site selection
Dumas et al. 2010	Vanuatu	Mangaronga and Marow	Ecological	CB	NT	3, 4	0,006 and 0.023	CI	Not discussed
Dumas et al. 2013	New Caledonia	Southern Lagoon MPAs	Ecological	C	NT	19, 24, 32	Not listed	CI	Explicit discussion of site selection
Dumas et al. 2012	Vanuatu	Anelcowat, Mangaliliu, Marou and Takara	Ecological	CB	PC	3, 7, 11, 12	0.033, 0.137, 0.15, 0.255	CI	Not discussed
Egli et al. 2010	Fiji	Kiobo, Nakorovou and Navatu	Socioeconomic	CB	NT	4	Not listed	BACI & CI	Not discussed

Citation	*Country*	*Protected area name(s)*	*Data Type*	*Governance type*	*Management type*	*MPA age (years)*	*MPA size (km²)*	*Experimental design*	*Site selection and confounding factors*
Ferraris et al. 2005	New Caledonia	Abore	Ecological	C	NT	5	148	BACI for MPA removal	Explicit discussion of site selection
Goetze et al. 2011	Fiji	Namena and Namuri	Ecological	CB	NT	4, 12	4.25, 60.6	CI	Evidence for biased control sites
Goetze et al. 2015	Fiji	Kiobo and Natokalau	Ecological	CB	PC	5	2.07	BACI	Evidence for biased control sites
Goetze et al. 2017	Fiji	Nakodu, Natokalau, Nauouo, Tuatua	Ecological	CB	PC	3 and 7	0.73, 1.34, 2.17, 3.69	BACI	Not discussed but adjacent control sites
Goetze and Fullwood 2013	Fiji	Namena	Ecological	CB	NT	12	60	CI	Evidence for biased control sites
Goetze et al. 2016	Fiji	Kiobo, Nakodu, Natokalau, Nauouo, Tuatua	Ecological	CB	PC	3, 4, 7, 8	.73, 1.34, 2.07, 2.17, 3.69	BACI	Not discussed but adjacent control sites
Januchowski-Hartley et al. 2014	Vanuatu	Not listed	Ecological	CB	NT & PC	1.5, 6	0.08-0.10	BACI	Not discussed but adjacent control sites
Jimenez et al. 2015	New Caledonia	Not Listed	Ecological	C	NT	10, 14, 19	Not listed	CI	Explicit discussion of site selection
Jimenez et al. 2016	New Caledonia	Not Listed	Ecological	C	NT	10, 14, 19	Not listed	CI	Explicit discussion of site selection
Jupiter and Egli 2011	Fiji	Namena, Namuri, Nasue, Nakali, Yamotu Lase	Ecological	CB	NT & PC	2, 10	.13, .77, 4.24, 8.14, 60.6	CI	Not discussed but adjacent control sites

Citation	*Country*	*Protected area name(s)*	*Data Type*	*Governance type*	*Management type*	*MPA age (years)*	*MPA size (km^2)*	*Experimental design*	*Site selection and confounding factors*
Jupiter et al. 2010	Fiji	Namena, Namuri, Nasue, Cakaulevu, Talai-i-lau, Vatuka, Nakali, Yamotu Lase	Ecological	CB	NT & PC	2, 3, 10	.13, .77, 4.24, 8.14, 14.05, 15.52, 18.85, 60.6	CI	Not discussed
Jupiter et al. 2012	Fiji	Cakaulevu	Ecological	CB	PC	3	15.5	BACI	Evidence for biased control sites
Jupiter et al. 2013	Fiji	Totoya, Moala, Tuvuca, Cicia, Vanuabalavu	Ecological	CB	NT	Not listed	Not listed	CI	Not discussed
Jupiter et al. 2017	Fiji	Cakau Bavu, Cakau Naitaga, Cakaulevu, Nakali, Nakodu, Tuatua, Vatunibalagi	Ecological	CB	PC	3.5, 4, 5, 7, 8	0.2, 0.7, 1.3, 2.1, 3.7, 4.7, 15.5	CI	Not discussed but adjacent control sites
Kulbicki et al. 2007	New Caledonia	Abore	Ecological	C	NT	5	148	BACI for MPA removal	Explicit discussion of site selection
Lalavanua et al. 2014	Fiji	Batiki	Ecological	CB	NT	Not listed	0.02	CI	Not discussed
Langlois et al. 2006	New Caledonia	Southern Lagoon MPAs	Ecological	C	NT	Not listed	Not listed	CI	Not discussed but adjacent control sites
Leopold et al. 2009	Fiji	Navakavu	Ecological	CB	NT	3	Not listed	CI	Explicit discussion of site selection
Moore et al. 2013	Tuvalu	Funafuti protected area	Ecological	C	NT	17	33	CI	Not discussed
Pascal 2009	Vanuatu	Emura, Piliura, Unakap, Laonamoa and Worasifiu	Socioeconomic	CB	NT	4 to 6	0.12 to 0.24	CI	Explicit discussion of site selection

Citation	Country	Protected area name(s)	Data Type	Governance type	Management type	MPA age (years)	MPA size (km2)	Experimental design	Site selection and confounding factors
Pascal and Seidl 2013	Fiji and Vanuatu	Emua, Siviri, Tanoliu, Mangaliliu, Tassiriki, Namada, Tagage, Vatu-o-	Socioecono mic	CB	PC	Not listed	Not listed	CI	Unclear
Peters 2017	Fiji	Bikini and Rabbit	Ecological	CB	NT	Not listed	Not listed	CI	Not discussed
Powel et al. 2016	New Caledonia	Southern Lagoon MPAs	Ecological	C	NT	18	Not listed	CI	Not discussed
Preuss et al. 2009	New Caledonia	Abore	Ecological	C	NT	3 to 11	50	BACI for MPA removal	Explicit discussion of site selection
Rasher et al. 2010	Fiji	Votua	Ecological	CB	NT	4	Not listed	CI	Not discussed
Rasher et al. 2013	Fiji	Namada, Vatu-o-lalai and Votua	Ecological	CB	NT	11	Not listed	CI	Not discussed
Siaosi et al. 2011	Tuvalu	Funafuti protected area	Ecological	C	NT	15	33	CI	Not discussed
Thaman et al. 2017	Fiji	Navukavu	Socioecono mic	CB	NT	Not listed	Not listed	RC	Unclear
Thiault et al. 2019	French Polynesia	Moorea MPA network	Ecological	C	NT	6	Not listed	Progressive change BACIPS	Explicit discussion of site selection
Tran et al. 2016	French Polynesia	Pihaena	Ecological	C	NT	10	0.578	CI	Not discussed but adjacent control sites

Citation	***Country***	***Protected area name(s)***	***Data Type***	***Governance type***	***Management type***	***MPA age (years)***	***MPA size (km^2)***	***Experimental design***	***Site selection and confounding factors***
Wantiez et al. 1997	New Caledonia	Southern Lagoon MPAs	Ecological	C	NT	5	Not listed	BACI	Evidence for biased control sites
Webster et al. 2017	Tonga	O'ua	Socioeconomic	CB	NT	Not listed	Not listed	RC	N/A
Zimmerman et al. 2018	Fiji	Rabbit	Ecological	CB	NT	Not listed	Not listed	CI	Not discussed

Seventy six instances of 49 socioeconomic variables were recorded. Overall, 72% of these reported positive socioeconomic impacts (Figure 2.3). Socioeconomic variables were grouped into five categories for summary analysis: i) catch (e.g. CPUE, maximum catch size); ii) economic impacts (e.g. income growth, revenue from tourism); iii) resource management decision-making (e.g. participation, inclusion of marginalised groups); iv) perceptions of ecological change (e.g. perception of coral cover, fish biomass); and v) perceptions of socioeconomic change (e.g. perceived change in remittance, change to income from fishing). All five categories had generally positive impacts, most frequently for catch, economic impacts, and perceived socioeconomic benefits. Neutral perceptions of ecological change were reported most frequently for changes in fish abundance, size and diversity, habitat health, and giant clam abundance. The most frequent negative impacts were recorded for participation which, compared to control villages, comprised four studies in which community members reported less ability to participate in meetings or have their interests represented.

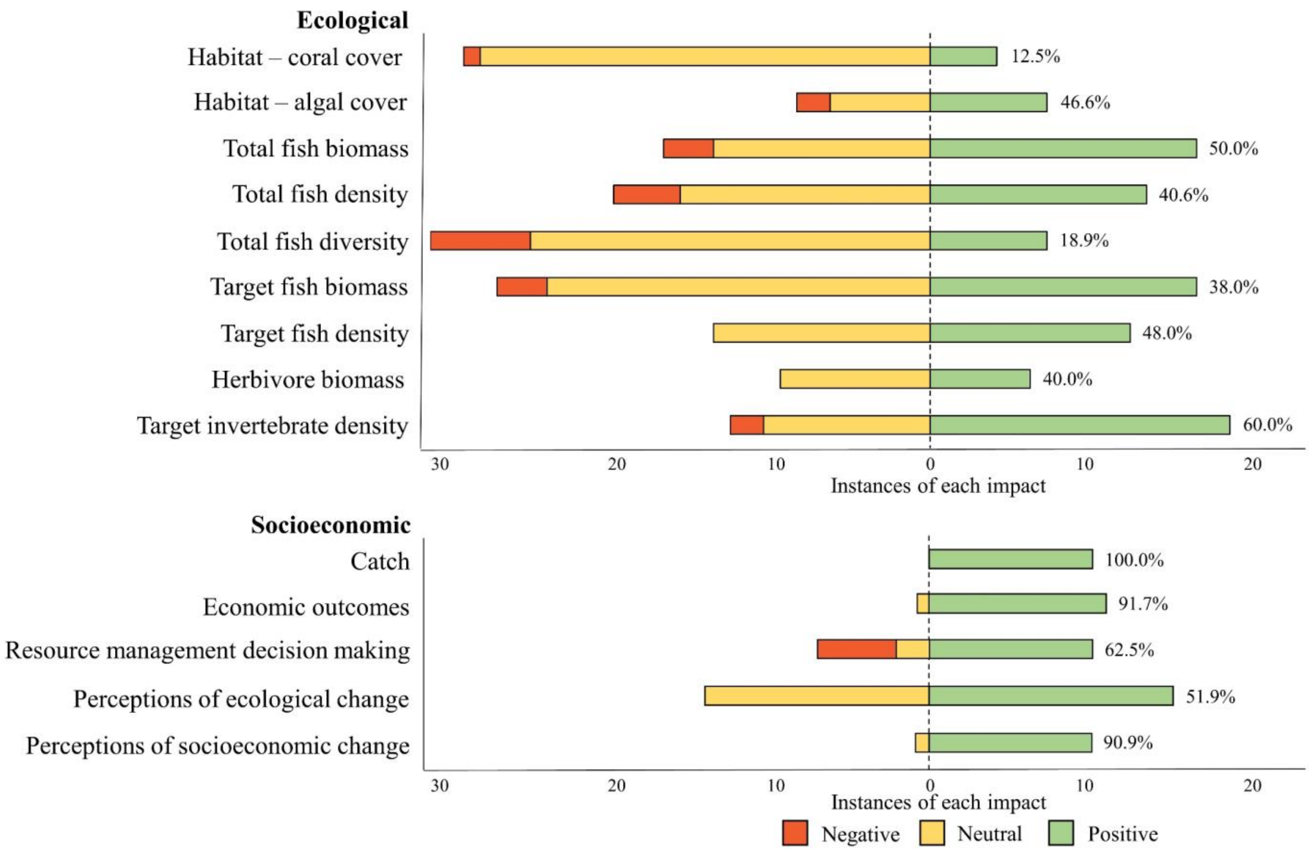

Figure 2.3. Positive, neutral, and negative impacts of MPAs. Ecological impacts shown include only a subset of the most frequently measured 151 ecological variables. Decreasing algal cover was considered a positive ecological impact. Socioeconomic impacts containing 76 study variables were divided into five groups. Numbers to the right of each bar indicate the percentages of measured impacts that were positive. Neutral impacts were included to the left of zero because MPAs aim to create positive impact.

Factors related to MPA success and failure

Both centrally governed and community-based MPAs had similar percentages of positive ecological impacts (48% and 43% respectively) (Figure 2.4), while socioeconomic impact variables were largely positive, regardless of the governance approach or management strategy (Figure 2.4). Community-based governance was the most commonly measured governance type. Thirty-six studies, examining 43 MPAs, measured the impact of community-based governance (both no-take and periodic closures), compared to 15 studies examining 14 MPAs that assessed impacts from central governance. These numbers were biased by country, with most centrally governed studies originating in New Caledonia and most community-based studies coming from Fiji and Vanuatu.

The greatest percentage of neutral and negative ecological impacts was for periodic closures (71%), which were implemented only under community-based governance approaches. Five studies examining 10 MPAs quantified the impacts of harvesting and recovery on periodic closures (Figure 2.5). Pre-harvest and post-harvest measurements were typically taken within one month each side of harvesting events, while recovery was measured one year later. There was a clear decline in the number of positive ecological impacts after harvest events and limited instances of recovery. In only two instances did a variable have a positive ecological impact following recovery and these were for the biomass of low and moderately vulnerable fish species.

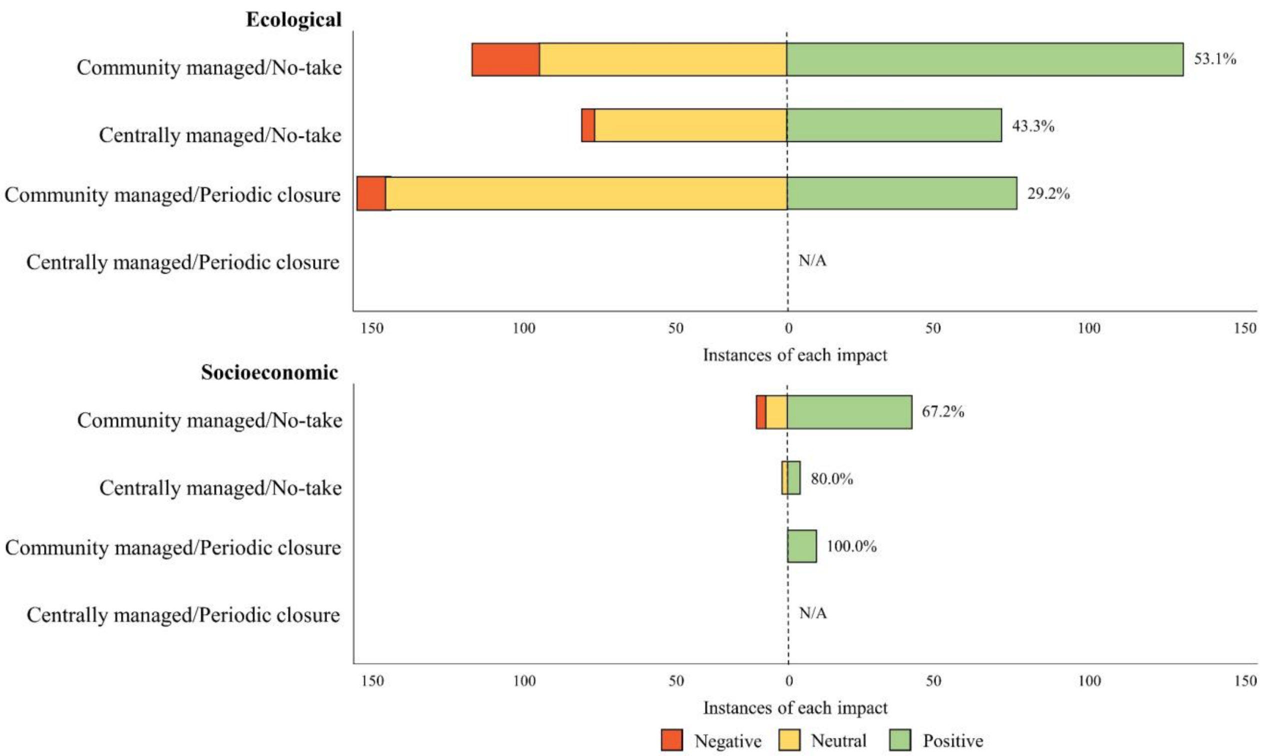

Figure 2.4. Positive, neutral, and negative ecological and socio-economic impacts in relation to governance and management strategies of MPAs. Numbers to the right of each bar indicate the percentages of measured impacts that were positive. Neutral impacts were included to the left of zero because MPAs aim to create positive impact.

Centrally governed MPAs were larger (mean 81 km^2 ± 29.3 SE) and older (mean 12 years ± 9.84 SE) than community governed MPAs (mean size 5 km^2 ± 1.9 SE; mean age 5 years ± 0.4 SE) (supplementary materials). There was no significant correlation between the proportion of positive impacts and either the age or size of MPAs (age: r=0.238, n= 47, p=0.107; size: r=-0.030, n=45, p=0.844). However, the mean percentage of positive impacts for MPAs less than ten years old was 36% (± 4.9 SE), while for MPAs greater than 10 years old it was 67% (± 6.0 SE).

Few studies discussed additional factors associated with positive MPA impacts beyond those listed above (governance, management, age, and size). However, 26 additional factors suggested to explain neutral or negative MPA impacts were identified in studies that did not observe expected positive impacts (Figure 2.6). These additional factors can be divided broadly into six categories, the relative frequency of which varied between governance approach and management strategy. When centrally governed MPAs failed to achieve positive impacts, it was generally suggested that the reasons were environmental (e.g. sediment discharge from a river mouth) or biological (e.g. changing predator dynamics). In contrast, when community-based MPAs failed to achieve positive impacts, factors most often suggested were related to reserve design (e.g. close to human populations), management (e.g. lack of compliance), or social constraints (e.g. poacher aggression). Compared with no-take reserves, failure of periodic closures to achieve positive impacts was suggested to be more likely associated with reserve design.

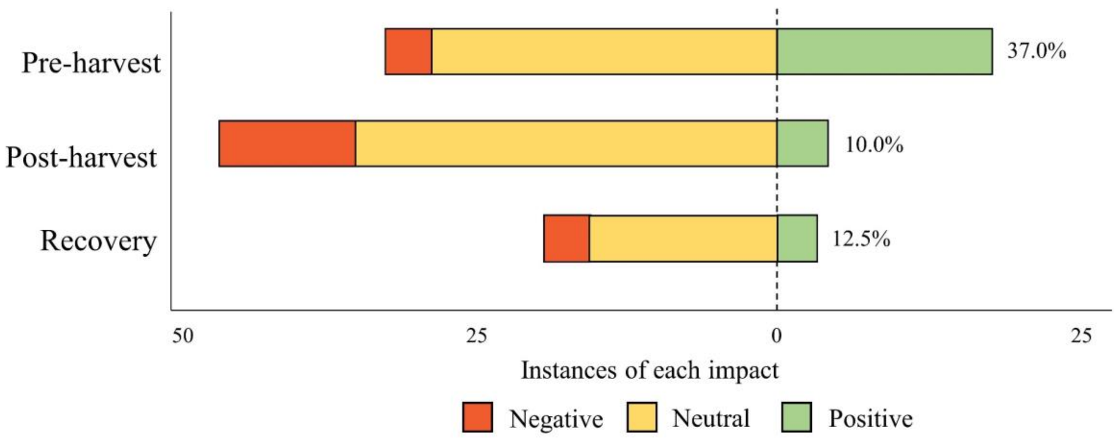

Figure 2.5. Numbers of ecological variables measured with positive, neutral, or negative impacts for periodic closures. Results are shown for pre-harvest (<1 month), post-harvest (<1 month), and following a recovery period (~1 year). Numbers to the right of each bar indicate the percentages of measured impacts that were positive. Neutral impacts were included to the left of zero because MPAs aim to create positive impact.

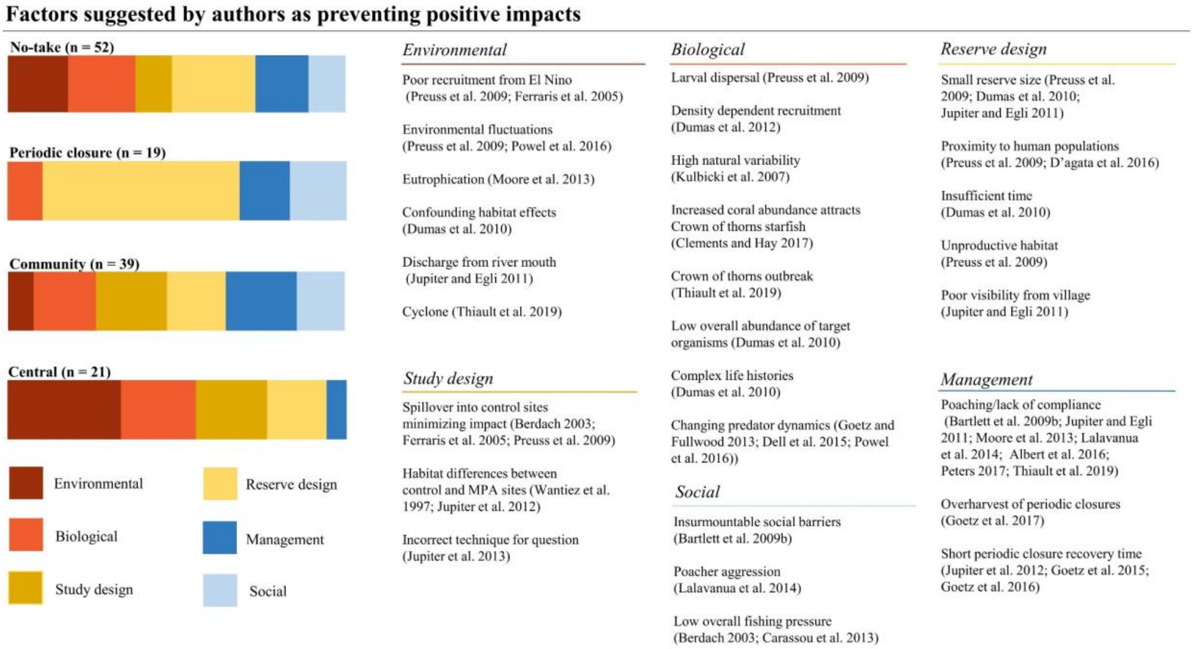

Figure 2.6. Additional factors suggested by authors of reviewed studies when MPAs failed to achieve positive impact. Factors, grouped into six categories, are allocated according to management (no-take MPAs or periodic closure) and governance (community-based or central). The sample size (n) indicates the number of studies included in each category.

Impact evaluation techniques

Most studies (73%) used control-intervention techniques (Table 2.2, Figure 2.7). No studies used only before-after data. Of the 52 studies, only 21 explicitly discussed any potential confounding factors in the selection of MPA and control sites. Within the studies that provided explanations for site selection, the reasoning was exclusively ecological; no studies considered any potential socioeconomic confounding variables in their sampling design. Of those that discussed ecological variables, the predominant consideration was habitat and, in a few cases, wave energy. While 20 studies selected control sites immediately adjacent to the MPAs, nine of these did so without explicit statements about what factors were being controlled for. Fourteen of the studies did not discuss the selection of control sites at all. Lastly, in five studies, it was clear that confounding factors were present that could influence outcome variables, potentially causing over- or under-estimations of true impact (Table 2.2, Figure 2.7).

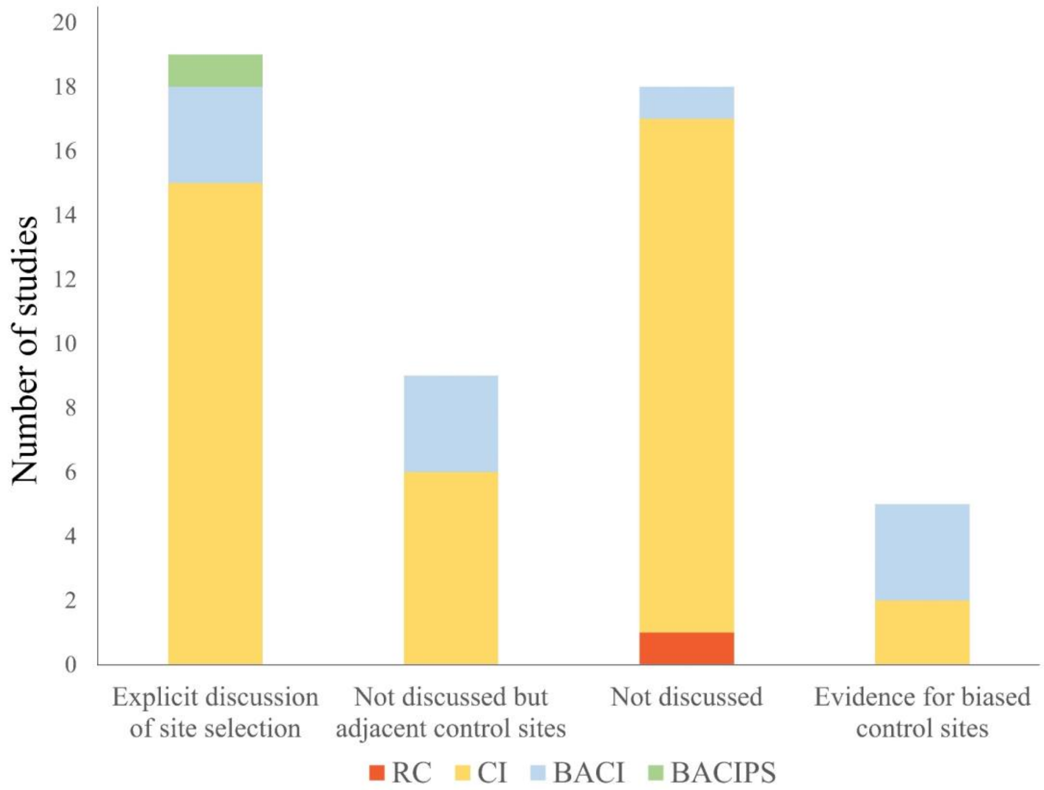

Figure 2.7. Evaluation of study quality, based on experimental design and criteria for site selection and consideration of confounding factors. See Table 1 for definitions.

2.5 Discussion

Study designs, methods for estimating impact of MPAs, and uncertainties around impact estimates can be understood with a theory of change that illustrates their relationships (Figure 2.8). In the sections that follow, we explore the implications of our findings and how they relate to different aspects of this theory of change. Specifically, we discuss: i) direct and indirect ecological and socioeconomic impacts; ii) factors related to MPA success and failure; and iii) the extent to which counterfactual thinking has been embraced by impact evaluation programs in the region.

Overall, we found that only half of all measured impacts of MPAs in the South Pacific were positive and, although from far fewer studies, the proportion of positive impacts was greater from socioeconomic studies. Community-based and centrally governed MPAs also had similar proportions of positive impacts, suggesting that both governance approaches are viable options in the region. Positive impacts were more common for no-take MPAs than periodic closures, and there was limited evidence of any ecological recovery potential in periodic closures following harvesting events. Although most of the reviewed MPAs had not been implemented for long, those that were older than 10 years had a higher proportion of positive impacts. A wide range of factors were reported by authors as being related to neutral or negative impacts, and these differed between management strategies and governance approaches. However, all the results of this study must be considered in the context of the MPA evaluation literature from the region rarely embracing explicit counterfactual thinking.

Impacts and related factors

Direct and indirect ecological impacts

The most commonly measured ecological impacts were fish biodiversity and target species biomass, likely reflecting broad conservation and local community objectives respectively (Jupiter et al. 2014). However, the neutral and negative results for these two variables in the majority of studies indicated that MPAs in the region failed to achieve these objectives more than 50% of the time. MPAs are a management strategy that directly affects only target species (Mosquera et al. 2000), so the impacts of MPA implementation should be most evident in these organisms. All other outcomes will depend largely on changes in ecosystem dynamics based on the response of target species (Allison et al. 1998). It is therefore important to understand why, in many instances, MPAs failed to increase target species populations. Of the 47 MPAs in our study with a known age, 76% were less than 10 years old, which, considering the long recovery times for many target species, could account for these poor results.

Benthic cover is rarely affected directly by MPA implementation, except where extensive damage occurs from anchoring, destructive fishing practices, collecting, or development (Milazzo et al. 2004). Rather, indirect mechanisms (Figure 2.8) by which MPAs can affect benthic cover are primarily through increases in herbivory, which reduces the competitive dominance of algal assemblages on corals (Lirman 2001; McCook et al. 2001; Hughes et al. 2007). However, while the relationship between coral-algal interactions and herbivory is well documented, few studies have demonstrated the ability of MPAs to change these relationships (but see Rasher and Hay 2010; Bonaldo and Hay 2014; Dell et al. 2016). This lack of evidence might be driven by discrepancies between short funding cycles for monitoring MPA impact and the time required for changes in coral cover to occur. Ultimately, as herbivores increase after protection, the balance between algal and coral dominance should shift. However, these results might take decades to manifest (Abesamis et al. 2014) and might also be masked by additional confounding factors that affect coral-algal interactions, such as wave energy (Adey 1998) or nutrient levels (McManus and Polsenberg 2004).

The greatest percentage of positive ecological impacts was found for the density of target invertebrates. In the South Pacific, invertebrates are often highly targeted and easily harvested, making them vulnerable to overexploitation (Uthicke and Conand 2005). However, given that many species reproduce and mature quickly (Battaglen 1999) and have small home ranges (Purcell and

Kirby 2006), they also have a high potential for rapid recovery following harvest reductions. Nonetheless, these life-history characteristics have also resulted in the massive overharvest and functional collapse of several target invertebrate populations in the South Pacific (Conand 2003). For MPAs to be effective at allowing stock recovery of target invertebrates, it is critical that meta-populations are sufficiently intact to allow recruitment into the protected areas after closure (Uthicke and Conand 2005).

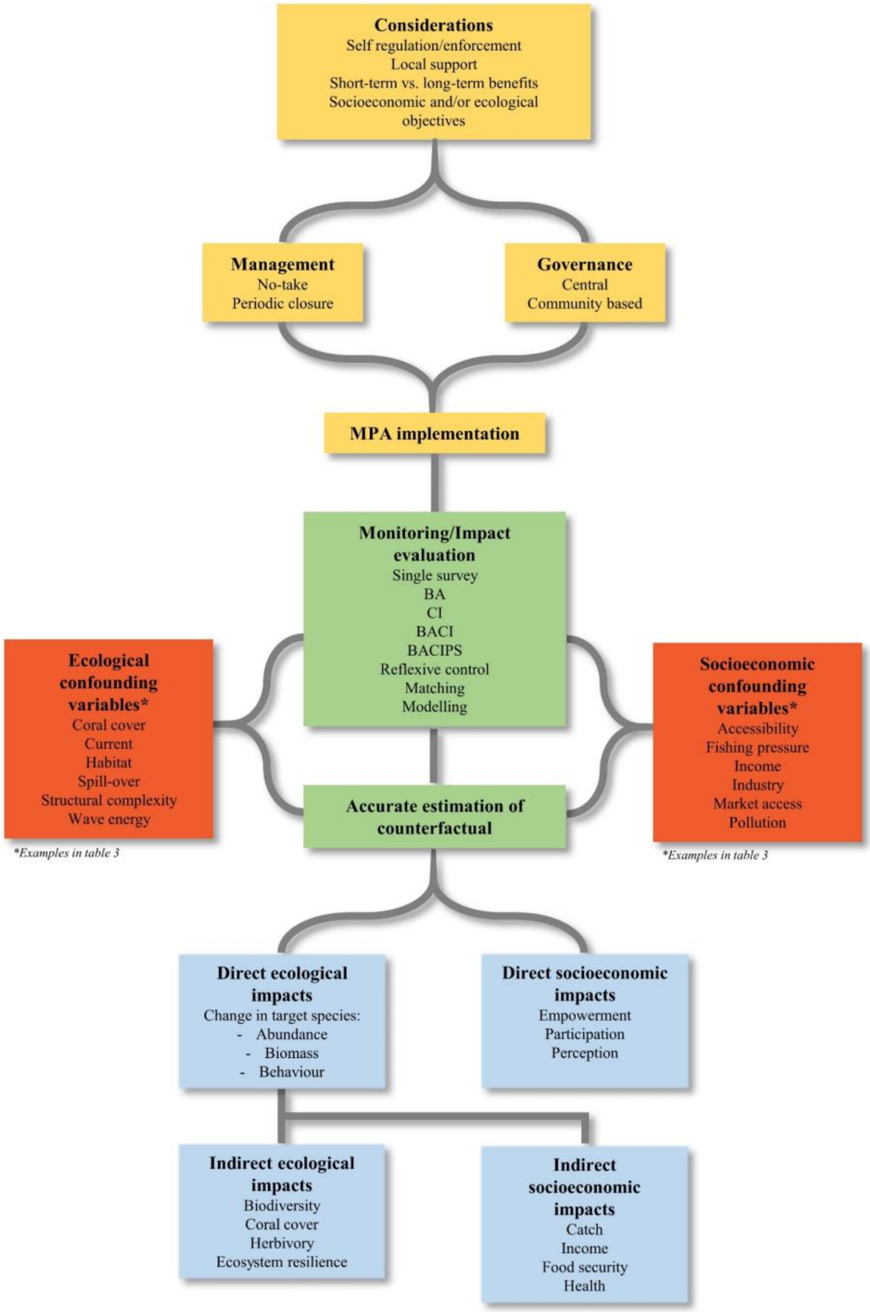

Figure 2.8. Theory of change depicting the pathway from MPA implementation to ecological and socioeconomic impacts. The yellow boxes indicate the considerations for implementing MPAs with different management strategies and governance approaches. The green boxes show methods by which impact was assessed. The red box lists examples of potential confounding factors that should be considered to accurately assess impact. The blue boxes provide examples of direct and indirect ecological and socioeconomic impacts of MPAs that can be determined through rigorous monitoring and evaluation.

Direct and indirect socioeconomic impacts

Most of the recorded socioeconomic impacts identified were positive, which suggests that, for the South Pacific, MPAs can be a viable strategy for both conservation and development. Given that some socioeconomic impacts are not mediated by ecosystem change (Figure 2.8; Gurney et al. 2014), they can likely manifest over much shorter periods of time, which, given the young age of most MPAs, could explain the higher percentage of positive socioeconomic than ecological impacts. In addition, perceptions of ecological change, a socioeconomic impact, might not always be aligned with actual changes in the environment. For example, Bartlett et al. (2009a) and Yasue et al. (2010) highlighted how perceptions of ecological variables are generally much greater than quantified ecological outcomes. Despite these considerations, the results suggest that the evidence for positive social impacts from MPAs in the South Pacific is strong.

Our review identified only eight studies that quantified socioeconomic impacts. Many socioeconomic studies in the South Pacific have focused on factors leading to successful MPA implementation (e.g. Govan 2006; Abernathy et al. 2014; Cohen et al. 2014) or discussed the importance and revitalization of traditional management (e.g. Johannes 1978; 2002; Govan 2009). The relatively small number of impact studies likely arises from key challenges that make quasi-experimental designs difficult to implement in social science research. These challenges include achieving a sufficient sample size, particularly at the level of villages, and finding appropriate control villages, which are both similar to MPA villages and have people willing to be used as controls. Outcomes can also vary between subgroups (Gurney et al. 2015), and inequality among social groups can lead to conflict (Fabinyi et al. 2013), jeopardizing achievement of goals for both social and ecological projects (Persha and Andersson 2014). Because of these problems, traditional control-intervention and BACI sampling designs are somewhat less feasible with social data (but see included studies and Gurney et al. 2014; 2015). An additional caveat on the results of this review is that favouring quasi-experimental designs and quantitative data might also risk ignoring many potential impacts of MPAs on people that are not easily quantified. Such impacts are likely to be related to non-material connections between humans and nature (such as cultural and relational values; e.g. Chan et al. 2016, Lau et al. 2019) that are increasingly emphasised in recent literature. An example of a conceptual framework that includes non-material values is that of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, emphasising nature's contribution to people (Diaz et al. 2018). Therefore, assessments of the socioeconomic impacts of MPAs should ideally take a mixed-methods approach (e.g. Sterling et al. 2017), drawing on both quantitative and qualitative methods, while recognising the inherent limitations of both.

Factors related to MPA success and failure

Of the two governance approaches examined (Figure 2.8), community-based governance, which can be established for a range of purposes (e.g. food security, maintaining traditional tenure), had similar proportions of positive impacts as centrally governed MPAs, which are ostensibly focused on nature conservation (Figure 2.4, Figure 2.6). While community-based MPAs are rarely systematically configured to maximize impact, their configurations can still be close to optimal (Smallhorn-West et al. 2018). This is because community-based MPAs are often situated close to villages for social reasons, such as ease of enforcement, which can result in higher impacts in otherwise heavily fished areas. In contrast, while centrally governed MPAs have the potential to be systematically configured to achieve the greatest impact, in practice they can often be situated residually (Devillers et al. 2015) where impacts can be limited.

Of the two management strategies examined (Figure 2.8), no-take MPAs, which were generally more effective than periodic closures (Figure 2.4), often have straightforward enforcement and simpler regulations, with clear benefits accruing both inside no-take reserves and from spillover to adjacent areas (Abesamis and Russ 2005). Given the direct objectives of MPAs are typically to increase target species biomass and density, it is also clear that strategies that minimize harvest effort should have the greatest conservation impacts. However, while there is a general consensus on the greater potential benefits of no-take MPAs, Giakoumi et al. (2017) suggested that periodic closures can still be useful for heavily human-dominated regions because multiple users have interests that are often in conflict, and no-take MPAs can be considered obstacles to some of their activities. This is further supported by the results of Bartlett et al. 2009, who found that periodic closures were more effective ecologically than no-take MPAs. These authors suggested that no-take MPAs in the Asia-Pacific region commonly fail to meet their objectives due to low compliance (McClanahan et al. 2006) and insurmountable social barriers (Cinner 2007). They concluded that, in the community context, periodic closures can provide an acceptable alternative to no-take reserves because they are both practical and locally appropriate.

Our review also suggests that the ecological benefits of periodic closures are limited to pre-harvest conditions, when they are effectively acting as recently implemented no-take MPAs, with little evidence of any post-harvest recovery. The recovery time following highly intensive harvesting events can be between 5 and 20 years (Abesamis et al. 2014), much greater than the one-year post-harvest timing used by most studies to measure recovery (Jupiter et al. 2012; Goetze et al. 2016). Periodic closures are therefore most likely to achieve short-term objectives such as increasing fisheries yields from single, repeated harvest events, and are unlikely to achieve longer-term conservation objectives (Goetze et al. 2017b).

The equal proportion of positive impacts for MPAs regardless of size indicates that small MPAs can be effective and that size should not be the sole consideration during the design phase. Residually situated reserves with low impact can be larger in size and more typical of centrally governed areas, because proclaiming MPAs in areas of little value to resource extraction industries is likely to face little opposition whilst providing a means for governments to apparently fulfil their conservation commitments. In contrast, smaller, community governed reserves can be configured in less residual areas where potential impact can be higher (Smallhorn-West et al. 2018). Further studies should clarify the trade-offs between reserve size and management strategies while accounting for differences in potential impact.

The factors suggested to account for MPAs failing to achieve positive impacts differed between management and governance strategies. While this disparity could result from differences in the characteristics of the reserves themselves, the articles reviewed did not suggest noticeable ecological or socioeconomic differences in MPAs between governance or management types. An alternative explanation is that, when MPAs fail to have a positive impact, researchers studying community governed MPAs might focus more on socioeconomic factors while those studying centrally governed MPAs might give more consideration to biological or environmental factors.

Impact evaluation techniques

While acknowledging that many MPA studies are opportunistic, it is clear that, in the South Pacific, counterfactual thinking has yet to be fully embraced and that more consideration is needed of the potential for confounding factors to obscure actual impacts (Figure 2.8). Therefore the results of part 1 of this review must be considered while acknowledging the limited efficacy of most studies to quantify actual impacts. The non-random placement of MPAs can result in biases towards specific locations (e.g. high-quality environments, residual areas), leading to over- or under-estimations of impact. While control sites selected by many of the included studies could, in reality, represent fairly accurate counterfactual conditions, unless these conditions are quantified explicitly, or at the very least clearly considered, it is difficult to attribute causality to MPA implementation. No studies in the South Pacific quantitatively accounted for confounding factors, and 60% of studies did not explicitly discuss any selection criteria for MPA or control sites. There was some evidence of a trend for studies to select control sites with a similar habitats as MPA sites, but few other ecological factors were expressly considered. Furthermore, there is no evidence that any potential socioeconomic confounders were considered during the selection of survey sites. This result is particularly relevant given the growing body of literature demonstrating the key role of social dynamics in MPA impacts (e.g. Pollnac et al. 2010). Lastly, there may also be more general biases towards better performing MPAs being published in the literature over those with neutral or negative impacts.

Thiault et al. (2019), utilizing the BACIPS approach, provided the most robust methods used to date in the South Pacific and did so while explicitly discussing estimates of counterfactual conditions. The conceptual framework for this study is therefore an ideal starting point for researchers aiming to develop sound impact evaluation programs in the South Pacific. However, even within this well planned study, the caveat remains that pairing appears to be based exclusively on geographic proximity and physical characteristics, with no mention of socioeconomic conditions, and it is unclear whether pairing was quantified or subjective.

Table 2.3 provides examples of how confounding factors might lead to over- or under-estimations of impact in this region. Five studies in this review had clear potential for confounding factors to mask the true impact of MPAs. Wantiez et al. (1997) compared five control sites situated adjacent to the capital of New Caledonia with five MPA sites located several to tens of kilometres away. Observed differences between sites could be due to population pressure or fishing pressure and not the MPAs per se, resulting an overestimation of impact. Jupiter et al. (2012) discussed the potential confounding factor of large habitat differences between two of their control sites and two of their four MPA sites, although they noted that the location and replication of survey sites was constrained by the opportunistic nature of the study. Goetze et al. (2011), and subsequently Goetze and Fullwood (2013) and Goetze et al. (2015), specifically selected control sites so that “fished [control] sites were placed in areas adjacent to the reserves where high levels of fishing are known to occur”. Control sites with high fishing pressure represent an accurate counterfactual only if the MPA sites also would have had equally high fishing pressure in the absence of management. This sampling design could over-estimate impact by failing to account for potential differences in fishing pressure in the absence of MPA implementation, such as low extractive potential of the MPAs. More generally, our review also found poor study design listed as a factor influencing observed neutral and negative MPA impacts, which indicates both that substantial improvements could be made to standard protocols and that the authors were aware of their studies’ limitations. These results highlight the lack of systematic process in the current MPA impact evaluation literature by which accurate estimates of counterfactuals are produced.

In addition to these five studies, nine studies from Fiji compared various impacts between three MPAs and adjacent control sites in Namada, Vatu-o-lailai, and Votua. These MPAs are exceptional in having some of the greatest percentages of positive impacts recorded, particularly on ecosystem processes (rates of herbivory, crown-of-thorns starfish abundance) and benthic cover (210-280% greater coral cover inside MPAs). However, given their small size (~0.5 km^2) and that were only recently implemented at the time of data collection (many <10 years), it would be necessary to demonstrate conclusively that the results were due to MPA implementation and not influenced by confounding factors such as a bias in reserve placement over high-quality habitats. While Bonaldo and Hay (2014) mentioned unpublished data reporting low coral cover in both MPA and control sites prior to implementation, the exceptional degree of impact reported underlines the need for all potential confounders to be considered.

Table 2.3. Examples of potential ecological and socioeconomic confounding factors that can influence estimates of the difference between MPA and counterfactual conditions.

Potential confounders	Examples of how poorly chosen control sites can lead to over- or under-estimation of impact
Coral cover and structural complexity	Greater coral cover and complexity increases the carrying capacity of an ecosystem. An MPA is configured to protect areas with exceptional coral cover. Subsequent control-intervention studies that fail to account for high coral cover will overestimate impact.
Displaced fishing effort	An MPA displaces current fishing activity to a nearby reef, which is subsequently used as a control site. Displaced fishing effort from the MPA will result in variables of interest declining in nearby areas, with overestimation of impact, even though the net stock remains the same.
Education	Education about ecological recovery is introduced by an NGO along with an MPA. Perceptions of ecosystem health in the MPA community therefore increase. At the same time they also conduct educational outreach in a nearby control village with no MPA, thereby increasing their understanding of the damage fishing is causing. Impact is overestimated because the difference in perceived change between MPA and control villages is the result of additional educational programs and not the implementation of the MPA.
Fishing pressure	Control sites are selected in areas with higher fishing pressure than would have occurred in MPAs, overestimating impact. Sites with high fishing pressure do not represent an accurate counterfactual unless the MPA sites would also have had equally high fishing pressure in the absence of management. (e.g. Wantiez et al. 1997; Goetze et al. 2011, 2015; Goetze and Fullwood 2013).
Habitat quality	High/Low-quality habitats are selected for protection by MPAs, which have a higher/lower carrying capacity of target species than control sites. Subsequent control-intervention studies over/underestimate impact. (e.g. Jupiter et al. 2012).
Income	A village with high average income is used as a control for an MPA village with low income. Fishing in the high-income village is conducted with new equipment and faster boats than the MPA village. Economic impact is underestimated because of failure to account for difference in fishing efficiency.

Industry	A tuna canning factory is introduced near a village heavily reliant on fishing. The factory employs people from a nearby village with an MPA but not from the village acting as the control. Dependence on fishing decreases in the MPA village but remains stable in the control village. Income rises in the MPA village. The biological impact of the MPA is overestimated because the number of people fishing in the MPA village has decreased. The economic impact of the MPA is overestimated because increased income stems from employment in the factory.
Market access	A non-MPA village has excellent access to a large market in the capital city. A nearby MPA village has greater catch rates, but economic impact is underestimated because they receive less income for their catch due to unequal market connection.
Politics	A recent election has empowered many community members in an MPA village to participate in village affairs. Social impact of the MPA is overestimated because empowerment was not the result of the MPA, but of the recent election.
Pollution	Sedimentation from a nearby agricultural enterprise has increased algal proliferation on an MPA reef. Impact is underestimated compared to a healthy control site.
Spillover from adjacent MPA	Control sites located too close to MPAs, within the radius of target species spillover, record a smaller difference between control and MPA sites and ultimately underestimate impact.
Wave energy and current	High-current environments (e.g. lagoon entrances) can have greater abundances of fish than surrounding areas. An MPA is in the middle of a reef but the lagoon entrance is used as a control site. Greater species abundance at the lagoon entrance results in an underestimation of impact.

Ways forward

With the current trend of rapid ecosystem degradation, researchers must often be both opportunistic when developing research methods and quick to draw robust conclusions from management interventions. Nonetheless, environmental policy must be evidence-based, and it is therefore imperative that either rigorous protocols are in place to demonstrate impact, or the implications of alternative evaluation methods are understood (Pressey et al. 2015; McIntosh et al. 2017; Pressey et al. 2017). Care must be taken to effectively manage two types of confounders, those that influence the observed variables themselves (e.g. effects on coral cover or fish biomass) and those that influence the placement of MPAs (e.g. residual locations with low inherent fishing pressure, or proximity to communities, where fishing pressure is high, for ease of enforcement). Ferraro (2009) and Ferraro and Hanauer (2014) provided the foundation for counterfactual thinking and impact evaluation in evaluating protected areas, and proposed both experimental and quasi-experimental designs to build the evidence base for the impact of environmental policy and conservation interventions. One such approach is the statistical matching of MPA sites to controls which, for ecological impacts of MPAs, is described in detail in Ahmadia et al. (2015). This approach can be used after MPA establishment to avoid observable selection bias and identify comparable control sites to accurately estimate counterfactual conditions (see R matching packages Matching and Matchit). Matching can therefore be used in both BACI (including BACIPS) and, when temporal data are not available, CI programs. A matched BACIPS approach (Thiault et al. 2017b & 2019) would represent the most robust non-experimental method to determine MPA impact.

However, we acknowledge that employing matching methods require specialist statistical and coding expertise and that training in such might be difficult to access by researchers and practitioners based in the South Pacific. We suggest that further research should focus on developing simpler techniques for the preliminary matching of MPA and control sites based on predefined variables, or ways to easily tabulate the most important ecological and socioeconomic factors that could influence the variables being measured. An important starting point, not requiring specialist expertise, is to carefully consider both the potential ecological and socioeconomic factors that influence the variables of interest and placement of MPAs during site selection (Figure 2.8), and to discuss these explicitly in subsequent publications. This approach would increase the robustness and clarity of conclusions regarding impact. To this end, we argue that MPA evaluation programs in the South Pacific should move towards fully embracing counterfactual thinking to allow researchers, managers, and stakeholders to draw robust conclusions regarding the difference made from both current and future marine protected areas.

Chapter 3: Tongan socio-environmental spatial layers for marine ecosystem management

P. Smallhorn-West, S. Gordon, A. Dempsey, S. Purkis, S. Malimali, T. Halafihi, P. Southgate, T. Bridge, R. Pressey, G. Jones

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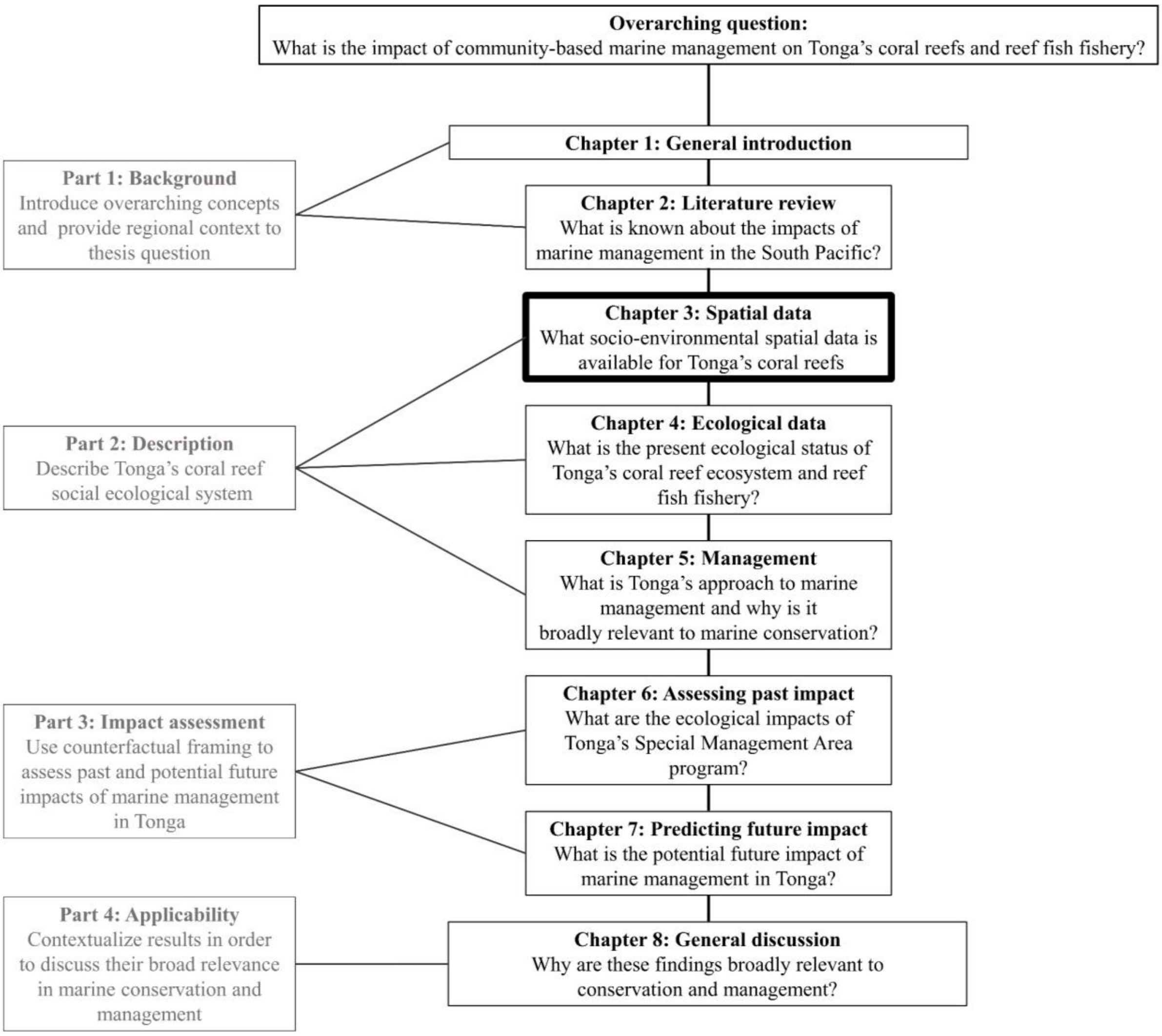

3.1 Abstract

Environmental conditions and anthropogenic impacts are key influences on ecological processes and associated ecosystem services. Effective management of Tonga's marine ecosystems therefore depends on accurate and up-to-date knowledge of environmental and anthropogenic variables. Although many types of environmental and anthropogenic data are now available in global layers, they are often inaccessible to end users, particularly in developing countries with limited accessibility and analytical training. Furthermore, the resolution of many global layers might not be sufficient to make informed local decisions. While the near-shore marine ecosystem of Tonga is extensive, the resources available for its management are limited and little is known about its current ecological state. Here we provide a marine socio-environmental dataset covering Tonga's near-shore marine ecosystem as compiled from various global layers, remote sensing projects, local ministries, and the 2016 national census. The dataset consists of eleven environmental and six anthropogenic variables summarized in ecologically relevant ways, spatially overlaid across the near-shore marine ecosystem of Tonga. The environmental variables selected include: bathymetry, coral reef density, distance from deep water, distance from land, distance from major terrestrial inputs, habitat, land area, net primary productivity, salinity, sea surface temperature, and wave energy. The anthropogenic variables selected include: fishing pressure, management status, distance to fish markets, distance from villages, population pressure, and a socioeconomic development index based on population density, growth, mean age, mean education level, and unemployment. This extensive and accessible dataset will be an essential tool for future assessment and management of marine ecosystems in Tonga.

3.2 Introduction

The effective management of Pacific marine ecosystems depends on an accurate understanding of ecological processes. Environmental conditions, such as temperature and wave energy, are key determinants of the structure of tropical marine ecosystems (Bradbury and Young 1981; Graus and Macintyre 1989). Likewise, anthropogenic impacts are increasingly overtaking environmental conditions as the main drivers of ecosystem structure (Hughes *et al.* 2017a; Cinner *et al.* 2012; 2018). For example, coral reef ecosystems are degrading globally due to the impacts of human induced climate change (Hughes et al., 2017b), while reef fish biomass is often most strongly associated locally with human population pressure (Cinner *et al.* 2012, Smallhorn-West *et al.* 2018). Scientists and managers in the South Pacific must therefore account for both environmental variation and human impacts in order to accurately model, manage and conserve tropical marine ecosystems (Stone *et al.* 2019).

The proliferation of satellite-based monitoring has stimulated the development of numerous global environmental and anthropogenic layers, many of which are publicly available (see Sbrocco and Barber 2013, Yeager *et al.* 2017). However, for end users, these can often require considerable computational power to process (Purkis 2018). In addition, the coarse resolution of many global layers (often 1 to 4 km) can also limit their applicability at the local level, where management relies on spatially-precise knowledge to inform decisions. Across smaller extents, although local socio-environmental data collected by government ministries and NGOs might be available, these data are often not publicly accessible and are rarely merged between parties. Both global and local data sets are critical to making informed management decisions, although both routinely suffer from limited accessibility. Although optimal for decision makers, it is also uncommon for global and local socio-environmental datasets to be consolidated into single, easily accessible outputs (but see Gassner *et al.* 2019). With increased acknowledgment of the importance of well-informed management, it is essential that existing data on environmental conditions and human impacts are freely available to end users.

The Kingdom of Tonga is a small island nation in the South Pacific with a substantial near-shore marine ecosystem. Tonga's population is distributed among its 659,558 km^2 of Exclusive Economic Zone (EEZ) and 169 islands and is strongly dependent on marine resources for food security and livelihoods (Stone *et al.* 2019). There have been substantial recent efforts by various parties to improve assessment and management of Tonga's marine resources through the establishment of community-based marine protected areas (Govan 2015; Gillet 2017) and international collaborative actions (e.g. Joint country strategy 2009-2013; Moore and Malimali 2016; Tonga Fisheries Sector Plan 2017). One available resource recently produced in this process is the Tonga marine atlas (Gassner *et al.* 2019). This document series contains information on many environmental and some anthropogenic conditions for several Pacific Island countries and is a valuable resource for marine management (http://macbio-pacific.info/). While the layers included in this report are available for users as links to global source layers, access to the layers often requires considerable analytical training and computational power to extract and process. In addition, the available global layers are coarse in resolution, and are directed at national management of Tonga's entire Exclusive Economic Zone (EEZ), limiting their suitability for fine resolution data extraction and planning. Despite high demand for accurate and accessible information on socio-environmental marine management factors, there is currently no resource available that compiles information from local governments, NGOs and international organizations on variables known to influence marine ecosystems, such as current management extent, census data and marine environmental conditions.

Here we present a marine socio-environmental dataset for Tonga's near-shore shallow marine ecosystem (Fig. 1) consisting of 17 environmental and socioeconomic variables compiled from global and local sources and made freely and easily available to end users. All layers are accessible as vector or raster files for download from the supplementary materials. This dataset is not intended to replace or discredit original source material or the exceptional efforts by the involved parties, but rather to consolidate all currently available and suitable resources in one location and produce additional novel data layers to complement existing information vital for managing and conserving Tonga's near-shore marine environment.

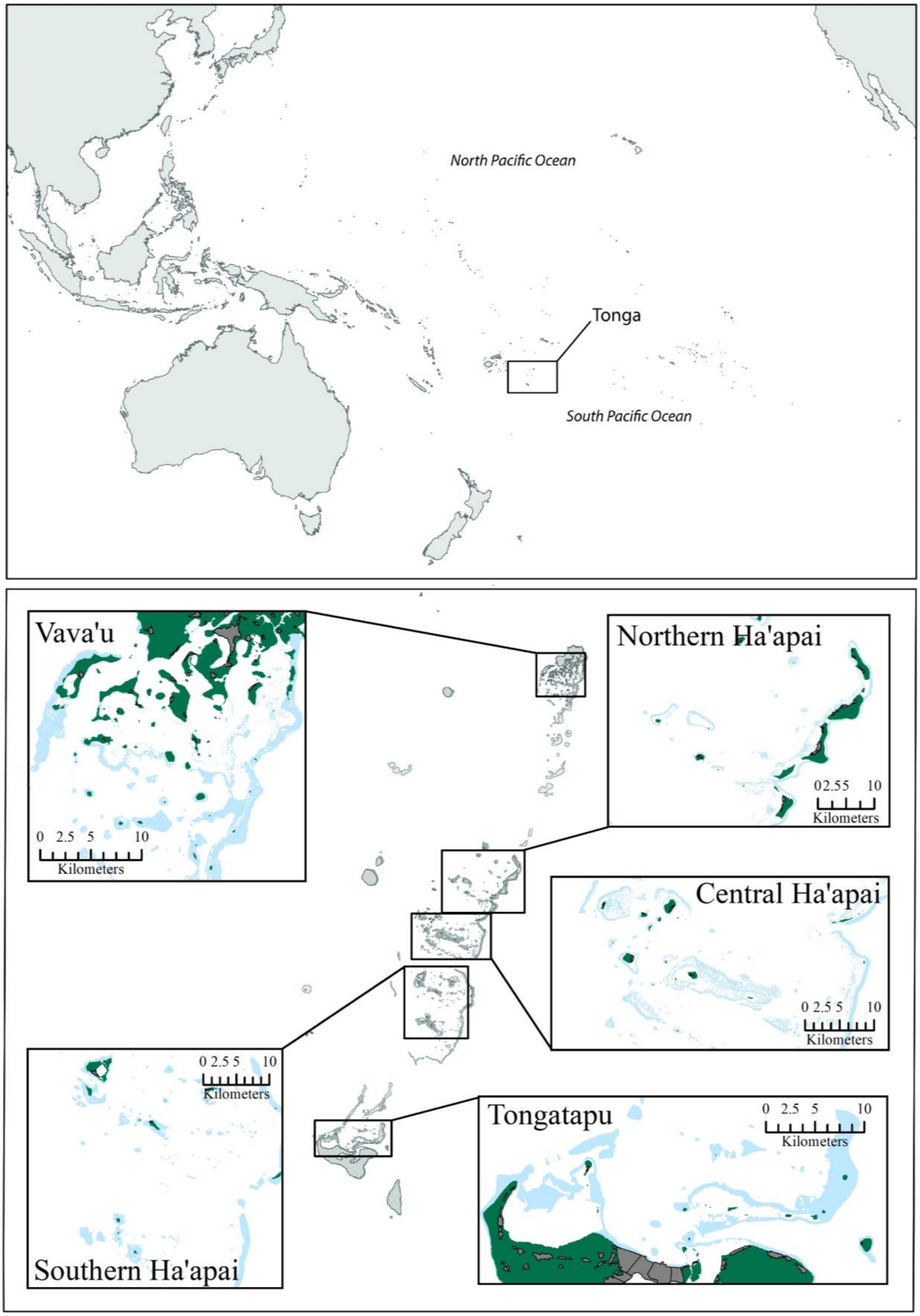

Figure 3.1. The near-shore shallow marine environment of Tonga. Light blue areas represent shallow water (0-20m), green represents land, and grey with black outlines villages.

3.3 Methods

Data sources

Eight data sources were used to develop 17 spatial layers for the near-shore marine ecosystem of Tonga (Table 3.1). Habitat and bathymetry data for Vava'u and Ha'apai, excluding the Nomuka group, were obtained from the Khaled bin Sultan Living Ocean Foundation Global Reef Expedition (KSLOF-GRE, Purkis *et al.* 2019, https://maps.lof.org/lof). Habitat information for Tongatapu and the Nomuka group were obtained from the Millennium Coral Reef Mapping Project (MCRMP, Andrefouet *et al.* 2006, http://imars.usf.edu/MC/products.html). Bathymetry for Tongatapu and the Nomuka group were obtained from satellite derived bathymetry layers developed by Land Information New Zealand (LINZ) (https://www.linz.govt.nz/) on behalf of the Tongan Government (Hartmann *et al.* 2018). While this manuscript was in review, the Allen Coral Atlas also completed coral reef habitat maps for all of Tonga. Although these layers are not included in this manuscript, they can be downloaded from the Allen Coral Atlas website (https://www.allencoralatlas.org/atlas). Net primary productivity was extracted from global layers developed by Yeager *et al.* (2017). Mean annual sea surface temperature and salinity were extracted from the MARSPEC global layers dataset developed by Sbrocco and Barber (2013). Wave energy models were developed using the University of Guam Marine Laboratory wave energy tool, which incorporates long-term wind data from the NASA QuikSCAT satellite and SeaWinds scatterometer (Jenness & Houk, 2014). The Tongan Census Bureau provided data on district-level fishing activities, while additional 2016 census data were extracted from the SPC Popgis website (Statistics Department Tonga, 2016; http://tonga.popgis.spc.int). Lastly, the Tongan Ministry of Fisheries provided up to date configurations of marine management status (as of May 2019), as well as supporting key informant interviews with local fishers required to develop fishing pressure models. Detailed methods pertaining to the creation of individual layers are provided as Supplementary Material. All layers are provided as downloadable files from the data publisher Pangaea (https://doi.pangaea.de/10.1594/PANGAEA.904800). All analysis was completed in ArcMap (10.4.1) and QGIS (2.14.20).

Extent

The extent of layers in this dataset encompasses the near-shore shallow (0-60 m) marine ecosystem of Tonga, as defined by satellite derived bathymetry (Figure 3.1; Figure 3.2). The extent of layers extracted from global datasets was sufficient to continuously cover all main island groups (total 77,000 km^2; Sbrocco and Barber 2013, Yeager *et al.* 2017).

Table 3.1. The eleven environmental and six anthropogenic variables included in this data set

Variable	Environmental/ Anthropogenic	Source	Resolution	Details
Bathymetry	Environmental	Purkis et al. (2019) and Land Information New Zealand (LINZ) (2018)	10 m	Adapted from 2 m resolution data available in Purkis et al. (2019) and LINZ (2019). 0-60 m depth available for Vava'u and northern Ha'apai by Purkis et al. (2019). 0-20 m depth available for southern Ha'apai and Tongatapu by LINZ (2018).
Coral reef density	Environmental	Current study	10 m	Total area of coral reef habitat (m^2) within 5 km and 15 km.
Distance to deep water	Environmental	Purkis et al. (2019) and LINZ	10 m	Distance to the 10 m and 20 m depth contours.
Distance to land	Environmental/ Anthropogenic	Current study	10 m	Distance (m) to the nearest land.
Distance to major terrestrial inputs	Environmental/ Anthropogenic	Current study	10 m	Distance (m) to the nearest major lagoon/estuary. Major lagoons and estuaries in Tonga include Fanga'uta lagoon, Puke to Ha'atafu estuarine area, Vaipua to Leimatua lagoon and the Makave to Ta'anea lagoon.
Habitat	Environmental	Purkis et al. (2019) and MCRMP (Also see Allen Coral Atlas)	10 m/30 m	Purkis et al. (2019) used for Vava'u and northern Ha'apai. Millenium Coral Reef Mapping Project data used for Tongatapu and southern Ha'apai. During the review process the Allen Coral Atlas also released habitat maps for all of Tonga, available for download at: https://www.allencoralatlas.org/atlas
Land area	Environmental	Current study	10 m	Total land area (m^2) within 5 km and 15 km.
Net primary productivity	Environmental	Yeager et al. (2017)	4.4 km	Net primary productivity model corrected for shallow water reflectance and incorporating satellite measurements of photosynthetically available radiation, sea surface temperature and chlorophyll a concentrations.
Salinity	Environmental	Sbrocco and Barber (2013)	1 km	Mean annual sea surface salinity from 1955-2006 (psu).
Sea surface temperature	Environmental	Sbrocco and Barber (2013)	1 km	Mean annual sea surface temperature from 2002-2010 (degrees Celsius).
Wave energy	Environmental	University of Guam Marine Laboratory Wave Energy Tool	10 m	Average daily wave energy (joules per m^2).
Distance to fish market	Anthropogenic	Current study	10 m	Distance (m) to the nearest urban fish market (Nuku'alofa, Pangai and Neiafu).
Distance to village	Anthropogenic	Current study	10 m	Distance (m) to the nearest village.
Fishing pressure	Anthropogenic	Tongan Census Bureau	10 m	Normalized (0-100) abundance of commercial and subsistence fishers (adjusted for catch) extrapolated across the coral reefs of Tonga. It constitutes a unit-less value of relative long-term fishing effort throughout the region.
Management status	Anthropogenic	Tongan Ministry of Fisheries	10 m	All Fish Habitat Reserves (FHRs) and Special Management Areas (SMAs) in Tonga as of May 2019.
Population density	Anthropogenic	Tongan Census Bureau	10 m	Total population within 5, 15 and 30 km.
Socioeconomic development index	Anthropogenic	Tongan Census Bureau	10 m	Socioeconomic development of nearby villages within 2, 5 and 10 km. Values represent a socioeconomic PCO axis (40.6%) based on village density, growth, education, age and unemployment (see supplementary materials).
Villages	Anthropogenic	Tongan Census Bureau	Polygon	Polygon of each of 142 villages in Tonga with associated census data in the attribute table

(https://doi.pangaea.de/10.1594/PANGAEA.904800)

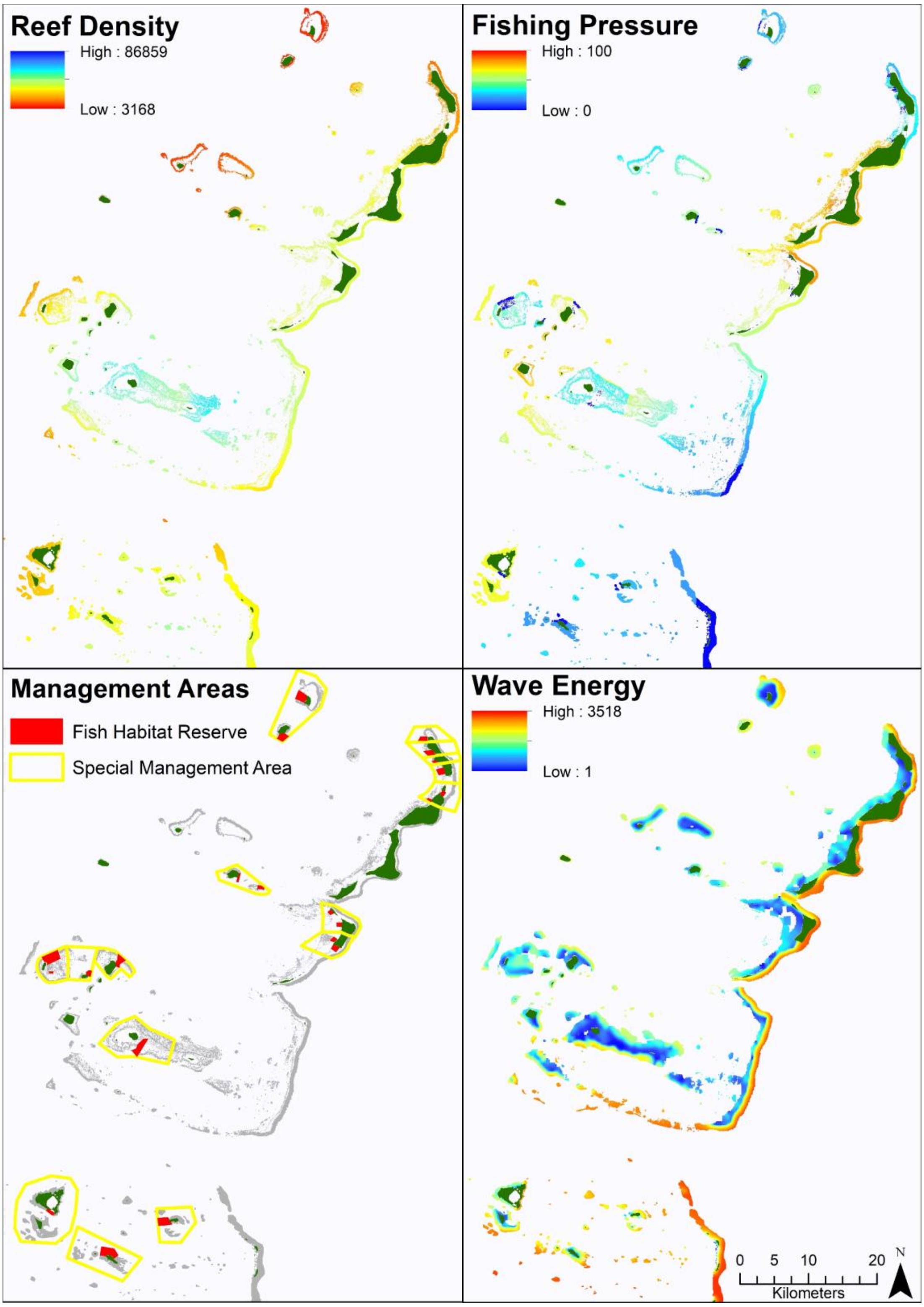

Figure 3.2. Examples of spatial layers for the Ha'apai island group of Tonga. A. Coral reef area is the amount of reef habitat (m^2) within a 15 km radius. B. Fishing pressure represents the normalized (0-100) abundance of commercial and subsistence fishers, adjusted for catch and extrapolated across the coral reefs of Tonga. It constitutes a unit-less value of relative long-term fishing effort throughout the region. C. Management status is the occurrence of Special Management Areas and Fish Habitat Reserves in Ha'apai as of May 2019. D. Wave energy represents the daily joules per m^2 of wave energy for each 10 m^2 pixel. Coloured areas represent coral reef habitat and green represents land.

3.4 Results and Discussion

Information on variables

A total of 17 socio-environmental spatial layers describing Tonga's near-shore marine environment are presented in the current dataset. This study provides vital data on >15,000 km^2 of reef habitat, and its relationship to 90% of Tonga's population and >160 individual islands. An additional layer outlining the extent of each of the 142 villages in Tonga is also provided, embedded with the associated village census data (Statistics Department Tonga, 2016; http://tonga.popgis.spc.int).

Limitations

The spatial layers presented in the current study have several limitations to be considered by end users. One consideration is that their extents cover only the near-shore marine ecosystems of Tonga and do not include offshore oceanic habitats. The reason for this is two-fold. Firstly, methods used to build several layers (e.g. wave energy, fishing pressure) depend on shallow water or reef environments and would require a different modelling approach with alternative inputs to apply to oceanic habitats (Jenness and Houk 2014). Secondly, by focusing on near-shore, shallow-water environments, this dataset can be used to complement the existing Tonga marine atlas (Gassner *et al.* 2019), which primarily describes offshore marine habitats, extending to the EEZ boundary. An additional consideration is that the current spatial layers do not constitute an exhaustive list of all environmental and anthropogenic influences on Tonga's marine environment. Impacts such as industrial development, tourism intensity and pollution can be significant drivers of ecological processes but, due to lack of data, these factors were not included in the current dataset. Lastly, the development of these layers is an ongoing and iterative process so, given time some might become outdated. For example, the Special Management Area (SMA) program in Tonga has expanded rapidly in recent years and is likely to continue to change as new SMAs are implemented. As more data become available, this dataset could therefore be expanded and revised to incorporate new and updated information to ensure this resource remains useful and relevant to end users.

Applicability

The spatial layers provided in the current study can be used to facilitate a wide range of projects within Tonga's shallow-water marine ecosystems, relating to research, conservation, management and marine industry development. The dataset will be an essential resource to assist in achieving key outputs identified by the Tongan Fisheries Sector Plan (2017) and addresses key national knowledge gaps highlighted in the Report on the National Biodiversity Strategy and Action Plan (2014). For example, with the expansion of the SMA program in Tonga, these layers could be used to assist with planning and site selection to ensure maximum long-term conservation impact of new SMAs. Similar spatial layers describing the island group of Vava'u (described by Smallhorn-West et al., 2018) were instrumental in identifying the importance of configuring no-take zones close to villages in order to increase their predicted impact on reef fish recovery. This dataset could also be used to aid in the development of Tonga's growing mabé pearl (half-pearl) industry (Gordon et al. 2018, 2019) and other aquaculture commodities through identification of suitable aquaculture sites as demonstrated previously in Japan, Italy and Brazil (Radiarta *et al.* 2008, Dapueto *et al.* 2015, de Novaes Vianna and Filho 2018, respectively). Lastly, as a research tool the spatial layers could also be combined with *in situ* ecological measurements to model various metrics of coral reef ecosystem health and species distributions. These examples constitute only a fraction of the potential projects in Tonga that could benefit from easy access to open source spatial data sets of this kind.

3.5 Conclusion

Managing and conserving Tonga's marine ecosystem requires accurate and accessible information on environmental conditions and human impacts. This resource consolidates a wide range of material and is provided in an easily accessible format that can be used for projects across many disciplines. The original layers not specifically produced by this study are also still available at their source locations, and these results are not intended to replace or discredit the exceptional efforts involved in generating those previous datasets. Rather, we hope that this extensive and accessible resource will build on previous efforts and be an essential tool for the future assessment and management of marine ecosystems in Tonga.

Chapter 4: Ecological status of Tonga's coral reefs and associated fish resources: national trends and socio-environmental drivers

P. Smallhorn-West, S. Gordon, K. Stone, D. Ceccarelli, S. Malimlai, T. Halafihi, M. Wyatt, T. Bridge, R. Pressey, G. Jones

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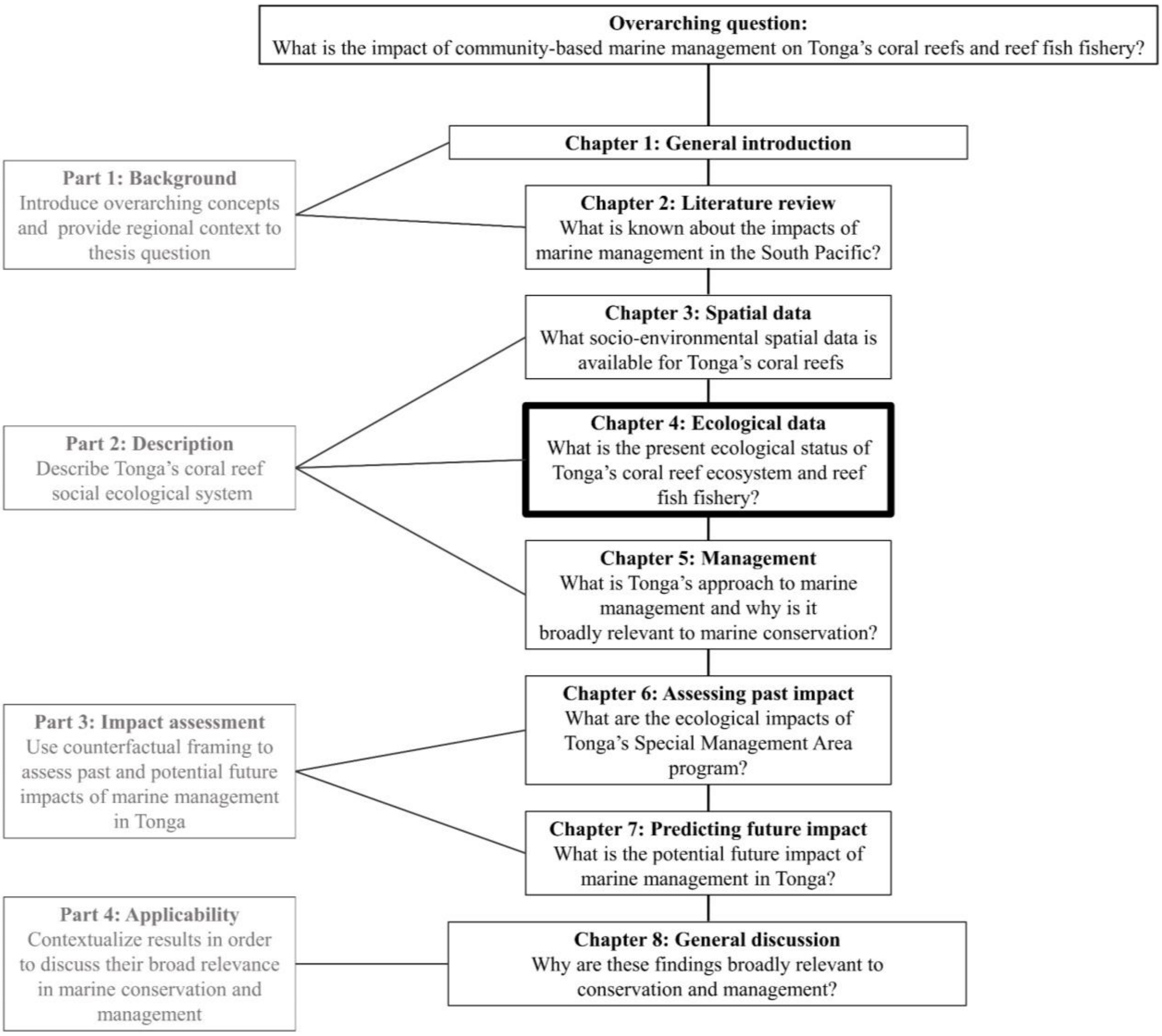


4.1 Abstract

Despite increasing threats to Tonga's coral reefs from both local (e.g. overfishing and pollution) and global (e.g. climate change) stressors, there is yet to be a systematic assessment of the status of the country's coral reef ecosystem and reef fish fishery stocks. Here, we provide a national ecological assessment of Tonga's coral reefs and reef fish fishery using ecological survey data from 375 sites throughout Tonga's three main island groups (Ha'apai, Tongatapu and Vava'u), represented by key metrics of reef health and fish resource status. Boosted regression tree analysis was used to assess and describe the relative importance of 11 socio-environmental variables on these key metrics of reef condition. Mean live coral cover across Tonga was 18%, and showed a strong increase from north to south correlated with declining sea surface temperature, as well as with increasing distance from each provincial capital. Tongatapu, the southernmost island group, had 2.5 times greater coral cover than the northernmost group, Vava'u (24.9% and 10.4% respectively). Reef fish species richness and density were comparable throughout Tongatapu and the middle island group, Ha'apai (~35 species/transect and ~2500 fish/km^2), but was significantly lower in Vava'u (~24 species/transect and ~1700 fish/km^2). Spatial patterns in the reef fish assemblage were primarily influenced by habitat-associated variables (slope, structural complexity and hard coral cover). The biomass of target reef fish was greatest in Ha'apai (~820 kg/ha) and lowest in Vava'u (~340 kg/ha), and was negatively associated with higher human influence and fishing activity. Overall mean reef fish biomass values suggest that Tonga's reef fish fishery can be classified as moderately to heavily exploited, with 64% of sites having less than 500 kg/ha. The clear effects of anthropogenic variables on observed patterns of reef condition provide strong evidence that Tonga's coral reefs are being affected by human influences such as the local overexploitation of reef fish resources. This study provides critical baseline ecological information for Tonga's coral reefs that will (1) facilitate ongoing management and research and (2) enable accurate reporting on conservation targets locally and internationally.

4.2 Introduction

Coral reefs are increasingly threatened by cumulative human-induced disturbances (Bellwood et al., 2019; Morrison et al., 2019). These range from large-scale global impacts such as climate-driven coral bleaching (Hughes et al., 2017a,b) to more local stressors such as overfishing, destructive fishing practices, and pollution (Stone et al., 2019). Furthermore, many of these impacts are increasing both in frequency and severity (Vercelloni et al., 2020). Despite the widespread acknowledgement of large-scale coral reef decline, many reef ecosystems remain poorly studied, with little data available to accurately quantify current ecosystem health (Chin et al., 2011). In addition, the specific ecological and socioeconomic factors associated with key metrics of reef health are often unknown (but see Cinner et al., 2016; Darling et al., 2019 and Jouffray et al. 2019). Managing the multiple threats facing coral reef ecosystems requires accurate data on both the status of the ecosystem and the dominant drivers of reef condition.

Assessing the ecological status of a country's coral reefs and associated fishery resources requires a comprehensive assessment of both benthic habitat structure, reef fish communities, and exploited species at large spatial scales (Knudby et al., 2010; Mellin et al., 2009). Within this context, a number of metrics are currently considered particularly important. Hard corals are the dominant ecosystem engineers on coral reefs, providing both food and three-dimensional structure for reef-associated fish and fisheries (Jones and Syms, 1998; Alvarez-Filip et al., 2013; Graham and Nash, 2013). The proportional cover of live hard corals on a reef is therefore one of the key variables used to measure reef health, as high coral cover is a generally accepted desirable state for coral reef benthic communities (Bruno et al. 2009, 2019; Ceccarelli et al., 2019). In addition, the proportional coverage of other benthic categories, such as Soft corals, Crustose Coralline Algae (CCA) and Turfing algae, are also considered important for understanding overall reef condition (Fabricius and De'Ath 2001).Given the importance placed on biodiversity conservation, reef fish species richness is also commonly used as a metric of reef status, under the assumption that areas with higher species richness are more likely to contribute to both biodiversity targets and ecosystem function (Roberts et al., 2002). The overall density of reef fish is also a common metric that is used as a proxy indicator of reef condition and to quantify differences between sites (Chapman et al., 1999). In addition to ecosystem condition, characterizing the status of multi-species fisheries, which are typical for coral reef environments, often creates challenges due to the many life-history traits within a single fishery (McClanahan et al., 2016). To handle this complexity, the biomass of target reef fish species has been demonstrated as a key proxy for the status of reef fish fisheries, with predictable declines in ecosystem condition as biomass diminishes (McClanahan et al., 2015; 2018, 2019).

Coral reef community structure and health is likely to be determined by complex interactions between socioeconomic and environmental variables that influence reef condition (Cinner et al., 2018;

Darling et al., 2019; Wedding et al., 2018). Managing reef ecosystems therefore relies not only on quantifying current reef community structure, but also on understanding the patterns and processes responsible for observed conditions (Ceccarelli et al., 2019). While investigating influences on reef community structure has been underway for decades, more recent advances in our ability to measure and analyse many socio-environmental variables concurrently has enabled simultaneous examination of the relationships and interactions between a broad range of variables (Ceccarelli et al., 2019; Darling et al., 2019). However, in developing countries, few resources are available to monitor national reef status, and management and governing authorities are often required to make wide-reaching decisions for both people and ecosystems based on limited information. When records are unavailable, reporting for both national and international commitments can rely on data of questionable quality or limited scope, resulting in false impressions of progress (Visconti et al., 2013). Good quality data at the correct spatial scale are therefore critical to maintain government accountability and understand the efficacy of management strategies.

As with many other South Pacific nations, coral reefs in Tonga are increasingly threatened (Chin et al., 2011). In the past decade alone, five severe tropical cyclones (Category 4-5) have affected Tonga (Wilma 2011, Evan 2012, Ian 2014, Wintson 2015 and Gita 2018), and coral bleaching events were reported in 2012, 14 and 16 (personal communication, Vava'u Environmental Protection Association (VEPA)). Concerns about overfishing and destructive fishing practices have also been raised for decades, with multiple management strategies employed with varying degrees of success (Gillett, 2017). Land-based pollution from agricultural runoff and illegal dumping (of both rubbish and sewage) is also a concern, particularly around lagoonal areas in the island groups of Tongatapu and Vava'u (Aholahi et al., 2017). However, few data are available to determine the consequences of these impacts for reef communities or food security (Anon, 2014).

While several local-scale projects and reports exist (Table 4.1), there is yet to be a systematic assessment of Tonga's coral reef ecosystems and reef fish fishery at the national level. The fifth national report to the Convention on Biological Diversity (Anon, 2014) described marine biodiversity trends as "not clearly defined" (page 62) and "unknown", and while "the lack of resource assessment is the key issue for [Tonga's] marine ecosystem, only few select fisheries are known" (page 59). Likewise, the status of coral reefs in the Pacific for 2011 (Chin et al., 2011) described the status, health and resilience of Tonga's coral reefs as "data deficient" or "not considered", and that "the available data are insufficient to describe the health and resilience of these reefs (and) there has been little scientific monitoring and assessment of most reef areas and many have not been mapped or surveyed" (page 199). In more recent years several expeditions have conducted ecological surveys, primarily in the Vava'u group and northern Ha'apai (summarized by Stone et al. 2019). The most notable two studies were Atherton et al. (2015) and Purkis et al. (2020). Purkis et al. (2020) surveyed coral reefs at 60 sites in Vava'u and northern Ha'apai as part of the 2013 global Living Oceans

Foundation expedition. Atherton et al. (2015) conducted a rapid biodiversity assessment (BioRap) on coral and reef fish communities at twenty-seven sites in the Vava'u archipelago.

A clear knowledge gap exists in the information available regarding Tonga's coral reefs and reef fish fishery for government, managers and other stakeholders. The aim of this study was therefore to compile and analyse the first national dataset on the current ecological status of Tonga's coral reefs (375 sites) and provide baseline ecological information that can be used to facilitate ongoing management and research. In addition, we used boosted regression trees to test the relative association of eleven socio-environmental variables with seven key metrics of reef health. Specifically, we ask: (1) What are the differences and similarities in benthic cover (hard coral, soft coral, CCA and turf algae), fish diversity and abundance, and target fish biomass among the main island groups of Tonga? and (2) What are the most influential variables associated with reef status across Tonga's coral reef ecosystem?

4.3 Methods

Survey design

From 2016 to 2018, 375 sites were surveyed across Tonga as part of four separate projects but using a standardized methodology (for individual reports see Ceccarelli 2016 and Stone et al. 2017) (Fig 4.1, Table 4.2). Underwater visual census was used to survey fish and benthic community composition around the three main island groups of Tonga: Tongatapu, Ha'apai and Vava'u. Due to the large latitudinal gradient across Ha'apai, this island group was further divided into southern, central and northern Ha'apai. All research activities were conducted in accordance with James Cook University Animal Ethic Guidelines (permit approval A2454) and approved by the Tongan Prime Minister's Office and Tongan Ministry of Fisheries.

At each site, three to six 30 m transects were deployed parallel to the depth contour in depths ranging from two to twelve meters, depending on the reef slope, depth and topography at each site. The abundance and size of all large mobile fish were recorded to the species level within a five-metre belt along each transect. All small, site-attached reef fish species were recorded along a two-metre belt width. The length and abundance of reef fish were converted to biomass following published length-weight relationships for each species (Kulbicki, Guillemot, & Amand, 2005). All data was summarized to the site level using mean values.

In situ estimates of habitat complexity (rugosity) and reef slope were also collected for each site on a five-point scale from low and sparse relief (score = 1) to exceptionally complex with numerous caves and overhangs (score = 5), and from $< 10^{o}$ (score = 1) to 90^{o} (score = 5), respectively (Gurney and Darling, 2017).

Table 4.1. Literature available on the status of Tonga's coral reef ecosystem. This list includes only publications and reports that present ecological data on metrics of reef health or reef fish fisheries. It does not include publications or reports that describe only livelihoods, fishing activities or management.

Publication	Location	Additional information
Adjeroud et al., 2013	Tongatapu	Examined spatial distribution of coral assemblages across ten sites in the lagoon of Tongatapu.
Aholahi et al., 2017	Tongatapu	Detailed current status of the Fanga'uta lagoon in Tongatapu, including benthic assemblages and water quality. Earlier reports are also available.
Atherton et al., 2015	Vava'u	BioRap rapid assessment of biodiversity surveys conducted throughout the Vava'u archipelago including reef fish, invertebrates and benthic composition.
Bruckner, 2014	Ha'apai, Niuatoputapu and Vava'u	Initial report of reef fish, invertebrates and benthic assemblages surveyed across 59 sites as part of the global Khaled bin Sultan Living Ocean Foundation reef expedition. See Purkis et al. (2020) below.
Buckley et al., 2017	Vava'u	Eleven sites established in the Vava'u archipelago as permanent benthic monitoring sites.
Chin et al., 2011	National	Synthesis as of 2011 of the current known status of Tonga's coral reef ecosystems. Conclusions about status varied between data deficient, not considered or low confidence
Ceccarelli, 2016	Vava'u	Baseline ecological surveys across 36 sites for seven Special Management Area (SMA) communities. Included benthic composition, invertebrates and reef fish. Data from these surveys are also included in this report.
Friedman et al., 2008	Two villages in each of Ha'apai and Tongatapu	Part of the PROCFish/C program to provide baseline information on the status of reef fisheries. Reef fish, benthic and invertebrate surveys were conducted around two villages in both Ha'apai and Tongatapu.
Government of Tonga, 2014	National	National report by the Tongan government to the Convention on Biological Diversity on the current status of Tonga's environment, including coral reefs. Coral reef ecosystems were classified as primarily data deficient or unknown.
Holthus, 1996	Vava'u	Coral assemblages across thirty-six sites in the Vava'u archipelago were surveyed in 1990 to determine their suitability for coral harvesting.
Kronen, 2004	Around two villages in each of Ha'apai, Tongatapu and Vava'u	Underwater visual census of target reef fish and total reef fish size, density and diversity were conducted around several villages in each island group.
Lovell & Palaki, 2000	National	Ecological surveys conducted of benthic assemblages and reef fish, although extent is unclear.
Malimali, 2013	Five communities across Ha'apai, Tongatapu and Vava'u and associated comparison sites	Reef fish, invertebrates and benthic composition were compared between managed and open areas for five communities as part of PhD thesis.
Mayfield et al., 2017	Ha'apai, Niuatoputapu and Vava'u	Part of the Khaled bin Sultan Living Oceans Foundation surveys of 59 reefs in Tonga. *Pocillopora damicornis* and *Pocillopora acuta* colonies were sampled to determine whether they differed physiologically although being difficult to distinguish *in-situ.*
Pakoa et al., 2008	Tongatapu lagoon	Extensive ecological surveys conducted of invertebrates and benthic composition around the Tongatapu lagoon, with an emphasis on their relevance for the Trochus fishery.
Purkis et al., 2020	Ha'apai, Niuatoputapu and Vava'u	Final report of reef fish, invertebrates and benthic assemblages surveyed across 59 sites as part of the global Khaled bin Sultan Living Ocean Foundation reef expedition. See Bruckner (2014) above.
Richardson, 2010	Five SMA communities across Ha'apai, Tongatapu and Vava'u	Ecological surveys of benthic community composition around five SMA communities and comparison sites.
Smallhorn-West et al. 2019	Vava'u	Publication as part of this project and data therefore included in this analysis. Predicted the potential recovery of target species biomass under various protected area configurations based on data from 129 sites in Vava'u.
Smallhorn-West et al. 2019	Southern Ha'apai	Specific surveys of six coral reef sites around the newly erupted Hunga-Tonga Hunga-Ha'apai volcano near southern Ha'apau. Data were not included in this analysis.
Smallhorn-West et al. 2020	Same data as this manuscript	Impact evaluation of seven Special Management Areas (SMA) in Tonga.
Smallhorn-West et al. 2020	Same data as this manuscript	Public report on baseline reef condition throughout Tonga.
Stone et al., 2017	Ha'apai and Vava'u	Reef fish, invertebrate and benthic community composition across 56 sites as part of the WAITT Institute Vava'u Ocean Initiative. Data from these surveys are included in this report.
Stone et al., 2019	National	Reviews the current known status of coral reefs in Tonga prior to the surveys used in this report. Conclusions derived mainly from Atherton et al. (2015).
Vieux et al., 2005	National	Discusses monitoring in the South Pacific, including Tonga. Concludes that while "efforts are now under way to conduct baseline and monitoring studies … there are considerable constraints due to poor capacity for monitoring, surveillance and enforcement".
Vieux, 2005	Two villages in Vava'u	Reef fish, invertebrate and benthic community composition at two villages in Vava'u.

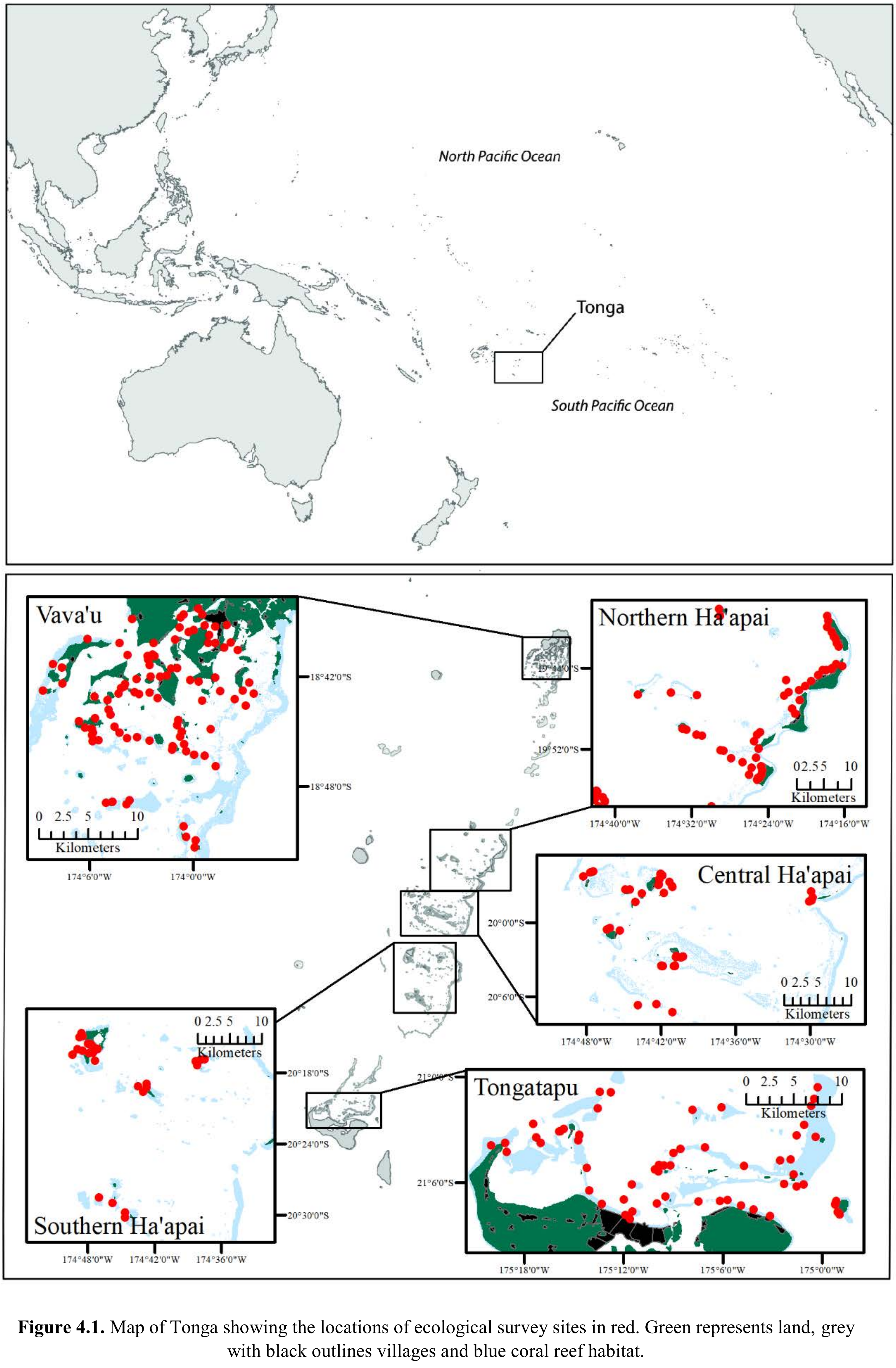


Figure 4.1. Map of Tonga showing the locations of ecological survey sites in red. Green represents land, grey with black outlines villages and blue coral reef habitat.

Table 4.2. Summary of fish survey data sets available to the project. ARC CoE CRS = Australian Research Council Centre of Excellence for Coral Reef Studies. ADB = Asian Development Bank. VEPA = Vava'u Environmental Protection Agency.

Project	Department	Funding	Surveyor	Island group	Number of sites	Year
James Cook University National monitoring project	Ministry of Fisheries	James Cook University	Patrick Smallhorn-West	Tongatapu	60	2018
		ARC CoE CRS		Ha'apai	125	2018
		McIntyre Adventure/ Halaevalu Mata'aho Marine Discovery Centre National Geographic Society		Vava'u	93	2017
ADB Vava'u Special Management Areas baseline surveys (Ceccarelli, 2016)	Ministry of Fisheries	ADB	Dr. Daniela Ceccarelli	Vava'u	36	2016
	Department of Environment		Karen Stone			
	VEPA					
VEPA Special Management Areas baseline surveys	VEPA	VEPA	Karen Stone	Vava'u	4	2017
WAITT Institute field surveys (Stone et al., 2017)	Department of Environment	WAITT Institute	Heather Kramp	Ha'apai	18	2017
	VEPA		Karen Stone	Vava'u	39	2017

Benthic composition

Benthic community composition was estimated using image analysis of ten 1 x 1 m benthic photoquadrats per transect with 15 points randomly overlaid across each image (total 150 points per transect). Given the large number of images and points required for annotation (images = 11020; points = 165300), we used the machine learning software BenthoBox (www.benthobox.com) to assist with the benthic annotations. BenthoBox automatically classifies points into benthic substrate categories from images based on training provided by a human annotator. The aim of the automated annotation method is to learn from human annotations and automatically analyse the remaining images to within an acceptable margin of error (Beijbom, 2015a,b). While automation typically captures similar trends but with higher variability than among human annotators (Beijbom et al. 2015; González-Rivero et al., 2016), the impact of this error on interpretation depends on the relative abundance of organisms, taxonomic resolution and ecological relevance of the variables in question. Typically, the noise around automated annotations may lead to misinterpretations of rare categories (<5 % total cover) for which the average abundance is similar to the error in quantification. However, the impact of automated analysis error on more dominant benthic groups (>5 % total cover) is less pronounced, and usually has marginal effects on derived cover estimates (González-Rivero et al., 2016). For the purposes of this study we therefore included four common benthic categories each with mean cover greater than five percent: Hard Coral, Soft Coral, CCA and Turf Algae. Details of the automated annotation process are available in the supplementary materials (supplementary materials).

A small subset of benthic data was annotated using the point intercept method when photographic equipment was not available. When this was the case a single benthic point sample recorded every 50 cm (60 samples per transect; n=320 transects from 61 sites). In addition, no benthic data were collected at 32 sites surveyed in the Vava'u island group and for these only fish-related analysis was conducted

Metrics of reef status and socio-environmental predictor variables

Seven metrics of reef status were used to assess the current ecological condition of Tonga's coral reef ecosystem and reef fish fishery. Based on acceptable error rates in benthic annotations, hard coral, soft coral, CCA and Turf algae were included as benthic response variables. For the status of coral reef fishes and Tonga's reef fish fishery we included total reef fish species richness (n/transect) and density (n/km^2) and the biomass of target species (kg/ha) larger than 20 cm. We selected this size cut off for biomass as this represents the fishable biomass of target reef fish species that is likely to be targeted by fishers.

Eleven socio-environmental variables known to affect coral reef community structure across Tonga were selected as potential explanatory variables (Table 4.3). Three of these were collected *in situ* (depth, rugosity and slope) and seven were spatially continuous across Tonga's coral reefs (Cyclone occurrence, Distance from provincial capital, Fishing pressure, Land area, Reef density, SST and Wave energy). Details of how these variables were calculated are available in the supplementary materials. In order to control for potential differences in sampling protocols, project (ADB, JCU or WAITT) was also included as an explanatory variable. Due to the small sample size the VEPA surveys were combined with the WAITT surveys, which happened at a similar time and with the same personnel. Finally, for reef fish metrics total hard coral cover was also included as an additional explanatory variable.

Table 4.3. Eleven socio-environmental variables included as potential influences on reef condition in Tonga. Details of their development are available in the supporting information

Variable	Description
Cyclone occurrence in the past 18 months	Occurrence of sustained wind speeds above 50 knots (category 2 cyclone) within the past 18 months.
Depth	Depth (m), collected *in situ*.
Distance from provincial capital	Distance (km) from the nearest provincial capital town (Tongatapu – Nuku'alofa, Ha'apai – Pangai and Vava'u – Neifau). The provincial capitals are both the main population centres for each island group and the locations of the main fish markets (Chapter 3).
Fishing pressure	Normalized (0-100) abundance of commercial and subsistence fishers (adjusted for catch) extrapolated across the coral reefs of Tonga. It constitutes a unit-less value of relative long-term fishing effort throughout the region. This fishing pressure metric also accounts for differences in fishing pressure due to management within marine protected areas (Table S1)(Chapter 3).
Habitat rugosity	Estimate of habitat complexity collected *in situ* on a five-point scale from low and sparse relief (score = 1) to exceptionally complex with numerous caves and overhangs (score = 5).
Hard coral cover	Percent total live hard coral cover. Only included for reef fish variables.
Land area within 5 km	Terrestrial influence calculated as the amount of land (km^2) within a 5 km radius of each $10m^2$ reef pixel (Chapter 3).
Reef density within 5 km	Calculated as the amount of reef habitat (km^2) within a 5 km radius of each $10m^2$ reef pixel (Fig S5).
Slope	Estimate of reef slope collected *in situ* on a five-point scale from < 10^o (score = 1) to 90^o (score = 5).
Sea surface temperature (SST)	Mean annual sea surface temperature from 2002-2010 (degrees Celsius)(Chapter 3).
Wave energy	Average daily wave energy (joules per m^2) (Chapter 3).

Data analysis

The four variables of reef condition were compared between island groups using generalized linear models with Tukey's post hoc comparisons. Models were fit with either raw data, log, or log(x+1) transformations and model performance assessed by i) comparing AIC scores ii) visual inspections of qqplots and plotted fitted vs. residuals and iii) calculating goodness of fit and overdispersion. Analysis outputs are available in supplementary materials (supplementary materials). Patterns in the socio-environmental variables across island groups were explored using principal component ordination (PCO) of normalized data based on Euclidean distance with Primer-e Version 6.

Drivers of reef condition in Tonga were then explored using boosted regression tree (BRT) models (Elith et al., 2008; Elith & Leathwick, 2017). All BRT models were fitted using the *gbm.step* routine in the *dismo* package (Jane Elith & Leathwick, 2017) and the *ggBRT* package (Jouffray et al. 2019) within the R statistical and graphical environment (R Core Team 2016). BRTs fit a large succession of simple regression trees that each learn only a small fraction of the data, but with each successive tree focusing on the remaining most prominent patterns. By shrinking the contributions of many trees, BRTs are generally able to make accurate predictions from complex data sets (Ceccarelli et al., 2019). Overfitting can be countered through cross-validation, which strikes a balance between predictive performance and model fit (Hastie et al., 2011). BRTs are useful for exploring the relative impacts of a large number of predictors since, unlike linear models, they are not reduced to low-level approximations of system complexity. While BRT predictions are robust to multicolinearity and non-linearity (Ceccarelli et al., 2018), the relative influence of highly correlated variables (>0.6) can be pooled into one of the variables. Therefore, a correlation matrix was used to determine whether any combination of predictor variables was highly correlated (supplementary materials).

Optimal model parameters (bag fraction, tree complexity and learning rate) for each BRT were determined by running all iterations and selecting the one with the greatest explained deviance and a minimum of 1000 trees (supplementary materials) (Pittman & Brown, 2011). Based on histograms, BRTs for hard coral, soft coral, CCA, reef fish density and target biomass were analysed using a Poisson distribution, while turf algae and reef fish species richness were analysed using a gaussian distribution. Model performance was assessed by 10-fold cross-validation, which tests the model against withheld portions of the data. Following Jouffray et al. (2019), cross-validated percent explained deviance was calculated as (1-(cross-validated deviance/mean total deviance)). Spatial autocorrelation was assessed by estimating Moran's I from the model residuals (supplementary materials).

The relative importance of each predictor variable was calculated as the frequency of splits involving each variable weighted by the associated square improvement in the model averaged over all trees and scaled out of 100 such that larger values signify stronger influence (Ceccarelli et al. 2018). Since BRTs do not provide significance tests, but only variables' relative contribution to the model's predictive power, those that were disproportionately represented in the trees (i.e. above the threshold of 100%/n variables) were considered highly influential (Ceccarelli et al., 2019; Jouffray et al., 2019). Partial dependency plots with 95% confidence intervals obtained from 1000 bootstrap replicates were used to examine the relationships between the response and the most influential predictor variables, while keeping all other predictors at their mean (Buston and Elith 2011; Jouffray et al. 2019). The presence of interactions between influential variables were also examined and plotted following Elith et al. (2008).

4.4 Results

Spatial variability in reef status and structure

Overall, mean hard coral cover across the 343 sites in Tonga for which benthic data was available was 18% (+/- .625 SE). However, mean hard coral cover in Vava'u was less than half that of the other island groups (Fig 4.2; supplementary materials), and this pattern was also similar for soft coral. Many sites in Vava'u had 0% live coral cover, particularly around the inner islands where turfing algae and bare matrix were dominant, and coral cover generally increased towards the outer islands. While a few sheltered, inner island sites in Vava'u had hard coral cover >30%, this was dominated by two species from a single genus (*Porites rus* and *Porites cylindrica*). Coral reef communities around many of the shallow, fringing reefs in the inner, enclosed areas of Vava'u appeared to be characterized by little or no hard or soft coral cover. Sites near the mouths of the two large estuarine lagoons in Vava'u often had 0% live coral cover and large numbers of *Diadema sp.* sea urchins, which appeared to be destroying the reef matrix. Coral cover in the Ha'apai island group increased gradually from north to south, with exposed sites in the southern islands (e.g. Nomuka, Mango and Fonoi) having the greatest cover of the sites assessed. Likewise, sites along the outer western islands (e.g. Ofalanga, Mounga'one, Kotu and Muitoa) generally had greater coral cover than the sheltered sites along the margins of the ribbon islands in north eastern Ha'apai (e.g. Foa and Lifuka). There was widespread evidence of damage from multiple cyclones and bleaching events along the western, sheltered edges of the north east ribbon islands (Ha'ano, Foa, Lifuka and Uoleva). With the exception of southern Ha'apai, live coral cover in Tongatapu and near the capital Nuku'alofa was consistently greater than elsewhere in Tonga. Most sites within the central bay, and even fringing reefs adjacent to the city centre, had moderate coral cover. As in Ha'apai, there was evidence of bleaching damage along back reefs of the north eastern ribbons from Tao to Nuku island. As in Vava'u, near the mouth of the Fanga'uta lagoon, Tongatapu, there were large numbers of *Diadema sp.* sea urchins and very low (often 0%) live coral cover.

Patterns of CCA cover did not vary significantly between island groups in Tonga. Conversely, there were substantial differences in the mean cover of turfing algae throughout Tonga. These patterns were largely the inverse of live coral cover, with the greatest cover in Vava'u and lowest in Southern Ha'apai.

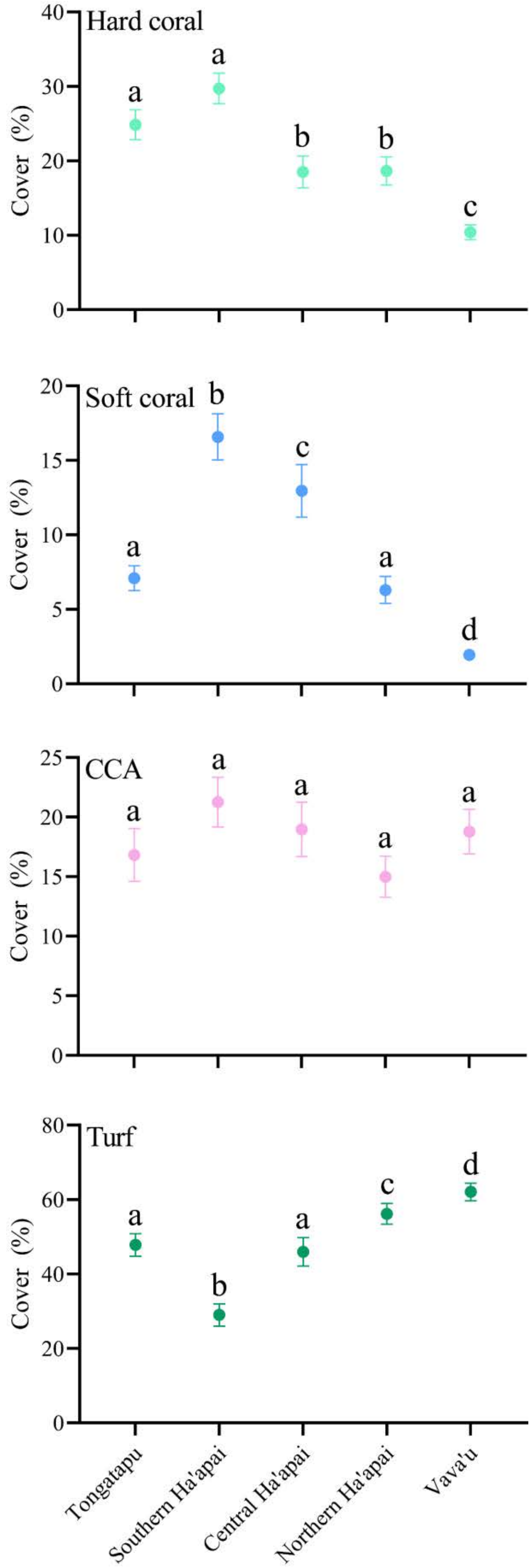


Figure 4.2. Patterns in benthic cover across the three main island groups of Tonga, arranged from south to north. Due to high latitudinal variation within the Ha'apai group, it was split into southern, central and northern Ha'apai. Values represent mean ± 95% confidence intervals. Letters denote significant groupings based on Tukey's post hoc comparisons.

A total of 510 individual reef fish species (supplementary materials) were identified throughout the surveys, and both species richness and density varied significantly between island groups (Fig 4.3; supplementary materials). However, post hoc analysis revealed that only the Vava'u island group clustered separately for both species richness and density and that there was little variation between the other island groups. The overall mean biomass of target species also varied between regions, with Vava'u having the lowest standing biomass in the country. However, there was also high variability in biomass within the other island groups, with southern and northern Ha'apai having the greatest standing biomass (963± 183 SE kg/ha and 914 ± 73 SE kg/ha respectively).

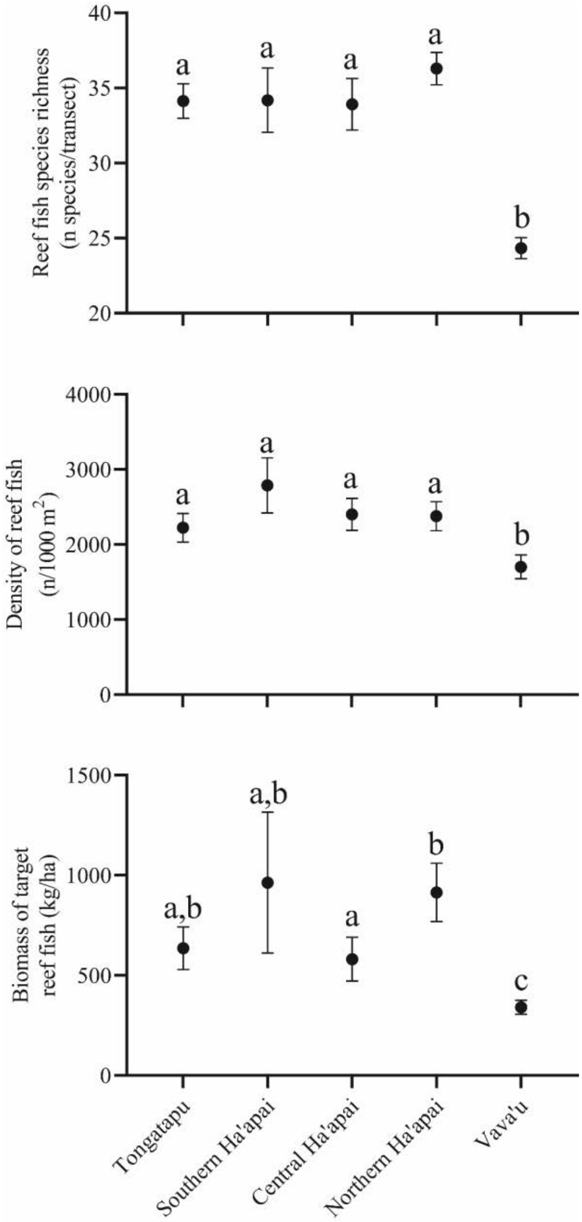


Figure 4.3. **Patterns of reef fish species richness, density and target biomass across Tonga's three main island groups arranged from south to north.** Due to high latitudinal variation within the Ha'apai group it was further split into southern, central and northern Ha'apai. Values represent mean ± 95% confidence intervals. Letters denote significant groupings based on Tukey's post-hoc comparisons.

Principal component ordination demonstrated clustering between island groups, although there is also substantial overlap and within island group variability (Fig. 4.4). Fishing pressure is substantially greater in Tongatapu than elsewhere. Both reef density and hard coral cover are greatest in Southern Ha'apai and Tongatapu. Sites in Southern Ha'apai are also the most remote as measured by distance from the provincial capital, and have the greatest wave energy. The greatest differences between Vava'u and elsewhere in Tonga are the warmer SST (by 2°C) and the low density of reef habitat.

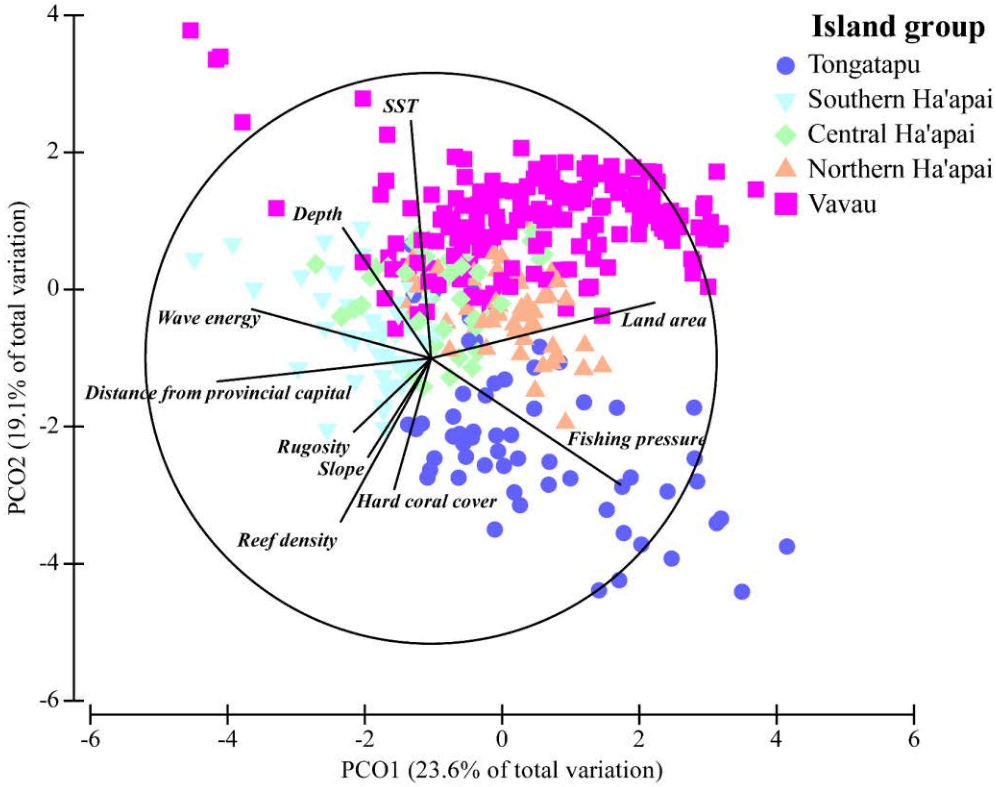


Figure 4.4. Principal component ordination of the distribution of socio-environmental variables across Tonga's island groups. PCO run on normalized data using Euclidean distances.

Predicting reef condition

BRT models performed well for all seven models, with explained deviance between 29.55% and 68.35%. Spatial autocorrelation was also low, with a maximum Moran's I values of 0.07. For each of the seven variables, figures are included which describe i) the spatial distribution of observed values across all surveyed sites, ii) the relative influence of each predictor variable, iii) the relationships of the most influential predictors, and, iv) significant interactions between influential variables.

Benthic variables

The five most influential predictors of hard coral cover in Tonga were SST, distance from the provincial capital, habitat rugosity, reef density and wave energy (Fig 4.5). Negative relationships were observed between hard coral cover and SST, reef density and wave energy. Positive relationships occurred between live hard coral cover and distance from the provincial capital and habitat rugosity. Four interactions between influential variables were present. Taken together, the partial plots and interactions predict that hard coral cover will be greatest in areas far from the provincial capitals, with high rugosity, lower SST and low reef density. The model explained 37% of the cross-validated deviance.

The three most influential predictors of soft coral cover in Tonga were SST, distance from the provincial capital, and wave energy (Fig 4.6). There was a strong positive relationship between soft coral cover and the distance from the provincial capital. As with hard coral, a negative relationship was observed between soft coral cover and SST. Unlike hard coral cover, soft coral cover was positively associated with increased wave energy. There were interactions between all three influential variables. Taken together, the partial plots and interactions indicate that soft coral cover is greatest at remote sites with high wave energy and cooler temperatures. The model explained 58% of the cross-validated deviance.

The six most influential predictors of CCA in Tonga were habitat rugosity, distance from the provincial capital, reef density, depth, land area and SST (Fig. 4.7). The two most influential predictors, rugosity and distance from the provincial capital, both had strong positive relationships with CCA. While reef density was influential at predicting CCA cover, there was not a clear pattern in the direction of the relationship. CCA cover was lowest around five meters depth, and increased towards shallower and deeper water. Model predictions suggest that CCA cover has a strong negative relationship with levels of terrestrial influence and very low levels (<0.05km^2), but that this relationship breaks down with greater terrestrial influence. As with coral cover, SST was negatively associated with the percent cover of CCA. Three variables had interactions with habitat rugosity (distance from provincial capital, land area and SST), all of which predicted greater cover of CCA at higher rugosity levels. The model explained 51% of the cross-validated deviance.

The five most influential predictors of turf algae cover in Tonga were distance from the provincial capital, habitat rugosity, SST, depth and land area (Fig. 4.8). However, the relationship between turf algae and the predictor variables was the opposite compared to other benthic variables. Turf algae coverage was greatest close to each provincial capital and declined with increasing distance from human influence. Likewise, lower levels of rugosity had the greatest cover of turfing algae. SST and land area were both positively associated with turf cover, although for land area, as with CCA, the relationship was greatest at low levels (<0.05 km^2) and plateaued at levels greater than this. Depth displayed the opposite relationship to turf algae than CCA, with greatest cover at 5 meters depth and lower levels in shallower and deeper water. There were five interactions that predicted turf cover. Taken together, these all suggest that turf algae is most dominant in shallow, low complexity reefs that are close to human influence and in warmer waters. The model explained 54% of the cross-validated deviance.

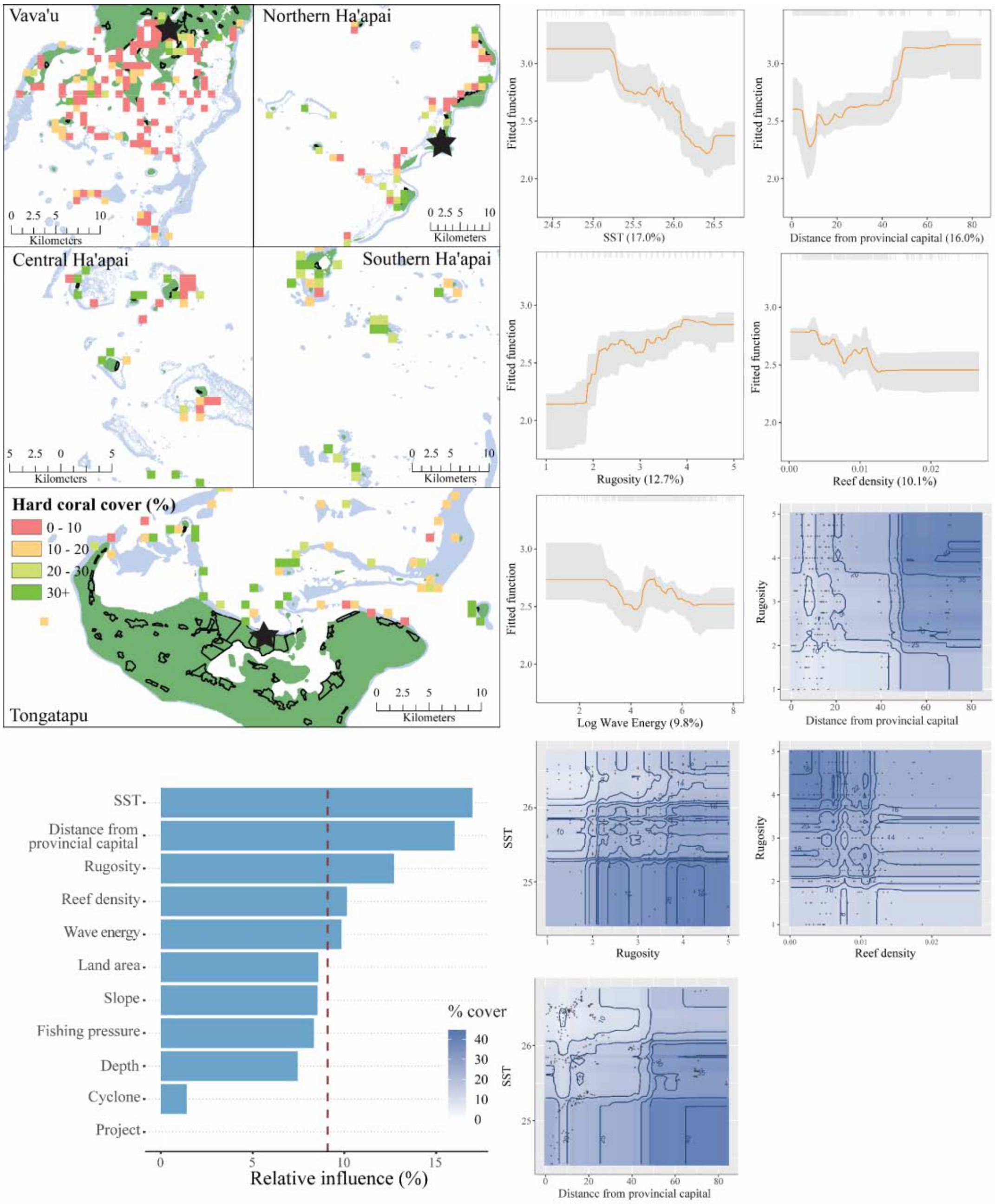


Figure 4.5. Hard coral cover. Top left: Map of hard coral cover at sites sampled across Tonga. Light blue represents reef, green land and black outlines villages. Each provincial capital is marked by a black star. **Bottom left:** Relative influence of the 11 predictor variables included in the Boosted Regression Tree (BRT). **Top right:** Partial dependency plots with 95% confidence intervals for the most influencial variables predicting hard coral cover. The plot shows the effect of each predictor on the repsonse while all other variables are at their mean. Relative influence of each predictor is reported in parentheses. Grey tick marks across the top of each plot indicate observed data points. **Bottom right:** Interactions plots of the strongest pairwise interactions between influencial variables. Contour lines indicate model predictions and points represent observed data. Units are as follows: SST – ^{O}Celcius, Distance from provincial capital – km, Rugosity – 1-5, Reef density – km^2, Log wave energy – Joules per m^2.

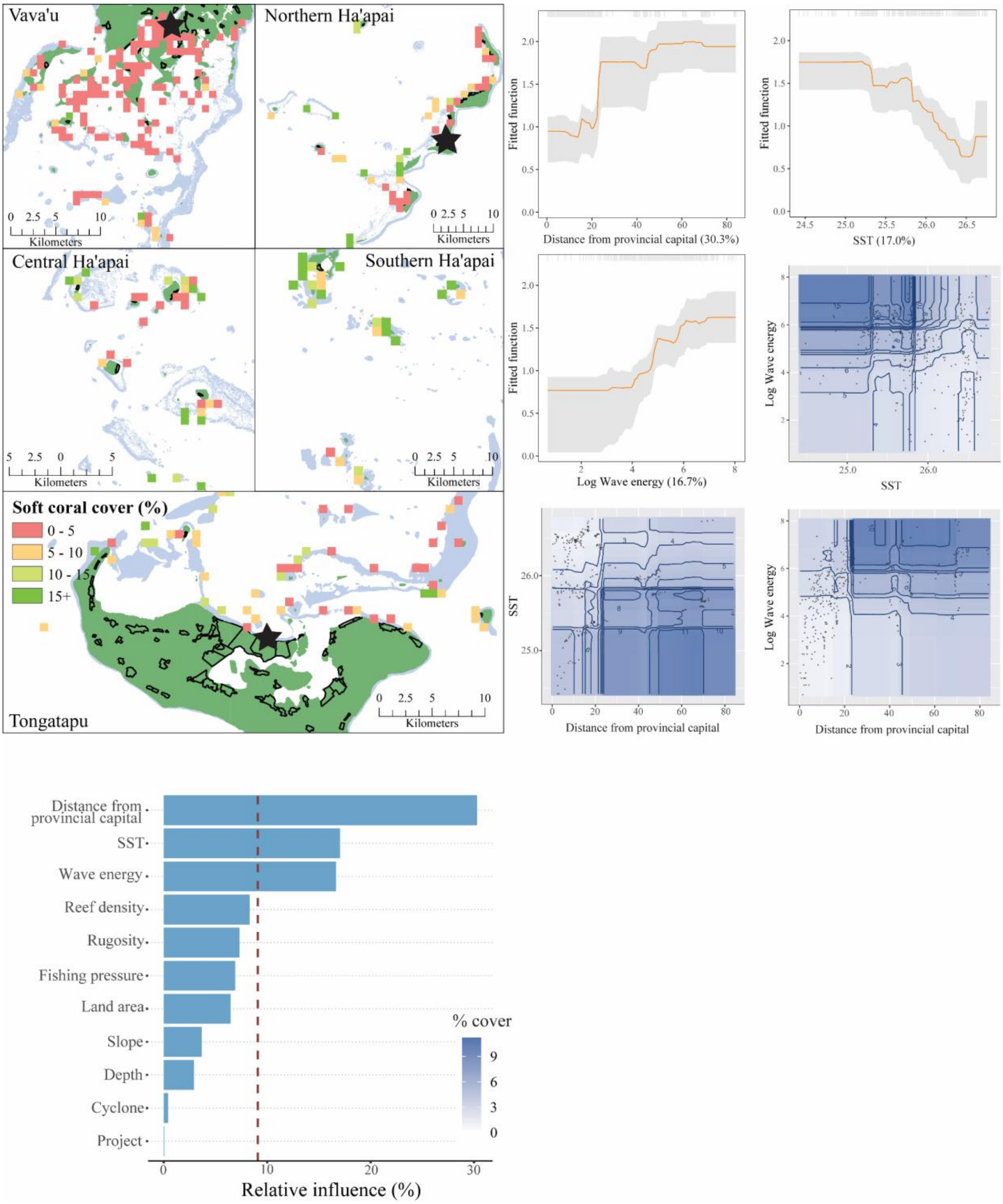


Figure 4.6. Soft coral cover. Top left: Map of soft coral cover at sites sampled across Tonga. Light blue represents reef, green land and black outlines villages. Each provincial capital is marked by a black star. **Bottom left:** Relative influence of the 11 predictor variables included in the Boosted Regression Tree (BRT). **Top right:** Partial dependency plots with 95% confidence intervals for the most influencial variables predicting soft coral cover. The plot shows the effect of each predictor on the repsonse while all other variables are at their mean. Relative influence of each predictor is reported in parentheses. Grey tick marks across the top of each plot indicate observed data points. **Bottom right:** Interactions plots of the strongest pairwise interactions between influencial variables. Contour lines indicate model predictions and points represent observed data. Units are as follows: Distance from provincial capital – km, SST – ^{O}Celcius, Log wave energy – Joules per m^2.

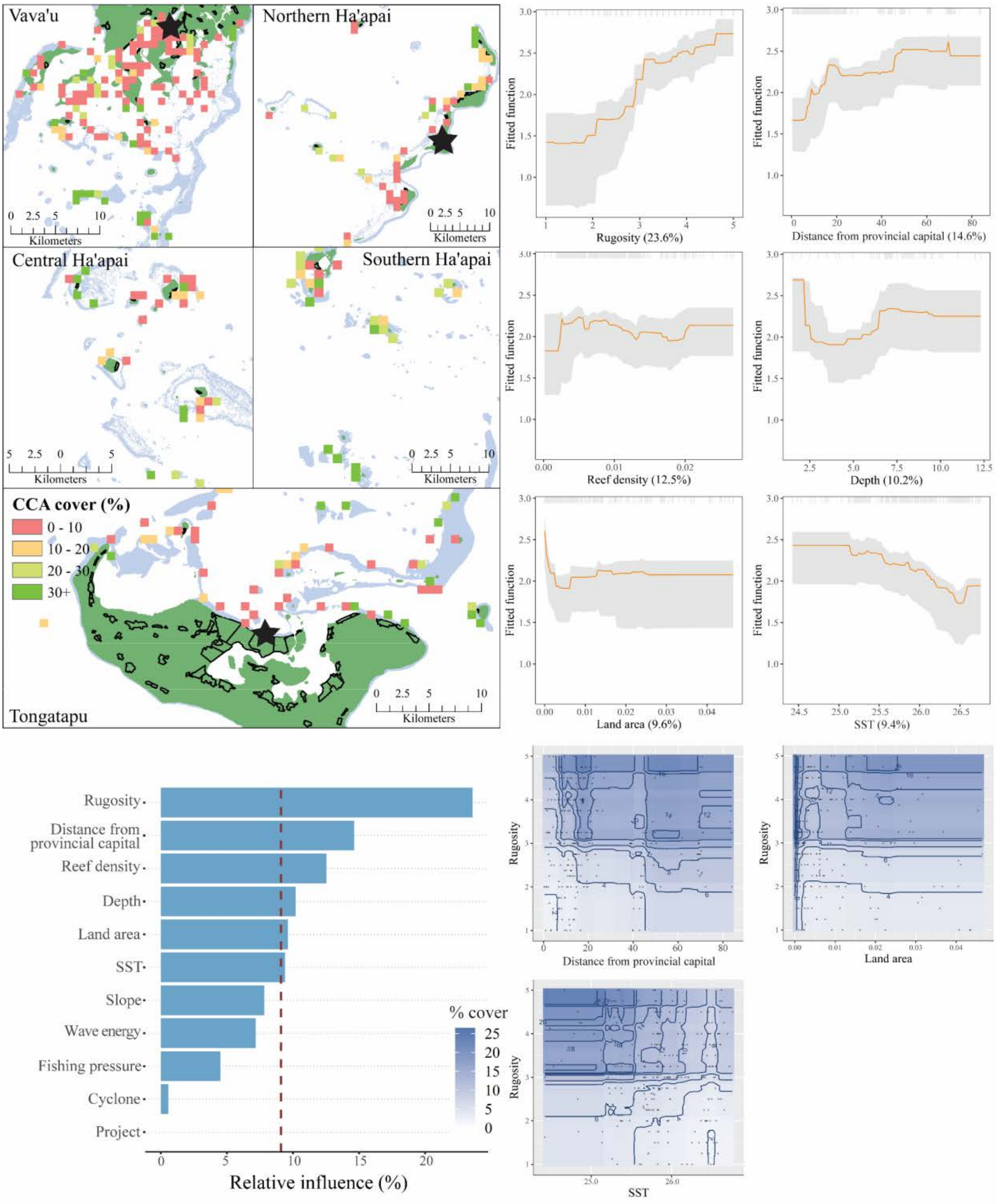


Figure 4.7. CCA cover. Top left: Map of CCA cover at sites sampled across Tonga. Light blue represents reef, green land and black outlines villages. Each provincial capital is marked by a black star. **Bottom left:** Relative influence of the 11 predictor variables included in the Boosted Regression Tree (BRT). **Top right:** Partial dependency plots with 95% confidence intervals for the most influencial variables predicting CCA cover. The plot shows the effect of each predictor on the repsonse while all other variables are at their mean. Relative influence of each predictor is reported in parentheses. Grey tick marks across the top of each plot indicate observed data points. **Bottom right:** Interactions plots of the strongest pairwise interactions between influencial variables. Contour lines indicate model predictions and points represent observed data. Units are as follows: Rugosity – 1:5, Distance from provincial capital – km, Reef density – km^2, Depth – meters, Land area – km^2, SST – ^{O}Celcius.

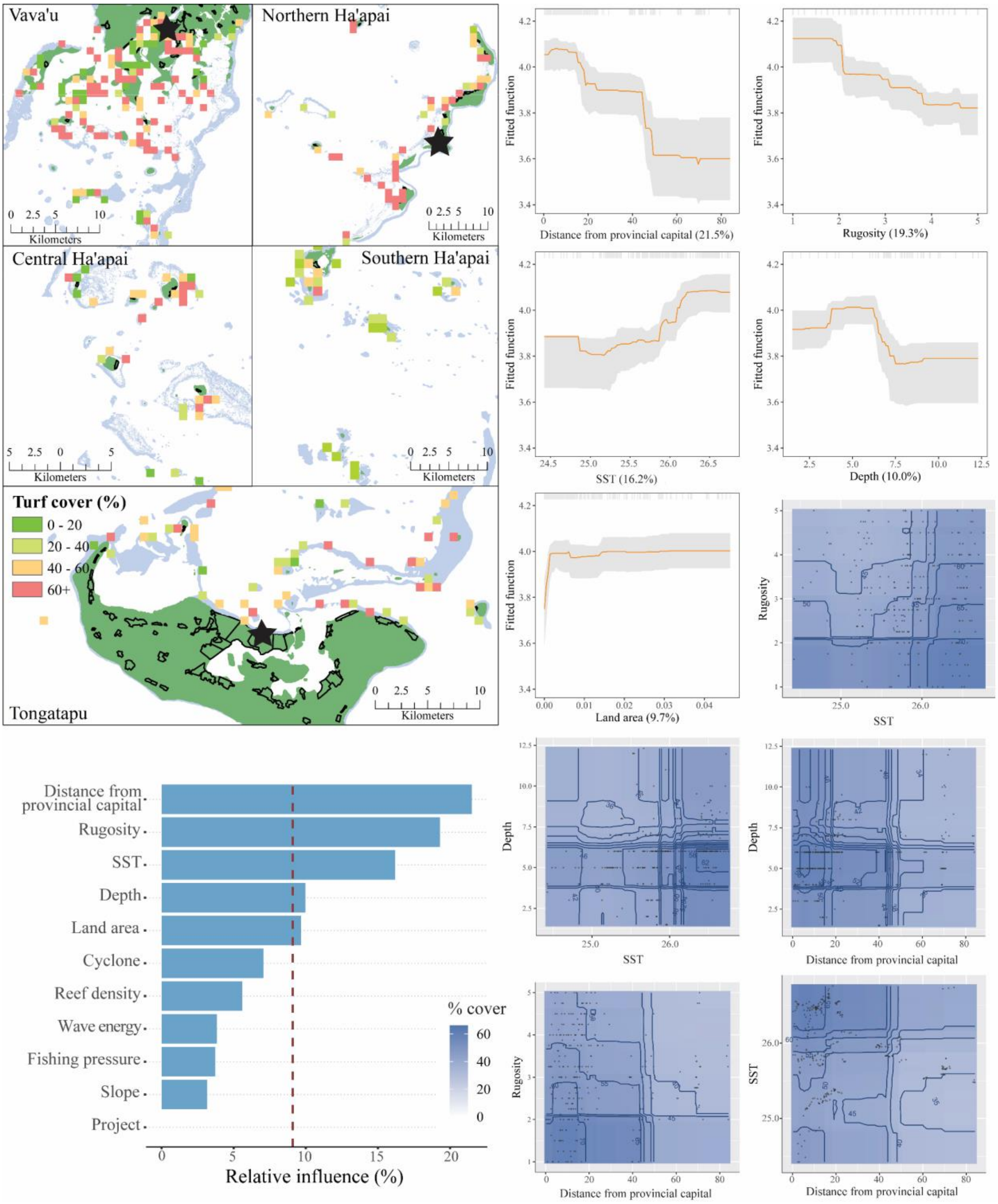


Figure 4.8. Turf algae cover. Top left: Map of Turf cover at sites sampled across Tonga. Note the colour scale here is the inverse of other variables. Light blue represents reef, green land and black outlines villages. Each provincial capital is marked by a black star. **Bottom left:** Relative influence of the 11 predictor variables included in the Boosted Regression Tree (BRT). **Top right:** Partial dependency plots with 95% confidence intervals for the most influencial variables predicting turf algae cover. The plot shows the effect of each predictor on the repsonse while all other variables are at their mean. Relative influence of each predictor is reported in parentheses. Grey tick marks across the top of each plot indicate observed data points. **Bottom right:** Interactions plots of the strongest pairwise interactions between influencial variables. Contour lines indicate model predictions and points represent observed data. Units are as follows: Distance from provincial capital – km, Rugosity – 1:5, SST – ^{O}Celcius, Depth – meters, Land area – km^2.

Fish variables

The four most influential predictors of reef fish species richness were habitat rugosity, hard coral cover, distance from the provincial capital and project (Fig. 4.9). Species richness increased substantially with rugosity values between one and three, but plateaued above three. The relationships between reef fish species richness and both hard coral cover and distance from the provincial capital were similar and positive, with increased in richness up to ~40% live coral cover and 30 km from the capital, before they also plateaued. Lastly, surveys conducted under the James Cook University led project consistently recorded a greater number of species than the other projects. Interactions were fitted between project and all three other influential predictor variables, which show that the same pattern is evident across projects despite this inconsistency. There was also an interaction between distance from provincial capital and rugosity, with complex reefs in remote areas having greater species richness. The model explained 68% of the cross-validated deviance.

The four most influential predictors of reef fish density were hard coral cover, reef slope, habitat rugosity and reef density (Fig. 4.10). Reef fish density increased substantially at both 20% and 40% hard coral cover. Density was also greatest at mid-levels of reef slope, which correspond to ~45°. More complex reefs, with rugosity scores above 3 also had greater densities of reef fish. The relationship between reef fish density and reef density was slightly negative. Three interactions between influential variables were present. Taken together, the partial plots and interactions predict that reef fish density is greatest with high coral cover and mid sloped reefs with relatively low reef density nearby. The model explained 30% of the cross-validated deviance.

The six most influential predictors of target species biomass were habitat rugosity, distance from the provincial capital, wave energy, hard coral cover, land area and fishing pressure (Fig. 4.11). Biomass increased consistently with increasing rugosity. The relationship between biomass and distance from the capital of each island group was positive, although the greatest increase was at distance greater than 60 km away. The relationship between biomass and wave energy was also positive. While hard coral cover was an influential predictor of biomass, the relationship was unclear. Both land area and fishing pressure displayed similar patterns in their relationship with biomass, with strong declines at low levels (land area <0.05 and fishing pressure <25), followed by plateaus. Four interactions between influential variables were present. Taken together, the partial plots and interactions predict that target fish biomass is greatest in high wave energy, structurally complex reef far from human pressures. The model explained 46% of the cross-validated deviance.

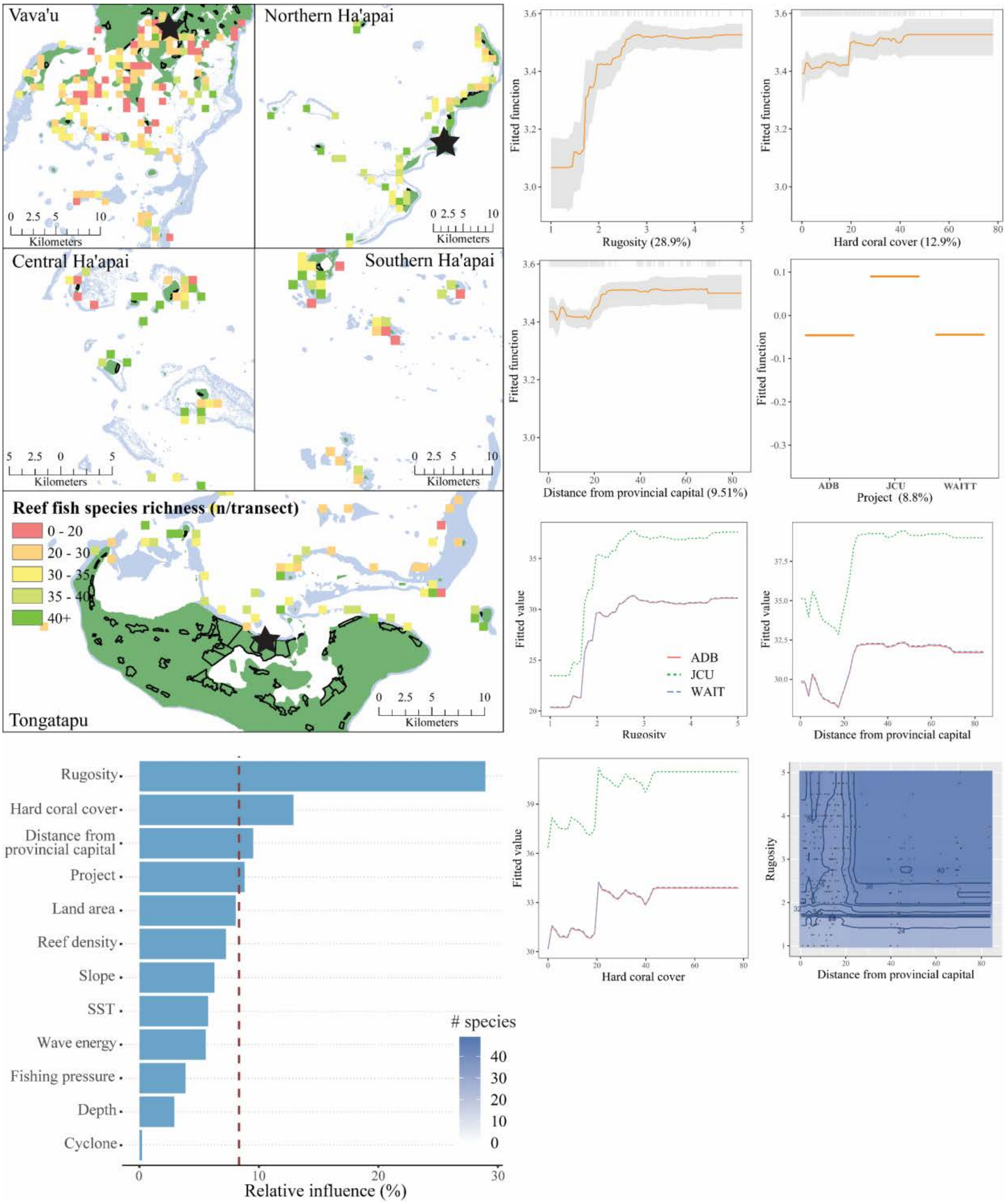


Figure 4.9. Reef fish species richness. Top left: Map of reef fish species richness at sites sampled across Tonga. Light blue represents reef, green land and black outlines villages. Each provincial capital is marked by a black star. **Bottom left:** Relative influence of the 12 predictor variables included in the Boosted Regression Tree (BRT). **Top right:** Partial dependency plots with 95% confidence intervals for the most influencial variables predicting reef fish species richness. The plot shows the effect of each predictor on the repsonse while all other variables are at their mean. Relative influence of each predictor is reported in parentheses. Grey tick marks across the top of each plot indicate observed data points. **Bottom right:** Interactions plots of the strongest pairwise interactions between influencial variables. Contour lines indicate model predictions and points represent observed data. Units are as follows: Rugosity – 1:5, Hard coral cover - %, Distance from provincial capital – km.

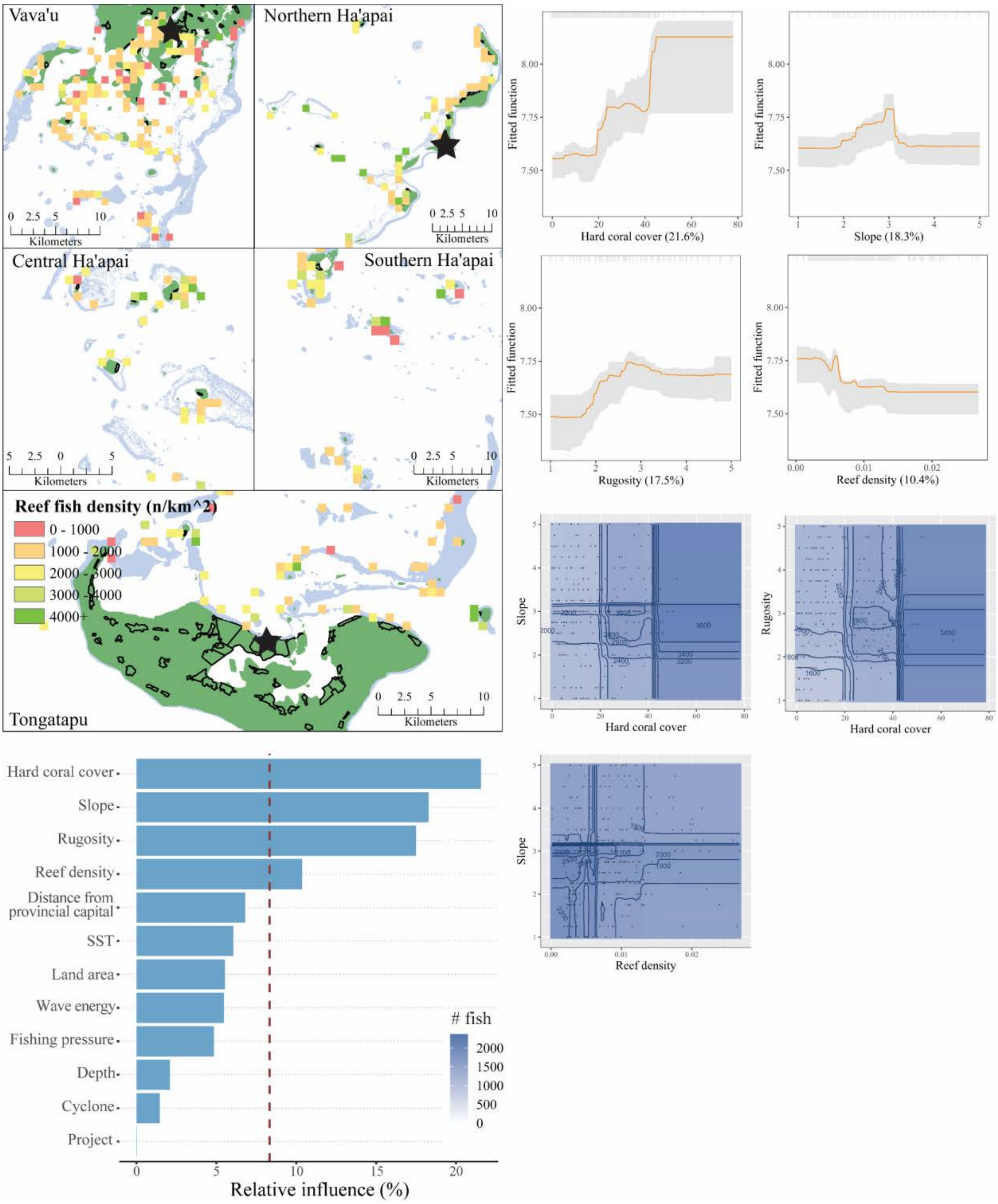


Figure 4.10. Reef fish density. Top left: Map of reef fish density at sites sampled across Tonga. Light blue represents reef, green land and black outlines villages. Each provincial capital is marked by a black star. **Bottom left:** Relative influence of the 12 predictor variables included in the Boosted Regression Tree (BRT). **Top right:** Partial dependency plots with 95% confidence intervals for the most influencial variables predicting reef fish density. The plot shows the effect of each predictor on the repsonse while all other variables are at their mean. Relative influence of each predictor is reported in parentheses. Grey tick marks across the top of each plot indicate observed data points. **Bottom right:** Interactions plots of the strongest pairwise interactions between influencial variables. Contour lines indicate model predictions and points represent observed data. Units are as follows: Hard coral cover - %, Slope – 1:5, Rugosity – 1:5, Reef density – km^2.

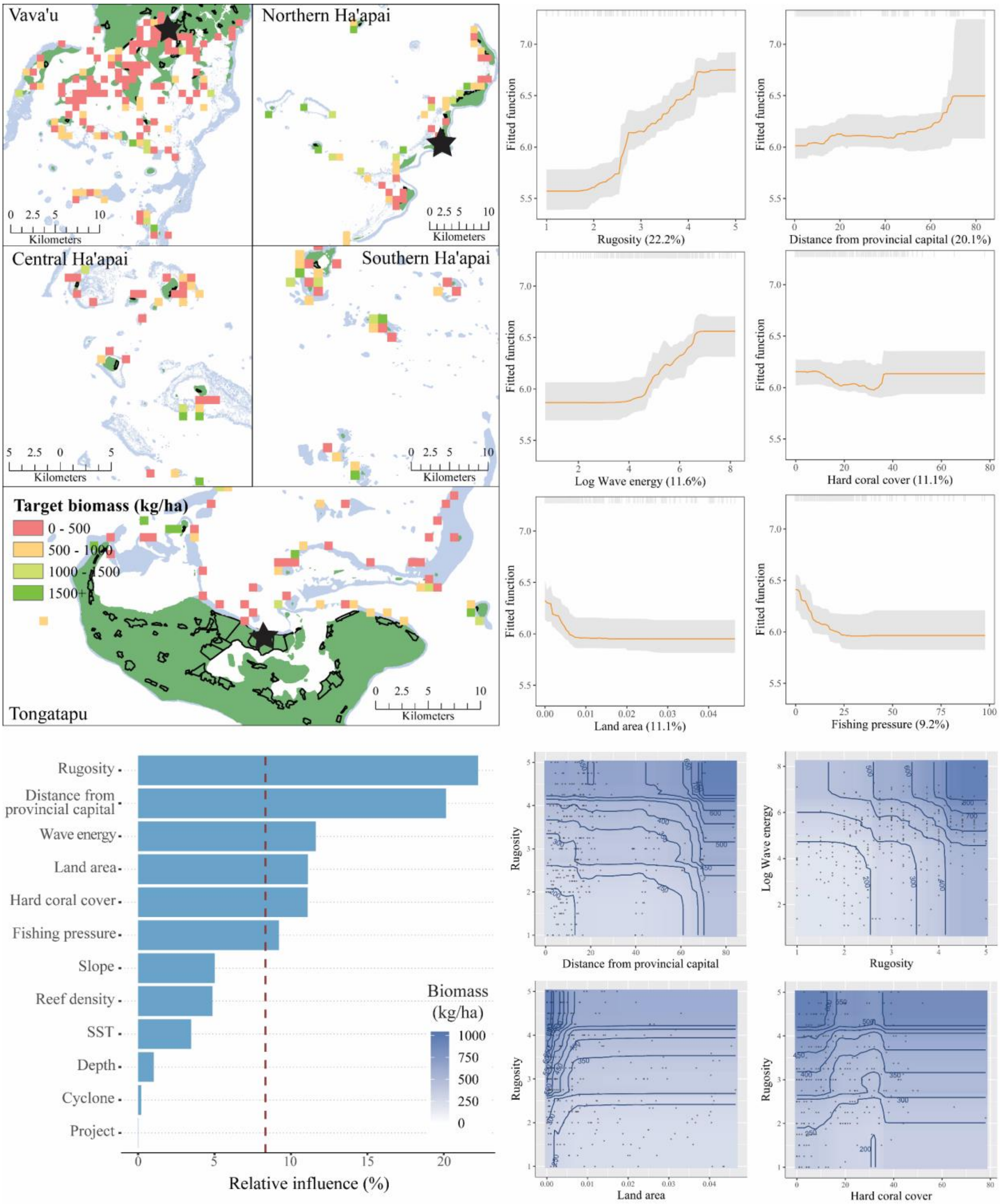


Figure 4.11. Target fish biomass. Top left: Map of target fish biomass at sites sampled across Tonga. Light blue represents reef, green land and black outlines villages. Each provincial capital is marked by a black star. **Bottom left:** Relative influence of the 12 predictor variables included in the Boosted Regression Tree (BRT). **Top right:** Partial dependency plots with 95% confidence intervals for the most influencial variables target fish biomass. The plot shows the effect of each predictor on the repsonse while all other variables are at their mean. Relative influence of each predictor is reported in parentheses. Grey tick marks across the top of each plot indicate observed data points. **Bottom right:** Interactions plots of the strongest pairwise interactions between influencial variables. Contour lines indicate model predictions and points represent observed data. Units are as follows: Rugosity – 1:5, Distance from provincial capital – km, Log wave energy – Joules per m^2, Hard coral cover - %, Land area – km^2, Fishing pressure – weighted abundance of fishers scaled to 100.

4.5 Discussion

We provide the first national description of the status of Tonga's coral reef ecosystem and reef fish fishery. Clear differences exist in the structure of coral reef ecosystems along a latitudinal gradient that corresponds to the different island groups of the country. These differences appear to be well explained by a combination of both natural biophysical variables (e.g. habitat rugosity, wave energy), and anthropogenic influences (e.g. distance from provincial capital, fishing pressure).Overall live coral cover was low (18%) but comparable to other regions throughout the Pacific affected by anthropogenic impacts (e.g. Great Barrier Reef: De'ath et al., 2012). Reef fish species richness for Tonga fell within the expected bounds for this part of the Pacific and these types of surveys (36). Overall mean reef fish biomass values suggest that Tonga's reef fish fishery can be classified as moderately to heavily exploited, with 64% of sites having less than 500 kg/ha (MacNeil et al., 2015; McClanahan, 2018; McClanahan et al., 2019;). In the sections that follow we i) discuss the major relationships between socio-environmental variables and key metrics of reef condition, including several caveats to our findings, and ii) provide further details of the observed patterns within each island group.

i) Major relationships between socio-environmental variables and reef condition

The strongest and most common two predictors of reef structure in Tonga were habitat rugosity and distance from the provincial capital. Habitat rugosity is a well-established driver of reef community structure, and is linked to both natural processes and anthropogenic disturbances (Harbourne et al. 2012; Graham and Nash 2013). Given its near ubiquitous effect, our findings also suggest that Tonga's human influence is having a clear and strong impact on the structure of their reefs. The authors are unaware of any natural biophysical variables that correlate strongly with distance from each island groups main city. However, distance from human population centres is not in itself a disturbance, but only a proxy for the many types of influence that humans may cause. While the clearest is likely to be fishing pressure (in addition to variability not accounted for in the fishing pressure metric), other disturbances may include pollution or development. Importantly, other metrics related to human influence have also been developed, most notable the Human Gravity metric developed by Cinner et al. (2018), which may be better at predicting patterns of human impact on reef structure. However, for the current analysis these metrics were of too coarse a resolution to be employed.

The negative effects of increased SST on scleractinian corals is well documented (Hughes et al., 2017, 2019). However, coral bleaching is primarily associated with heat stress events, including high variability in SST, or sustained temperatures above the thermal tolerance of coral species (e.g. degree heating weeks [DHW]). Due to the coarse resolution (5 km) of the National Oceanic and Atmospheric

Association (NOAA) Coral Reef Watch (CRW) layers for SST variability and heat stress events when compared to the resolution of this analysis, we were unable to include SST variability or DHW in the set of predictor variables (supplementary materials). Despite this caveat, many reefs in northern Ha'apai and Vava'u did display signs of recent bleaching events. In several instances bleaching of entire reefs had clearly occurred within the past five years, indicated by retained complexity of dead corals. The two-degree difference in mean SST between Tongatapu and Vava'u provides a potential mechanism by which coral cover in Vava'u is lower, as corals in this island group could be living closer to their thermal bleaching threshold. However, acute thermal anomalies have a greater influence on coral assemblages than long-term means and coral bleaching thresholds are relative to local thermal histories (Hughes et al. 2017). Therefore, the negative relationship between SST and live coral cover may also be related to differential acute heat exposure among locations due to oceanographic factors or local weather conditions mediating heat stress, or to additional factors that are collinear with SST. For example, the sheltered geography of Vava'u may limit flushing by cooler, oceanic waters, so that when extreme temperature events do occur, they may be more pronounced in duration and extent. It is also possible that local weather conditions (e.g. wind, cloud cover and rain) led to lower thermal stress in Tongatapu during periods of thermal stress, as was the case on the Great Barrier Reef in 2016 (Hughes et al. 2017). Given the lack of previous data, it is not possible to accurately determine the frequency or severity of mass bleaching events in Tonga, and further research is necessary to separate the effects of large-scale versus local oceanic conditions.

Patterns of reef fish diversity and density in Tonga appear to be associated primarily with natural biophysical variables, such as reef complexity (i.e. rugosity), slope and hard coral cover. The importance of both structural complexity and live coral cover as influences on coral reef fish communities is well recognized (e.g. Harbourne et al,. 2012; Graham and Nash, 2013) and, although often collinear, it is nonetheless important to distinguish the two. While scleractinian corals are the dominant habitat-forming organisms on reefs, larger-scale structural complexity can be more strongly associated with long-term patterns of reef accretion than the immediate presence of live coral, as well as habitat-forming organisms in addition to corals (Dustan et al., 2013). While project was also an important driver at predicting reef fish species richness, one of the benefits of BRT analysis and partial dependency plots is the ability to account for issues such as these. The relationship between all other predictor variables and reef fish species richness were therefore considered within this context and examined while controlling for variation in project methodologies.

Based on previous calculations of global baselines and benchmarks for fish biomass (MacNeil et al., 2015; McClanahan, 2018; McClanahan et al., 2019;), these results suggest that Tonga's reef fish fishery can be classified as heavily exploited in Vava'u, moderately exploited in Tongatapu and central Ha'apai, and with lower levels of exploitation in southern and northern Ha'apai. Target species biomass responded strongly to a combination of biophysical and local anthropogenic variables. While there was a sharp decline in biomass with increasing fishing pressure at low levels, the effects plateaued at higher levels of fishing. This also corresponds to the greatest increase in biomass at sites furthest from the provincial capitals. Together these findings support the hypothesis laid out previously from studies across large gradients of human population and fishing density that the highest absolute losses in reef fish can occur with relatively low fishing pressure (Dulvy *et al.* 2004; Bellwood *et al.* 2011; McClanahan et al., 2019). However, differences in population density and fishing pressure do not explain why target species biomass in Vava'u was lower than Tongatapu, where human impacts are greatest. Instead, given the importance of both biophysical and anthropogenic variables at predicting biomass in our data, similarities in biomass values between Tongatapu and Vava'u might be best explained by overfishing in Tongatapu and by poor quality habitat (e.g. low rugosity, coral cover and wave energy) in Vava'u.

ii) Considerations within each island group

Vava'u

The most prominent findings from these data are the poor condition of Vava'u across all metrics of reef condition. Coral cover in Vava'u was exceptionally low (mean 10.4%), but even this was buffered by several lagoonal sites with high percent cover of *Porites rus* and *Porites cylindrica*. With these sites removed, mean coral cover for the rest of Vava'u was closer to 5%. Likewise, richness, density and biomass estimates for Vava'u were all lower than other island groups. The geography of Vava'u is unique and very different to reefs in Ha'apai and Tongatapu that are more typical of Pacific reefs. The reefs of Vava'u are generally sheltered, narrow fringing reefs below limestone cliffs and adjacent to deep (60-100 m) water. Most reefs are likely to have very little current flow and are sheltered from the open ocean and prevailing weather conditions. Reefs in Vava'u might therefore be more susceptible to impacts from both coral bleaching and local pollution. When reefs are subjected to heatwaves, coral bleaching in more open areas could be limited by flushing from cool oceanic waters, while the geography of Vava'u would limit flushing and result in pockets of warm water persisting for much longer. Likewise, pollutants from local sources are unlikely to wash away given the topography of the islands and instead might persist at greater concentrations. However, limited data are available on current regimes around Vava'u to investigate this hypothesis. A further consideration is that there was extensive historical dynamite fishing in Vava'u, more so than in Tongatapu. While this practice has long been regulated against, it was still practiced up to the early 1990s, and may have driven a regime shift away from hard corals, with little recovery to date (personal communication).

Ha'apai

While reefs in southern Ha'apai were generally in the best condition of those assessed, there was also extensive evidence of recent bleaching along the western edge of the northern Ha'apai ribbon reefs. As with Vava'u, part of this problem could be associated with prevailing wind, wave and current conditions, which generally move from east to west. Many of the sites in northern Ha'apai are sheltered from the east by the main islands and therefore could also trap pockets of warm water, exacerbating bleaching at a local scale. Conversely, the reefs of Southern Ha'apai are much more exposed, which might therefore promote flushing by prevailing winds, waves and currents.

An additional point worth noting is that there are increasing numbers of fishers from Tongatapu travelling to Southern Ha'apai to fish, as well as fishers from Ha'apai transporting their catch to Faua Wharf in Tongatapu. Both of these factors could potentially confound the influence of local fishing pressure and distance from the provincial capital on target biomass. However, given the strength of these variables, it is still clear that within island group fishing activities are still strong predictors of fish biomass.

Tongatapu

The coral reefs around Tongatapu experience the greatest human pressures in Tonga, with 70% of the country's population on this island. The Fanga'uta lagoon is highly polluted (Aholahi et al. 2017) and flows directly onto reefs in the Tonga channel. Likewise, the number of fishers in Tongatapu is equal to that in Ha'apai and Vava'u combined (Statistics Department of Tonga, 2017). Despite this, the reefs in Tongatapu were overall in better condition than anticipated. Coral cover within the main bay was higher than elsewhere in the country and reef fish richness and density were moderate. These results could be due to the cooler waters in Tongatapu, which might buffer against the large bleaching events which appear to have impacted Vava'u and northern Ha'apai. Only target biomass was low, which is expected, given the clear relationship between human influence and biomass (Cinner et al., 2018). As explained previously, the similarities in fish biomass between Vava'u and Tongatapu might therefore be best explained by overfishing in Tongatapu and reef condition in Vava'u.

iii) Further considerations

Several sites of note were also identified with very poor coral cover (0%) in both Tongatapu and Vava'u, which warrant further investigation. In Tongatapu these were near the mouth of the Fanga'uta lagoon and in Vava'u in many of the inner island areas, particularly near the causeways. At these sites there were often no living corals and instead high densities of *Diadema sp.* sea urchins.

These appeared to be scraping away the reef matrix at a large scale and inadvertently destroying any recruiting corals. It is possible that pollution from Tonga's lagoonal areas might be causing outbreaks of *Diadema sp.* sea urchins. We recommend this possibility as a critical area for further investigation in Tonga.

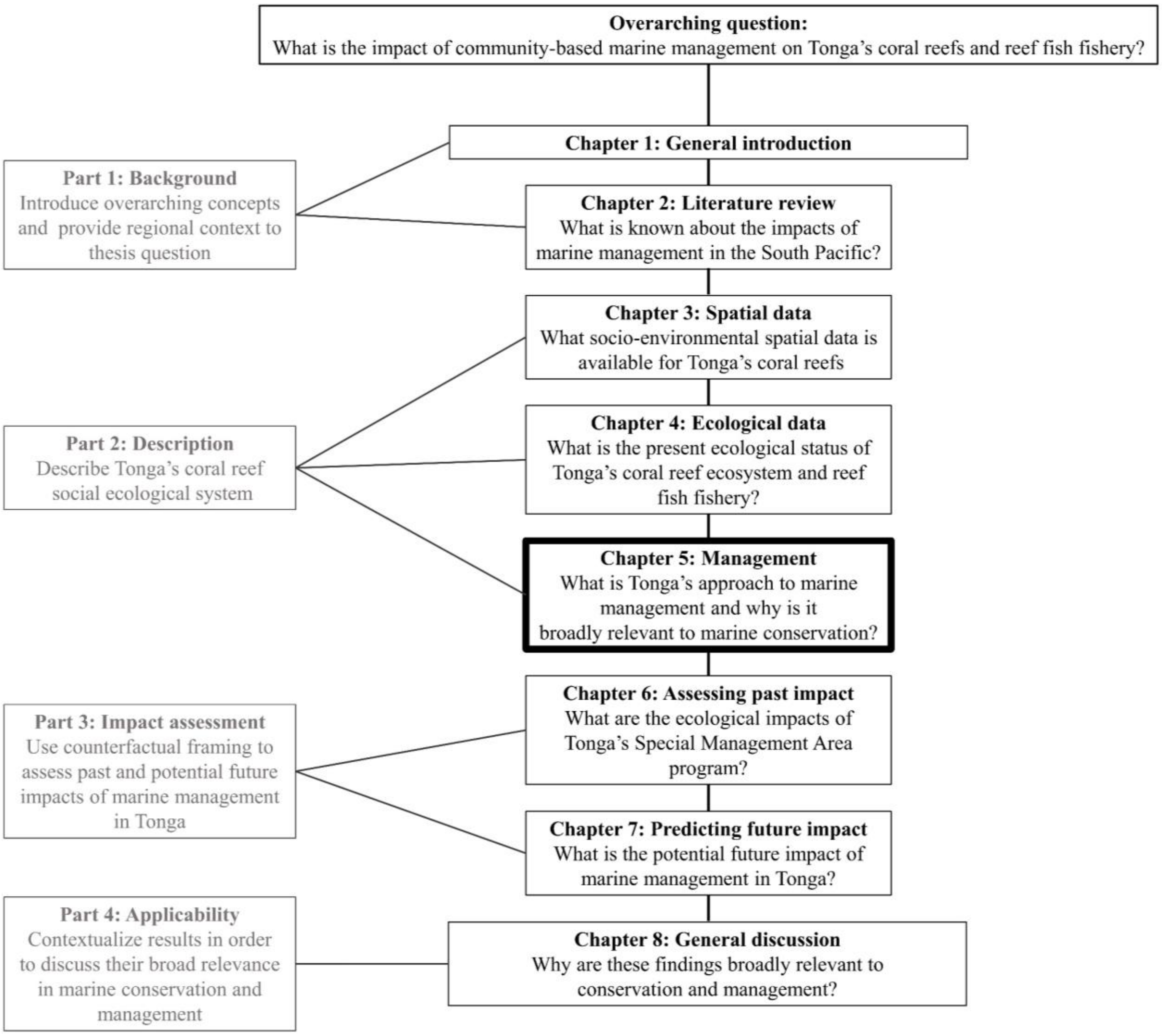
While fisheries management in Tonga has been historically open access, in recent years this has changed with the implementation of the Special Management Area (SMA) program. This locally driven initiative has now grown to include over 50 communities that each have at least one no-take marine protected area as well as an exclusive access zone (Smallhorn-West et al. 2020a). Previous studies using statistical matching have demonstrated positive impacts for reef fish biomass, density and species richness for the seven oldest community-based no-take zones in Tonga (Smallhorn-West et al. in press). While the present study did not test management status per se, differences in fishing pressure due to management were included within the fishing pressure metric, and therefore the positive impacts of management were incorporated into the analysis.

iv) Conclusions

Our data and analysis deliver critical baseline ecological information for Tonga's coral reefs that will both aid ongoing management and research and enable accurate reporting to local and international agencies. For example, future reports on Tonga's coral reefs need no longer classify them as 'data deficient', and some degree of accountability should now be expected from governments regarding effective policies and management. There is also now a great deal more information available for use in the SMA program, with data from these projects already being used to examine the impact of existing and potential new configurations of no-take reserves (Smallhorn-West et al. 2020a, Smallhorn-West et al. in press). This data has also been compiled into a large national report, available in both English and Tongan, to increase public awareness about reef status and the effects of management (Smallhorn-West et al. 2020b). Lastly, we anticipate that this extensive data set can be used as a benchmark for ongoing monitoring and future impact evaluation studies in Tonga.

Chapter 5: Incentivizing community management for impact: mechanisms driving the successful national expansion of Tonga's Special Management Area program

P. Smallhorn-West, J. Sheehan, S. Malimali, T. Halafihi, T. Bridge, R. Pressey, G. Jones



5.1 Abstract

The expansion of coastal marine protected areas can suffer from two key drawbacks: i) the difficulty of incentivizing local communities to also manage areas for conservation when their livelihoods also depend on resource use; and ii) that many protected areas get situated residually, or in areas with limited value for either biodiversity conservation or livelihoods. Here we discuss and analyse key characteristics of Tonga's Special Management Area (SMA) program, including both the mechanisms that have motivated its successful national expansion and its ability to configure no-take reserves in areas that are considered to have high value to resource users. Granting communities exclusive access zones in exchange for implementing no-take reserves has encouraged conservation actions while fostering long-term relationships with resources. Ensuring no-take reserves occurred within the boundaries of exclusive access zones enticed communities to protect areas of greater extractive values than they would have otherwise. We conclude that the success of this program offers a way forward in achieving targets in the global expansion marine protected areas.

5.2 Introduction

Food security and biodiversity are increasingly threatened by the depletion or collapse of marine resources (Diaz et al. 2019), and many proposed management strategies fail at scaling up to achieve meaningful national or international conservation results (Mills et al., 2019). Marine resources are also notorious for suffering from the “tragedy of the commons” (Hardin, 1968), whereby individuals or groups of individuals overexploit a resource and behave contrary to the common good of all users (Ostrom, 1999). While many marine management strategies have been implemented, each comes with its own suite of caveats (Ban et al., 2011; Jupiter et al. 2014). A key goal of conservation policy and management research is to identify solutions to the specific issues that limit the effectiveness of various management strategies (Diaz et al. 2019).

Protected areas are expanding globally as a key management strategy to address both declining food security and biodiversity (Diaz et al. 2019; Mills et al., 2019). While their management often takes the form of centralized governments gazetting areas for conservation, in practice, they are often compromised by a lack of resources for monitoring and enforcement (Gaymer et al., 2014). In response to continuing fisheries declines despite centralized management, many developing countries are increasingly focusing on decentralized, community-based or co-management approaches, which in many instances were already in place through customary marine tenure (Govan et al., 2009; Webster et al., 2017; Cinner et al. 2012). Here, we consider community-based management to be natural resource or biodiversity management by, for and with the local community (as defined by Western and Wright, 1994) and co-management as situations where communities share responsibilities for making and enforcing natural resource management rules with governments, civil society, and/or academia (Cinner and Huchery, 2013).

Implementing protected areas is often met with resistance unless local communities can be offered incentives to manage areas for conservation when their livelihoods depend on the resources within them (Brockington & Schmidt-Soltau, 2017; Ferraro & Hanauer, 2011). Typically, managers and conservationists argue that the long-term food security of an area and its biodiversity value outweigh immediate requirements for continued resource use (Hutton & Leader-Williams, 2014). However, offering long-term assurances of increased food security and ecosystem health might not always be important for people for whom finding food or making a living are immediate concerns (Hutton & Leader-Williams, 2014). The strategy of excluding resource extraction has attracted criticism from social scientists and human rights advocates for resulting in the forced displacement of populations and loss of food security (Cernea & Schmidt-Soltau, 2006). While compensatory incentive-based programs do exist, such as direct payment concessions for protected areas, they likely provide limited benefits to biodiversity conservation unless they are conditional on defined conservation actions (Mills et al., 2019; Sachedina & Nelson, 2009).

A second problem with the global expansion of protected areas is that many are residual, or situated in areas with limited value for extractive activities, and have correspondingly small conservation impact (Devillers et al., 2015; Ferraro & Hanauer, 2011; Joppa & Pfaff, 2011). Ultimately, protected areas are effective only if they change human behaviour (Pressey, Weeks, & Gurney, 2017). To achieve impact they must therefore be configured to influence either present day or potential future actions (Smallhorn-West, Bridge, Malimali, Pressey, & Jones, 2018). However, given the importance of involving local stakeholders in the planning process (Hutton & Leader-williams, 2014), it seems inevitable that resource users will aim to configure protected areas to minimize overlap with their current or planned activities.

The responsibility of identifying solutions to incentivize protected areas implementation and ensure they are situated to achieve impact should lie with planners as well as conservation policy and management researchers. Individual communities may have little choice but to prioritize their immediate needs for food and/or income. The question raised is therefore: *Is it possible to align the short-term requirements of communities with the goal of building sustainable use and biodiversity conservation into the future?*

Here, we address this question by discussing the recent rapid expansion and successful implementation of Tonga's co-management initiative, the Special Management Areas (SMA) program, at a national level. We use this program as a case study to identify solutions to the aforementioned problems of providing short-term incentives for long-term conservation, and ensuring conservation actions are non-residual. In a relatively short time (15 years), Tonga's SMA program has expanded from a few communities to over 40, covering roughly half of all coastal communities in the country and aiming to include 100% by 2025. Furthermore, SMAs are situated in places that are considered to have high value to resource users. We argue that, by providing the right balance of incentives, the SMA program has successfully avoided key pitfalls associated with protected area implementation, which has enabled the program to expand to a national level in a way which is non-residual. Specifically, we: i) describe the background and key characteristics of the program; ii) identify mechanisms by which the program has avoided problems that have constrained the effectiveness of other protected areas, including a) provisioning of appropriate incentives and b) avoiding conserving only residual areas; and iii) discuss potential limitations of the program and its expansion to other regions. We conclude that the success of this program offers insights into the successful expansion of protected areas globally.

5.3 Background of Tonga's Special Management Area (SMA) program

Fisheries management in Tonga was historically open access, with little to no effective regulations. A civil war in the mid-1800s resulted in the then king, Taufa'ahau Tupou I, abolishing all tenure – a key difference between Tonga and many other Pacific island nations where customary marine tenure is in place. The King also proclaimed that: i) all Tongans had equal fishing access to all Tongan waters; and ii) that any traditional claims of local control or management authority over fishing areas were abolished (Gillett, 2017). In modern times this open access approach has collided with commercial realities and the inability of inshore resources to sustain harvests (Gillett, 2017). Due to growing concern over the potential depletion or collapse of marine resources, several forms of centralized management and protected areas were attempted in the late 20th century (Smallhorn-West & Govan, 2018). However, due to the limited capacity of the Tongan Ministry of Fisheries (MoF), the main government agency charged with monitoring and enforcement, there is no evidence that resource extraction within these managed areas ever changed.

Figure 5.1. Map of a typical Special Management Area (SMA) in Tonga. The yellow denotes the SMA area, in which only members of the community are allowed to fish, similar to a territorial user rights fishery (TURF). The red denotes the FHRs, which are permanently closed to all fishing. Given that this SMA included both exposed reef and sheltered fringing reefs, this particular community (Ha'atafu) opted to implement two no-take FHRs instead of one. Figure credit: Jason Sheehan.

In the early 2000s, growing support for the concept of letting local communities manage their own resources resulted in the Fisheries Management Act 2002 (Gillett, 2017). Funding was provided by Australia to support the Tonga fisheries project and assist in the establishment of the early SMAs. The first, O'ua in the Ha'apai group, was designated in November 2006. While the program has since received funding from many sources (Gillett, 2017), it has largely been the Tongan MoF that has driven its expansion. Tongans are therefore justifiably proud in the fact that the successful implementation of this "home grown" program has largely been due to the efforts of Tongans.

The SMA program is a dual approach to marine management and conservation (Figure 5.1). First, through legislative action, each community is granted exclusive access to the marine environment adjacent to their village to the 50 m depth contour or 2500 m from shore (Figure 5.2). Within this area only registered members of the community are permitted to fish and it effectively acts as a territorial user rights fisheries (TURF) (Gelcich et al. 2008). Second, in exchange for this exclusive access, a subset of the area must be designated a permanent no-take zone, termed a Fish Habitat Reserve (FHR). The size and location of each FHR is determined by the community and, if desired, communities may implement multiple FHRs. The size and boundaries of each SMA are determined by the MoF in consultation with both the SMA communities and adjacent communities. Within each SMA, management and enforcement are the responsibility of the community, and each must establish a coastal community management committee and a coastal community management plan. Communities therefore take the leading role in managing their coastal resources, although assistance is provided by the Ministry as required.

Tonga's SMA program has become so popular with Tongan communities that there is more interest from communities than the capacity of the MoF can currently manage (Gillett, 2017). During the decade following the implementation of the first SMAs (2006-2015), the program grew slowly, with 11 SMAs in place (Fig. 2) (Table S1). The slow uptake was largely due to the lengthy process of raising awareness and educating communities and the public about the benefits of marine management. However, as awareness grew, interest in the program expanded exponentially. From 2016 to 2019, 31 new SMAs were established, resulting in roughly half of all coastal communities in Tonga having an SMA. This rapid uptake following 2016 was likely due in part to i) increased awareness from a "lessons learnt" conference in October 2015 implemented by the MoF and Civil Society Forum of Tonga and supported by the Marine and Coastal Biodiversity Management in Pacific Island Countries (MACBIO) project (Tupou Taufa et al. 2016), and, ii) increased financial support from various international donors to implement new SMAs in the Vava'u archipelago (e.g. Asian Development Bank and WAITT Institute). As of September 2019, an additional 46 SMA communities have either been confirmed, submitted to cabinet for approval, written a letter of interest, or been proposed, with the aim of including all coastal communities in the program by 2025 (Table S2).

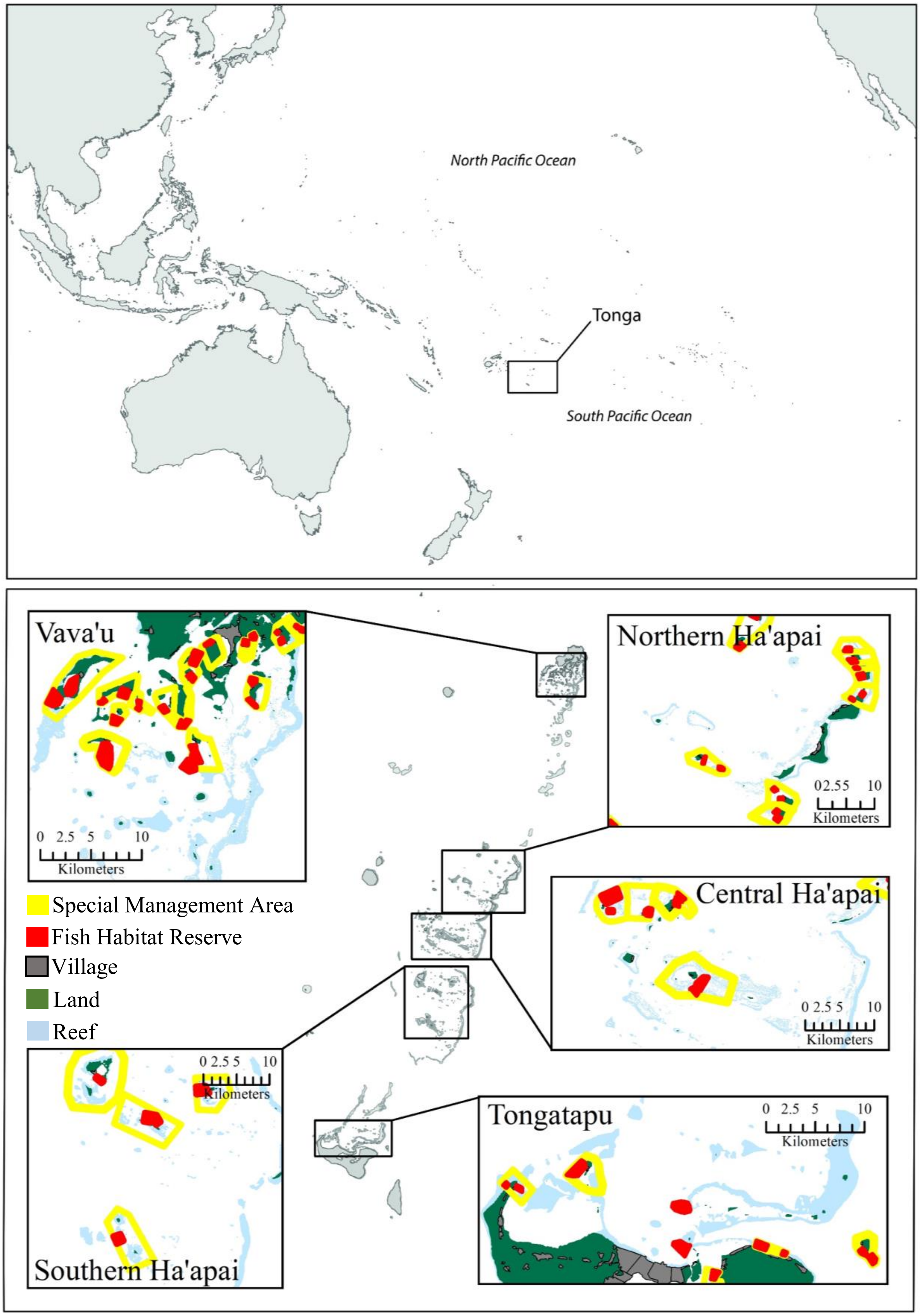


Figure 5.2. Overview of Tonga's Special Management Area program as of October 2019. Yellow denotes Special Management Areas (SMAs), red no-take Fish Habitat Reserves (FHRs) and grey with black outlines communities.

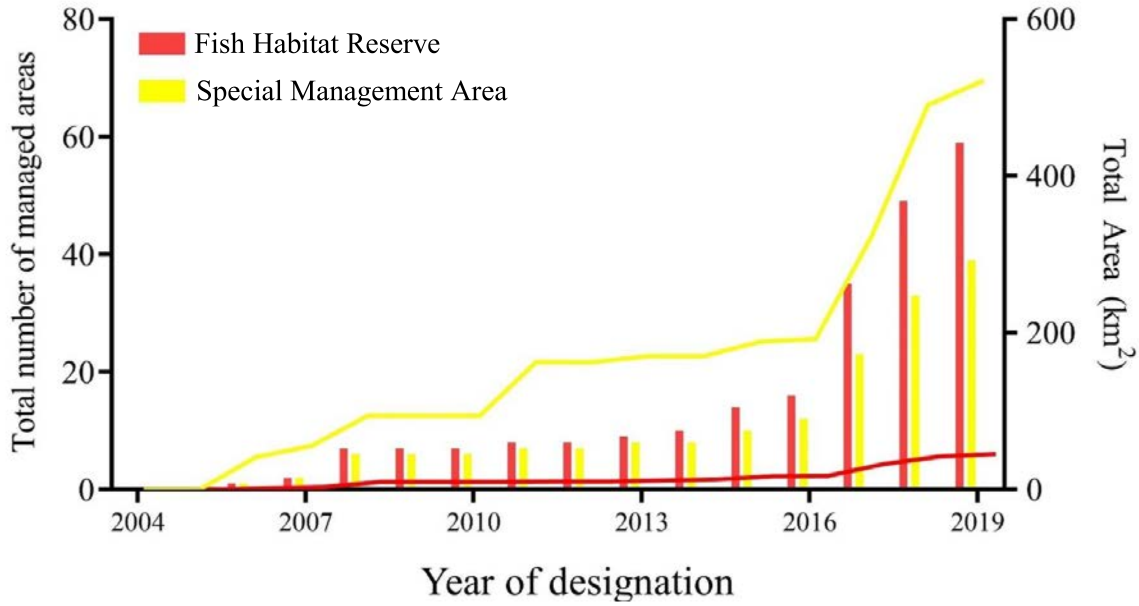


Figure 5.3. Growth of Tonga's Special Management Area (SMA) program, with bars indicating the total numbers of SMAs and Fish Habitat Reserves and lines representing the total areas.

5.4 Mechanisms by which the SMA program has avoided pitfalls common in the expansion of protected areas elsewhere

Providing short-term incentives for long-term conservation

The primary consideration of most communities for implementing an SMA is to exclude others from fishing "their" reefs (Figure 5.4a). Exclusive access rights are a substantial asset for any community, and it is inherently in the interest of each community to establish an SMA. However, given that in exchange for exclusive rights communities must also establish a no-take FHR, the SMA provides the incentive to achieve meaningful conservation results through the FHR. Considering the popularity of the program, the SMA incentive clearly provides ample compensation to communities for giving up the fishing grounds within the FHR.

Another mechanism by which short-term incentives have driven the expansion of the SMA program is through a positive feedback loop that increases pressure for remaining non-SMA communities to apply (Figure 5.4b). While SMA communities can fish both inside and outside their SMAs, non-SMA communities are blocked from fishing inside nearby SMAs. At the program's inception, when only a small number of SMAs were in place, this was not of huge consequence to non-SMA communities. However, as the program has expanded, each additional SMA implemented further reduces the fishing grounds for non-SMA communities while leaving their coastal areas vulnerable to fishing by all other communities.

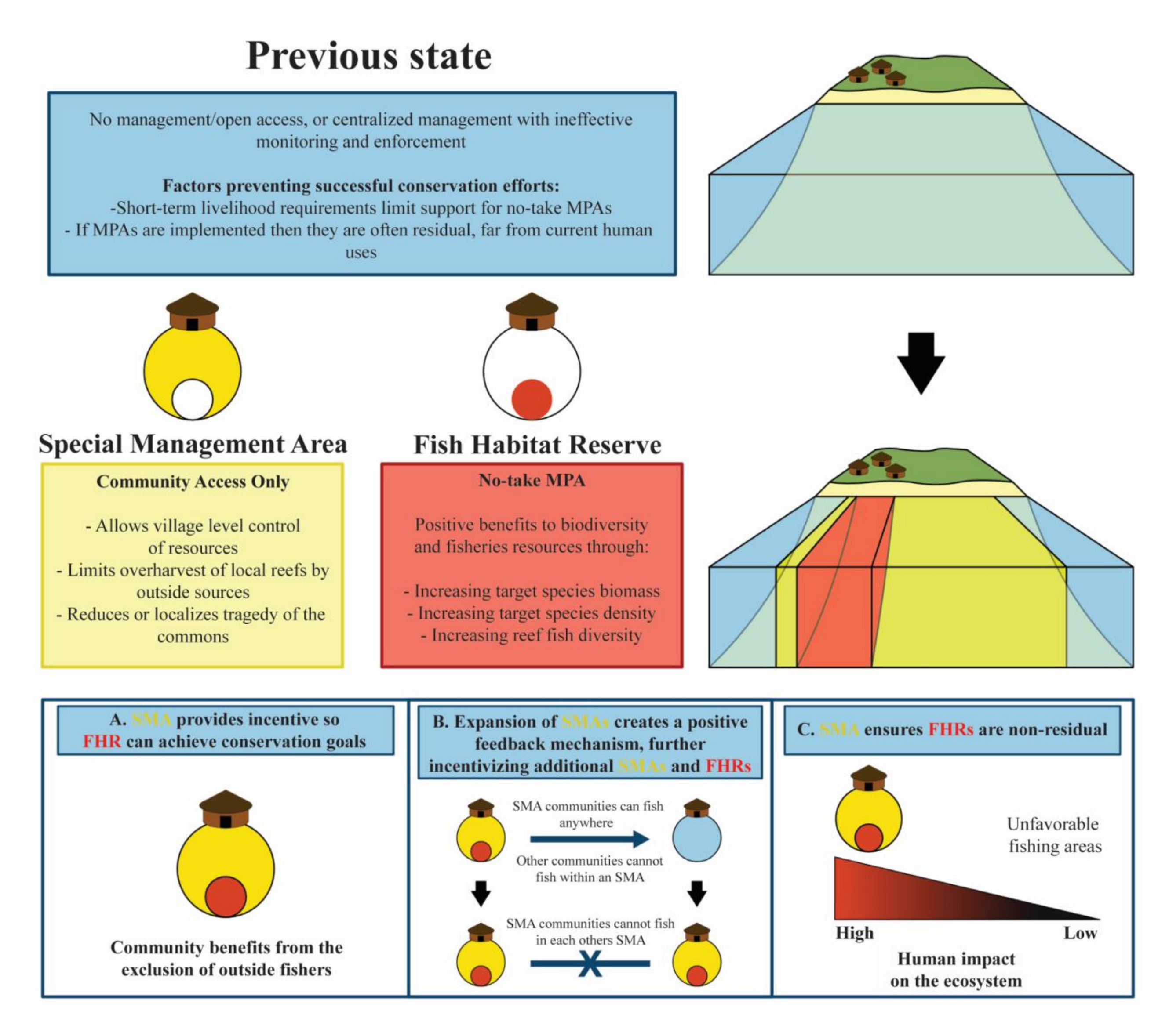


Figure 5.4. Conceptualization of Tonga's Special Management Area program. The top row represents the state prior to implementation of the program including problems with open access systems and factors preventing successful conservation efforts. The middle row represents the SMA program, with the expected outcomes of the SMAs and FHRs. The bottom three boxes represent key mechanisms by which the SMA program has avoided problems that have constrained the effectiveness of other protected areas. Figure credit Jason Sheehan.

As with TURF systems established elsewhere (Villasenor-Derbez et al. 2019), many communities have also developed a sense of pride and ownership over their SMAs and FHRs, and encouraged a sense of belonging and the development of long-term connections with 'their' reefs. The long-term success and failure of SMAs now largely depends on community-level actions, and this has created a sense of competitiveness by which communities are eager to demonstrate their SMAs' success.

Avoiding residual conservation

Residual conservation is now a well-recognised concern with protected areas globally (Devillers et al., 2015; Ferraro & Hanauer, 2011). We tested for the presence of systematic biases in the placement of SMAs and FHRs, compared to open areas, by assessing whether they were located in regions with low value to resource users across four metrics known to influence the configuration of protected areas.

The primary resource associated with Tonga's SMA program is the reef fish fishery (Parks, 2017). We therefore converted all reef area in Tonga into 100 m^2 raster cells in ArcMap (10.4.1) and labelled these as either SMA, FHR or Open based on their configuration as of October 2019. Four socio-environmental variables were selected to test for systematic biases in the placement of SMAs and FHRs: distance to village, distance to land, fishing pressure and wave energy. Fishing pressure inside management areas represents fishing pressure prior to management. These four variables, which were previously calculated for the entirety of Tonga's coral reef habitat (Chapter 3), were chosen because they are: i) known to influence the configuration of protected areas; and ii) are based on spatially continuous data across the region. For the whole of Tonga, null models were created of equal area to both total area of SMAs and total area of FHRs, but randomly sampled from the total reef area in Tonga (including SMAs and FHRs). These two null models were resampled 1000 times and the difference in all four metrics calculated between the actual SMA or FHR extent and each null model. In addition, to determine whether FHRs were systematically biased *within* SMAs, the same method was applied but only to the total combined area of FHRs and SMAs. One sample t-tests were then used to determine whether the bootstrapped differences varied significantly from 0. All analysis was conducted in R (V.3.5.3) (R core team, 2017).

With the exception of fishing pressure inside the SMAs, both FHRs and SMAs were biased towards areas of greater extractive value than expected by chance (Table 5.1, Figure 5.5). Distance to village, distance to land, and wave energy were all significantly lower within FHRs and SMAs than null models. Fishing pressure was greater within FHRs, but lower in SMAs. In addition, within SMAs, FHRs were also more likely to be configured in areas of higher fishing pressure, lower wave energy, and closer to villages and land than the null model.

These results demonstrate that Tonga's SMA program does have systematic biases in its configuration, but in the opposite direction to those commonly observed for protected areas. Rather than being residual, management areas in Tonga are systematically less likely to be placed in areas of low extractive value than by chance. The SMA program has therefore been able to avoid residual biases in protected area placement because no-take FHRs must be situated only within the boundaries of each SMA, and SMAs are implemented only near villages, where resource use is historically high (Figure 5.4c).

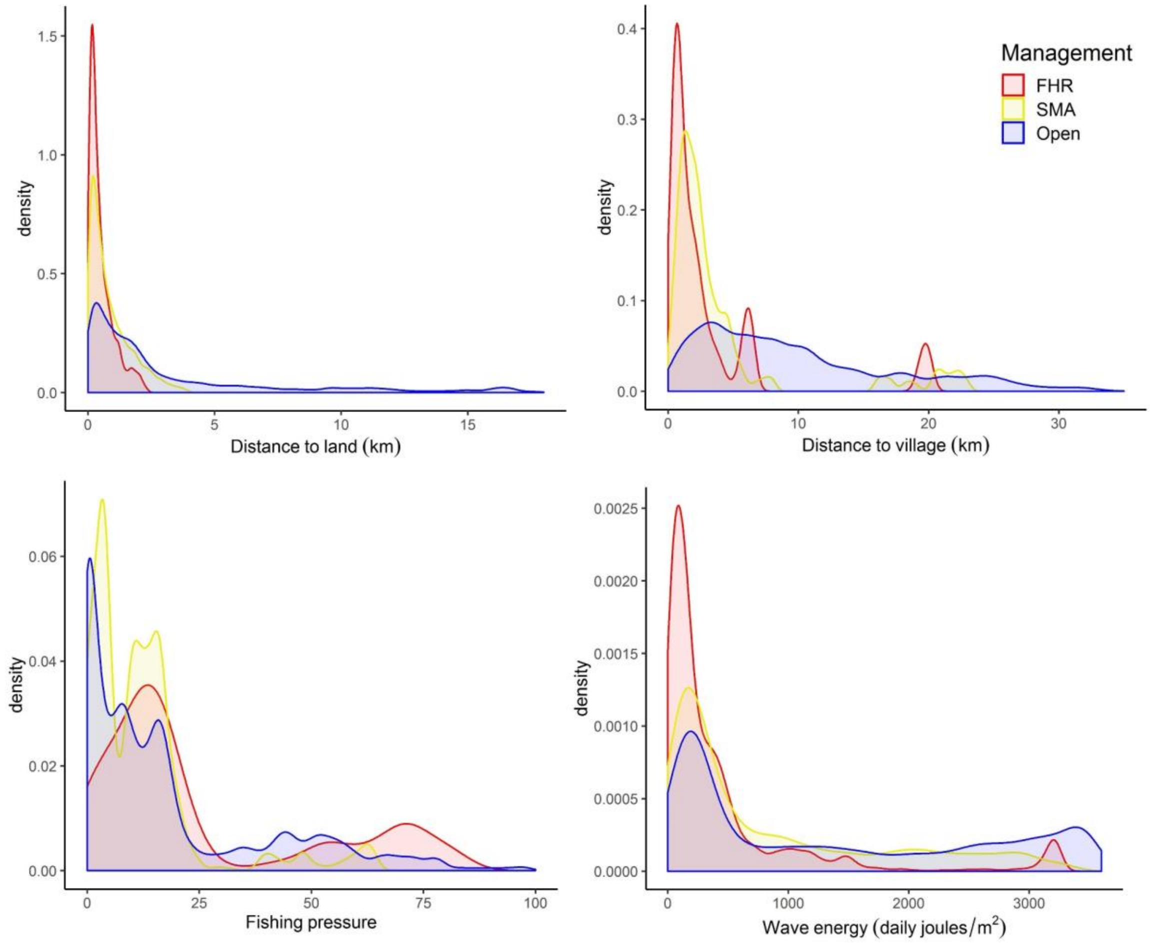


Figure 5.5. Density plots of overall differences in four socio-environmental variables between FHR, SMA and open areas in Tonga.

Table 5.1. The effects of four socio-environmental metrics on the presence of Fish Habitat Reserves and Special Management Areas in Tonga. Lower confidence limit (LCL) and upper confidence limit (UCL) represent 95% lower and upper confidence limits respectively. Negative estimate values indicate that values inside the management areas are lower than in areas open to fishing and positive estimate values indicate that values inside the management areas are greater than in areas open to fishing.

Variable	Estimate	LCL	UCL	t	df	p value
Fish Habitat Reserves						
Distance to land	-2048.59	-2055.93	-2041.25	-547.8	999	< 0.05
Distance to village	-5353.65	-5366.11	-5341.2	-843.36	999	< 0.05
Fishing pressure	6.59	6.55	6.62	352.31	999	< 0.05
Wave Energy	-855.94	-858.05	-853.84	-798.6	999	< 0.05
Special Management Areas						
Distance to land	-1627.38	-1630.25	-1624.51	-1111.6	999	< 0.05
Distance to village	-4242.97	-4247.68	-4238.26	-1767.7	999	< 0.05
Fishing pressure	-5.03	-5.05	-5.02	-809.19	999	< 0.05
Wave Energy	-367.79	-368.52	-367.06	-988.86	999	< 0.05
Fish Habitat Reserves within Special Management Areas						
Distance to land	-365.01	-370.27	-359.75	-136.16	999	< 0.05
Distance to village	-963.84	-972.53	-955.15	-217.67	999	< 0.05
Fishing pressure	11.07	11.04	11.09	846.45	999	< 0.05
Wave Energy	-339.33	-340.71	-337.96	-484.21	999	< 0.05

5.5 Limitations of the program and its expansion to other regions

At the outset, while it is clear that the implementation of Tonga's SMA program has been successful with respect to its rate of expansion, this does not demonstrate any difference made to the stated objectives of improving coastal fisheries resources or biodiversity conservation. Ultimately the success or failure of the SMA program is based on its impact, or the difference it makes compared to taking no action. However, determining impact relies on having an accurate understanding not only of the present state, but also counterfactual conditions that would be expected if management had never occurred (Pressey, Visconti, & Ferraro, 2015). While most SMA communities are enthusiastic about the benefits of the program, there is little quantitative evidence of any changes in ecosystem state and, ultimately, coastal fisheries resources. Several studies conducted in 2010 on five SMAs began to examine the impacts of the SMA program, with basic control-impact methodology (Malimali, 2013; Richardson, 2010). However, they were completed when most SMAs were still too young for discernible changes to have occurred. Webster et al. (2017) compared community-based catch data with community perceptions of change in the oldest SMA in Tonga, although their methodology did not test the impacts of the FHR and used data of questionable quality. While a large body of evidence supports the notion that the no-take FHRs should provide positive impacts (Smallhorn-West et al., 2018), given that fishing is still allowed inside the SMAs, albeit potentially at a lower rate, it is unreasonable to expect large changes in ecosystem state within SMAs.

While acknowledging the caveats associated with protected area targets (Pressey et al., 2017), it should also be noted that the present spatial coverage of no-take FHRs in Tonga is low and unlikely to make significant contributions to national protected area commitments. Currently total FHR coverage is 45 km^2, or 6.82×10^{-5} % of Tonga's EEZ, and 3.26 % of Tonga's coral reef habitat. Furthermore, given widely reported problems with misreporting protected area targets in the South Pacific (e.g. Smallhorn-West & Govan, 2018), SMAs could easily be mislabelled as no-take protected areas and give the false impression that Tonga is reaching its international commitments. Lastly, the large coastal coverage by SMAs, where fishing is still permitted, might also limit additional spatial planning and no-take marine protected areas not associated with the SMA program, or relegate them to areas far from population centres and with less conservation impact.

The establishment of an SMA effectively sequesters the tragedy of the commons at the village level, where ongoing resource conflicts might continue to persist, albeit within the community. However, in 2015 a project on Marine and Coastal Biodiversity Management in Pacific Island Countries gathered community members from existing SMAs to discuss "lessons learnt" (Tupou Taufa et al. 2016). Two key points raised were: i) to "acknowledge that there will always be community members who disagree; thus communities should move forwards after adequate consultation and majority agreement even if not 100% consensus"; and ii) that "where possible, include dissenting voices in the management of the SMAs". Therefore, while acknowledging that resource conflict might continue to exist within communities, it is at a level that allows for effective communication and collaboration between dissenting viewpoints.

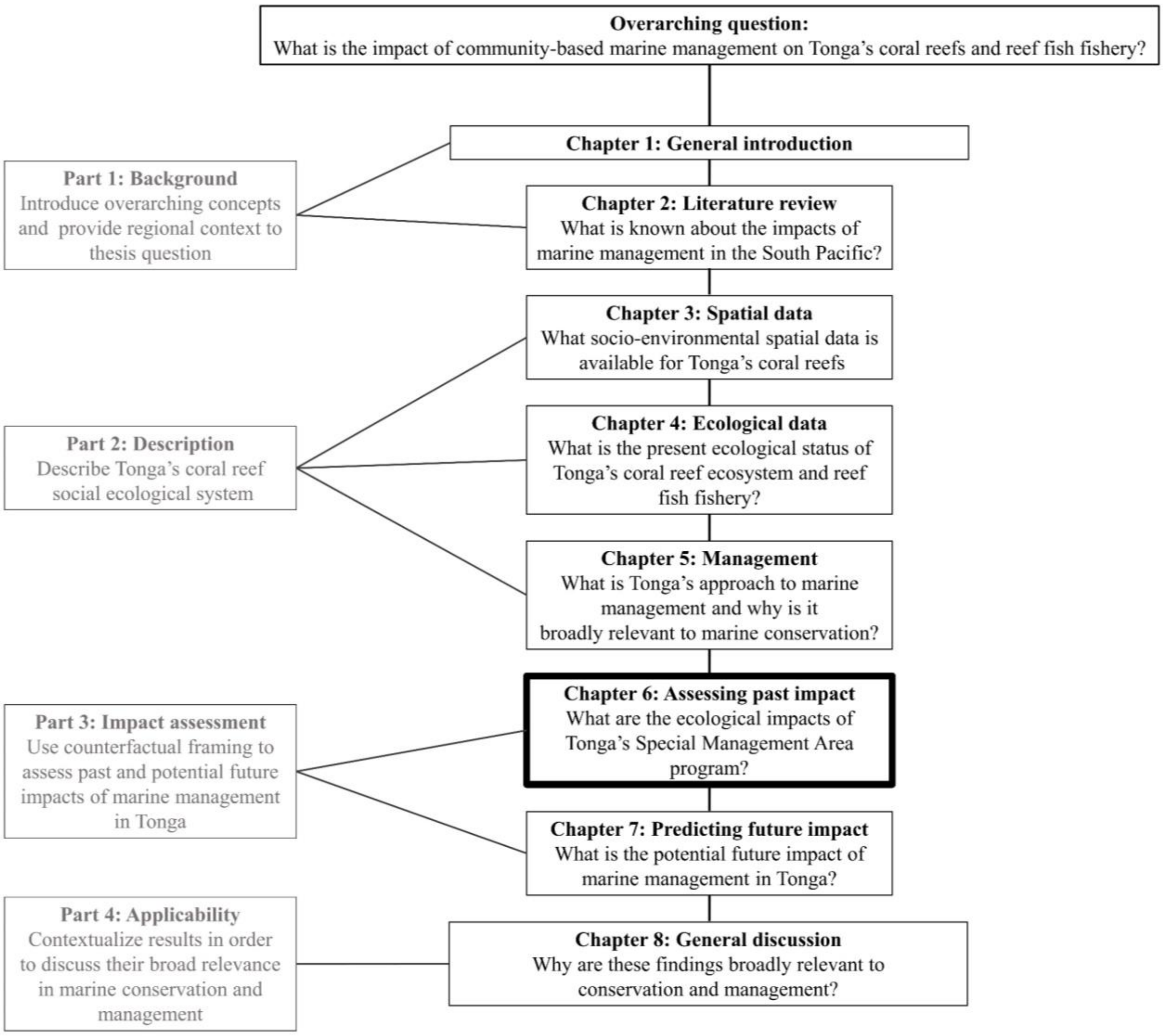
Lastly, it is important to note that the successful expansion of the SMA program in Tonga has relied largely on the fact that, prior to its inception, Tonga was entirely open access. Re-establishing a form of customary tenure has therefore been the prime incentive for strong engagement. A key consideration in expanding this program to other countries would be that support may be greatest in areas where existing management is weakest. For example, the SMA program in its current form might provide little incentive for groups in Vanuatu to implement no-take zones, where strong customary tenure already exists (Govan, 2009). However, other incentives such as providing formal recognition of customary tenure through legislation could provide similar enticements in these places.

5.6 Conclusions

The dual approach of Tonga's SMA program provides key insights into mechanisms by which to avoid known pitfalls in protected areas expansion. First, providing immediate incentives (e.g. exclusive access zones) that also foster long-term relationships with resources encourages groups that otherwise may be against management and conservation to implement protected areas. Then ensuring that protected areas occur within the boundaries of these exclusive access zones entices groups to protect areas of greater extractive value than they would likely do so otherwise. Applying this framework successfully to other regions will rely on understanding the specific short-term incentives that will ultimately foster the greatest long-term engagement in management and conservation.

Chatper 6: Community management yields positive impacts for coastal fisheries resources and biodiversity conservation

P. Smallhorn-West, K. Stone, D. Ceccarelli, S. Malimali, T. Halafihi, T. Bridge, R. Pressey, G. Jones



6.1 Abstract

Combining no-take marine reserves with exclusive access by communities to unreserved waters could provide the required incentives for community management to achieve positive impacts. However, few protected areas have been critically evaluated for their impact, which involves applying counterfactual thinking to predict conditions within protected areas if management had never occurred. Here, we use statistical matching to conduct a rigorous impact evaluation of dual management systems on coral reef fishes in Tonga, with communities having both full no-take areas and areas of exclusive fishing rights. No-take areas generally had positive impacts on the species richness, biomass, density and size of target reef fish, while exclusive access areas were similar to predicted counterfactual conditions. The latter is likely because overall fishing pressure in exclusive access areas might not actually change, although more fish could be exploited by communities with access rights. Our findings suggest that dual management is effective at incentivizing effective community-based no-take areas for biodiversity conservation and resource management.

6.2 Introduction

There is increasing evidence that appropriately situated marine protected areas (MPAs) with high compliance can produce positive outcomes for biodiversity and fisheries targets (Edgar et al., 2014; Gaines, White, Carr, & Palumbi, 2010). However, expansion of MPAs can be resisted by resource users over issues such as forced displacement, loss of access to seafood, and unfulfilled promises (Agardy, Notarbartolo, & Christie, 2011; Charles & Wilson, 2009). Balancing conservation priorities with human needs remains one of the key concerns in protected area research (Charles & Wilson, 2009).

Community-based marine management, whereby natural resource or biodiversity protection is conducted by, for, and with local communities (Western and Wright 1994) is seen as one of the best approaches to strike a balance between the interests of biodiversity conservation and resource users (Jupiter, Cohen, Weeks, Tawake, & Govan, 2014). However, despite widespread acceptance of community-based or co-management approaches, there are concerns that their expansion is driven by livelihoods and well-being objectives while benefits to biodiversity conservation are limited (Bartlett, Pakoa, & Manua, 2009). Therefore, even if positive ecological impacts are achieved locally as co-benefits with socially focused objectives, they might not scale to reach national or international biodiversity objectives (Gaines et al., 2010). Furthermore, if local priorities conflict with broader goals, then allowing resource users to take over management could result in prioritization of immediate benefits at the expense of long-term national or international objectives, such as biodiversity conservation or sustainable development.

In order for community management to achieve both local and national or international objectives, it is critical to identify incentives for local actions to ensure long-term change at a broad scale (Brockington & Schmidt-soltau, 2017; Ferraro & Hanauer, 2011). Access restrictions, such as found within territorial use rights for fisheries (TURFs) or locally-managed marine areas (LMMAs), are a fisheries management tool in which communities or groups of fishers are given distributed or inherited access rights to a portion of the ocean (Gelcich et al. 2012; Jupiter et al. 2014; Villasenor-Derbez et al. 2019). Access restrictions can promote a sense of stewardship and incentivize communities to sustainably manage their resources (Gelcich et al. 2012). Importantly, access restrictions and no-take marine reserves are not mutually exclusive (Jupiter et al. 2014; Villasenor-Derbez et al. 2019). Instead they can be combined, whereby access restrictions can act as the incentive for establishing no-take reserves when communities might not otherwise be willing to give up areas for conservation.

The effectiveness of community-based marine management should be assessed by its impact, defined as the intended or unintended consequences that are directly or indirectly caused by an intervention (Adams, Barnes, & Pressey, 2019; Pressey, Visconti, & Ferraro, 2015). However, determining impact can be challenging because it involves estimating the counterfactual condition if no action or a different action had been taken (Ferraro, 2009; Pressey, Weeks, & Gurney, 2017). Estimating counterfactuals requires quantifying the extent to which observed conditions are the result of the intervention, or whether environmental or social contextual factors are masking failure or exaggerating success (Adams et al., 2019). While impact evaluation techniques are well developed in many other fields of research (e.g. medicine, education and development aid) (White 2009), few established protected areas or conservation policies have been critically evaluated for their impact (but see Ahmadia et al., 2015; Ferraro, 2009; Gill et al. 2017) (Pressey et al., 2017; Smallhorn-West, Weeks, Gurney, & Pressey, 2019). Here, we conduct a rigorous impact evaluation using statistical matching to determine the ecological impact of a dual approach to community-based marine management combining access restrictions and no-take reserves. We focus on Tonga's national Special Management Area (SMA) program, in which communities are granted exclusive access to fishing grounds (SMAs) in exchange for making parts of them permanent no-take zones. The no-take zones are locally called Fish Habitat Reserves (FHRs), the size and location of which are determined at the communities' discretion. While the local objectives are based largely on reviving coastal fisheries resources, Tonga is also committed to various international biodiversity conservation targets (e.g. the Convention for Biological Diversity Strategic Plan for Biodiversity 2011-2020 and the 20 Aichi targets) (Anon, 2013) and the SMA program is the primary focus of conservation efforts in the country. We conducted ecological surveys and analysis to compare the current ecological state of Tonga's oldest SMAs to their estimated counterfactual conditions to determine whether both SMAs and FHRs can yield positive impacts for both coastal fisheries resources and biodiversity conservation.

6.3 Methods

Tonga's SMA program launched in 2006 and, as of October 2019, includes 93 SMA or FHR areas. Our impact evaluation covers only SMAs established prior to 2014 and at least 3 years old at the time of ecological surveys. These requirements applied to seven SMA communities (with corresponding FHRs) (Fig. 1), which were spread across the three main island groups in Tonga, with two in Tongatapu, four in Ha'apai and one in Vava'u.

Ecological surveys were conducted from 2016 to 2018 across 375 sites in Tonga, both inside FHRs and SMAs and either in areas open to fishing or where management had only recently been implemented (Table 1). Areas open to fishing and newly implemented management areas were classified as control areas, providing the pool of control transects that could then be matched with transects in managed areas. At each site, four to six 30 m belt transects were laid parallel to the reef contour at depths of three to twelve meters, resulting in a minimum of 12 transects within each SMA

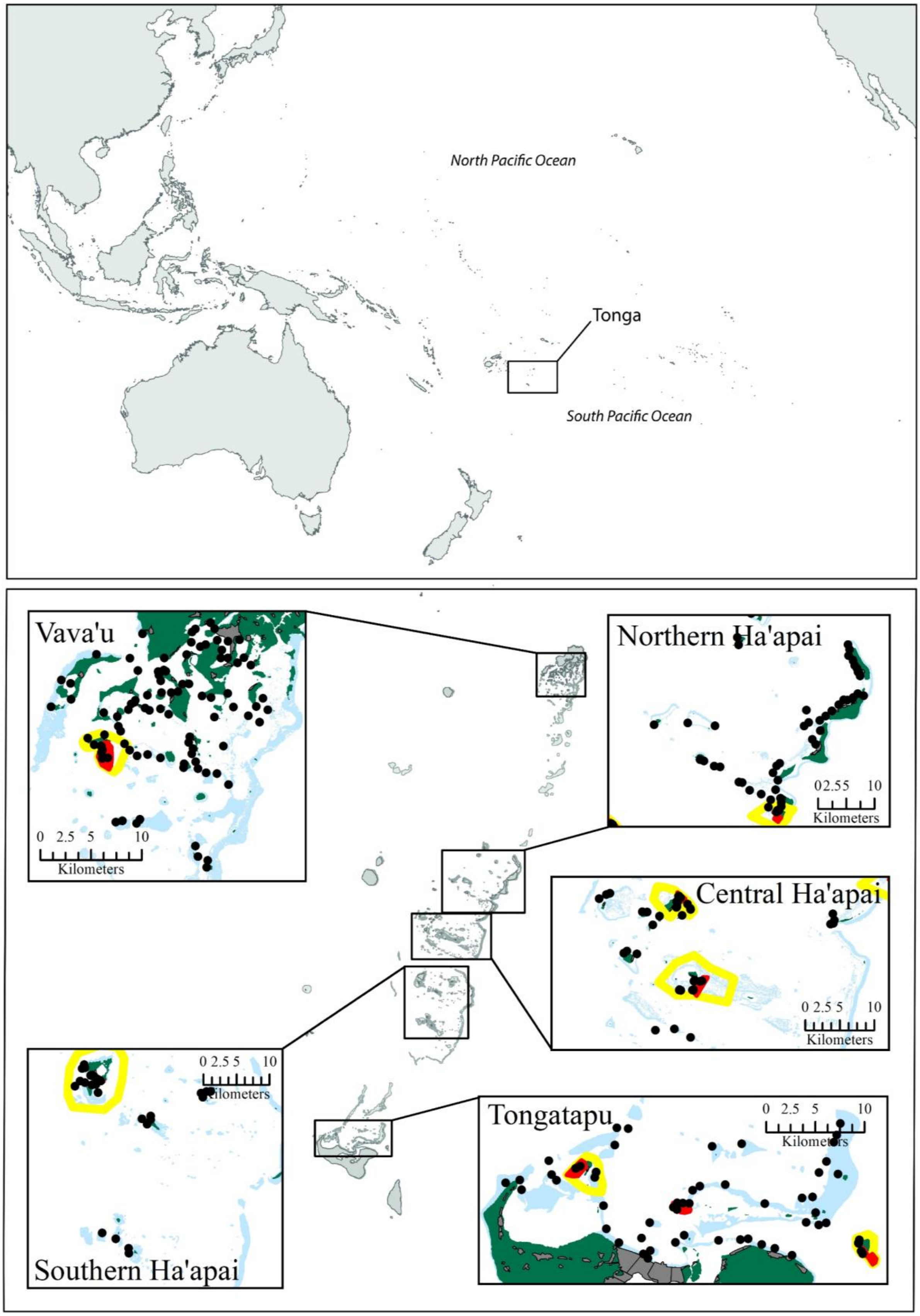


Figure 6.2 Map of Tonga showing the 14 Special Management Areas and Fish Habitat Reserves included in the impact evaluation. Yellow denotes Special Management Areas and red denotes no-take Fish Habitat Reserves. Black circles denote survey sites. Green represents land and grey indicate villages.

Table 6.1. Summary of fish survey data sets available to the project. ARC CoE CRS = Australian Research Council Centre of Excellence for Coral Reef Studies. ADB = Asian Development Bank. VEPA = Vava'u Environmental Protection Association.

Project	Department	Funding	Surveyor	Island group	n. sites	Year
National monitoring project	Ministry of Fisheries	James Cook University	Patrick Smallhorn-West	Tongatapu	60	2018
		ARC CoE CRS		Ha'apai	125	2018
		McIntyre Adventure/ Halaevalu Mata'aho Marine Discovery Centre National Geographic Society		Vava'u	93	2017
ADB Vava'u Special Management Areas baseline surveys (Ceccarelli, 2016)	Ministry of Fisheries Department of Environment VEPA	ADB	Dr. Daniela Ceccarelli Karen Stone	Vava'u	36	2016
VEPA Special Management Areas baseline surveys	VEPA	VEPA	Karen Stone	Vava'u	4	2017
WAITT Institute field surveys (Stone et al., 2017)	Department of Environment	WAITT Institute	Heather Kramp	Ha'apai	18	2017
	VEPA		Karen Stone	Vava'u	39	2017

n. FHR transects	n. SMA transects	n. control transects
143	200	1285

and FHR. The abundance and size of all large mobile fish were recorded to species level within a five-metre belt. All small, site-attached reef fish species were recorded along a two-metre belt. The length and abundance of reef fish were converted to biomass following published length-weight relationships for each species (www.fishbase.org). Nineteen outcome variables of reef fish community composition were selected as meaningful indicators that aligned with the intended management objectives of the SMA program and international biodiversity targets (Tonga Fisheries Division, Ministry of Agriculture & Food, 2010). These 19 outcome variables were: total reef fish species richness, total and family level biomass, density and mean total length of the five most commonly targeted reef fish families (Parks, 2017) (Acanthuridae, Lethrinidae, Lutjanidae, Scaridae and Serranidae). We selected a 20 cm size cut off for biomass and density values because larger sized fish represent the fishable biomass of target reef fish species currently available to fishers and likely to be targeted.

We then selected 11 contextual factors to use in the statistical matching model. These encompassed environmental and social features of coral reefs that are known to influence either the response variables or the configuration of protected areas (Table 2).

Impact evaluation

Counterfactual predictions for managed areas were estimated by statistically matching SMA and FHR transects to a large pool of control transects according to the characteristics of their covariates, using a combination of fixed and propensity score matching (Ho, Imai, King, Stuart, & Whitworth, 2018; R Core team, 2017). Propensity scores are a statistical technique that summarize many covariates into a single score (Olmos & Govindasamy, 2015). They are defined as the conditional probability of assigning a unit to a particular treatment (i.e. likelihood of management as SMA or FHR), given a set of observed covariates ($z = i|X$), where z = treatment, i = treatment condition and X = covariates. The probability of assignment is estimated using a logistic regression model, where treatment assignment is regressed on the set of observed covariates. The propensity score then allows matching of transects with the same likelihood of receiving management.

Matching was conducted at the transect level. FHR and SMA transects were analysed separately, but matched to the same overall pool of control transects (Fig. 2) (Table S1, S2). The variables habitat type, island group, and surveyor were all fixed so that control transects could be paired only with managed transects if they matched the exact combination of these covariates. Following fixed matching, all remaining covariates were weighted equally, and the nearest neighbour distance was used to match transects with the closest propensity score first. We sampled with replacement, meaning control transects could be matched with multiple managed transects. In addition, each managed transect could also be paired with multiple control transects and, if multiple matches occurred, the mean was used as the estimated counterfactual. A pre-specified tolerance (i.e. caliper) of 0.25 standard deviations of the sample estimated propensity scores was set to ensure only high-quality matches (Olmos & Govindasamy, 2015).

Covariate balance (i.e. the difference in the distribution of covariates across managed and control transects) was tested prior to and following matching by estimating the normalized difference between managed and control transects for each covariate in the model. An omnibus test, which tests whether at least one variable in the model is unbalanced, was conducted using the XBalance routine via a chi-squared test (Ho et al., 2018). For standardized differences, values over 25% between managed and control transects are considered unbalanced (Olmos & Govindasamy, 2015). Following matching there was no evidence of imbalance for FHRs or SMAs (Fig. 2) (Table S3). A total of 129 out of 143 FHR transects and 159 out of 200 SMA transects were matched to 247 and 397 control transects, respectively. All remaining unmatched managed and control transects were discarded from the analysis.

Table 6.2. Eleven contextual factors that were included in the matching model and used to estimate counterfactual conditions for transects inside Fish Habitat Reserves and Special Management Areas.

Variable	Description	Reference
Depth	Depth (m), collected *in situ.*	Lindfield et al. (2014)
Distance to land	Distance (m) from the nearest land source (Smallhorn-West et al., In review).	Cinner et al. (2013)
Distance to village	Distance (m) from the closest village (Smallhorn-West et al. In review).	Cinner et al. (2013)
Fishing pressure	Normalized (0-100) abundance of commercial and subsistence fishers (adjusted for catch) extrapolated across the coral reefs of Tonga. It constitutes a unit-less value of relative long-term fishing effort throughout the region (Smallhorn-West et al., In review).	Wilson et al. (2010)
Habitat	Exposed, semi-exposed or fringing, collected *in situ.*	Wilson et al. (2010)
Island group	Ha'apai, Tongatapu or Vava'u.	-
Total live coral cover (%)	Collected either by the point intercept method or from photo quadrats annotated using the automated image analysis software CoralNet and BenthoBox.	Wilson et al. (2010)
Habitat macrocomplexity	Estimate of habitat complexity collected *in situ* on a five-point scale from low and sparse relief (score = 1) to exceptionally complex with numerous caves and overhangs (score = 5)	Wilson et al. (2010)
Slope	Estimate of reef slope collected *in situ* on a five-point scale from < 10º (score = 1) to 90º (score = 5).	Ceccarelli (2016)
Surveyor	Dr. Daniela Ceccarelli, Heather Kramp, Karen Stone or Patrick Smallhorn-West.	-
Wave energy	Average daily wave energy (joules per m^2) (Smallhorn-West et al. In review).	Mumby et al. (2013)

Finally, linear mixed effect models with community and site included as random factors, with site nested within community, were used to test the overall differences between matched FHR or SMA and control areas across each of the 19 outcome variables. Models were created with both fixed and random slopes and the one with the lowest AIC score selected. All biomass and density variables were log(x+1) transformed. Model fit was examined using partial residual plots and tested with chi-squared tests on the residual sum of squares and residual degrees of freedom.

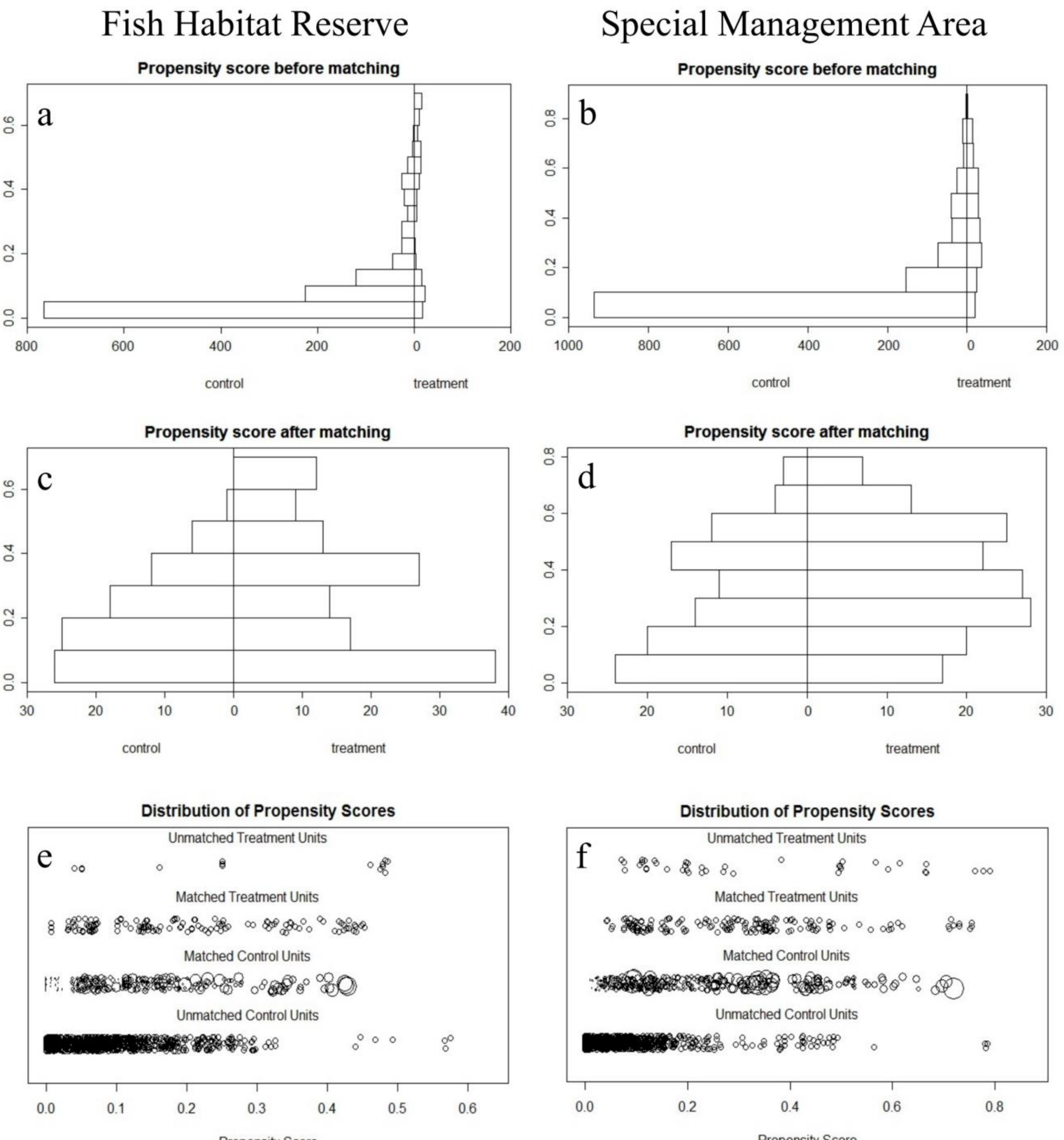


Figure 6.2. Details of the matching procedure. Parts aand c represent frequency histograms of propensity scores prior to and following matching for Fish Habitat Reserves. Parts b and d represent frequency histograms of propensity scores prior to and following matching for Special Management Areas. Parts e and f show jitter plots of the propensity score distributions of matched and unmatched transects following matching for Fish Habitat Reserves and Special Management Areas, respectively. The size of the each control circle represents the number of treatment transects with which it was matched. All unmatched transects were discarded from the analysis.

6.4 Results

Overall, there were consistent positive ecological impacts of FHRs (Fig. 3) (Table 3). Both overall target species biomass and density were approximately 5.3 and 3.6 times greater, and species richness 15% higher, inside no-take reserves than matched control transects. These impacts were most pronounced in the Scaridae family, with 3.7 times and 2.5 times as much biomass and density of scarids inside FHRs, respectively. Although the overall density of Lethrinidae and Lutjanidae were small compared to other families, FHRs still supported 70% greater densities than control transects. Fish were also on average larger inside FHRs, with the mean total length of four of the five main target reef fish families two to six centimetres greater inside FHRs than matched control transects.

There was limited evidence of ecological impacts inside the SMAs. The most consistent trend was a small increase in the average size of the five main target reef fish families inside SMAs, although this was significant at $p < 0.05$ only for Lethrinidae and Scaridae. This trend was not evident in the biomass or density of target reef fish and, in three instances biomass and density were significantly lower inside SMAs than matched control sites (Lutjanid biomass and density and Serranid biomass). There was no evidence of an SMA effect on overall reef fish species richness.

Table 6.3 (overleaf). Model results for mixed effect models examining the ecological impacts of Tonga's Special Management Area program shown as absolute mean +/- 95% CI values of matched Fish Habitat Reserve or Special Management Area and control transects. LCL = lower confidence limit. UCL = upper confidence limit. Controls represent the mean of all matched transects. Biomass is measured as kilograms per hectare of target species (>20 cm total length). Density is measured as the number of individuals per 1000 m^2 of target species (>20 cm total length). Species richness is measured as the number of reef fish species per transect. Length is measured as total length in centimetres.

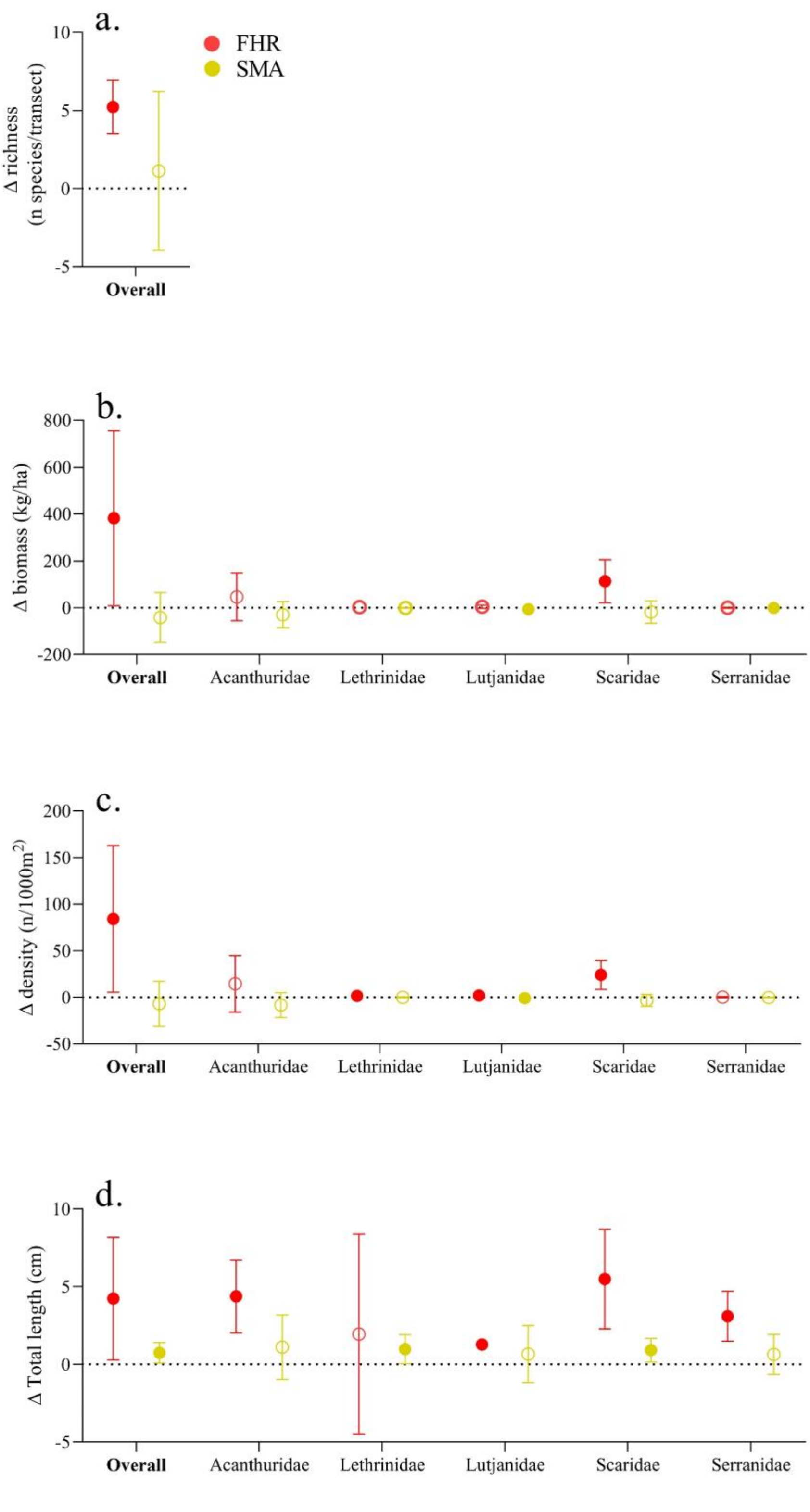


Figure 6.3. Ecological impacts of Tonga's Special Management Area program plotted as the mean difference between matched Fish Habitat Reserve or Special Management Area transects and control transects with +/- 95% confidence intervals. Closed circles represent values with margins not overlapping zero and statistically significant to $p < 0.05$. a) Total reef fish species richness; b) biomass of target species (>20 cm total length); c) density of target species (>20 cm total length); and d) mean total length of target species (juvenile to adult). Biomass and density plots represent differences in sum totals between transects and therefore overall values are cumulative of each family. The total length plot signifies differences in mean size of individual fish and therefore the overall columns represents the mean difference across all families.

Family	Variable	Fish Habitat Reserve Treatment	LCL	UCL	Control mean	LCL	UCL	df	t score	p-value
Overall	Richness	39.24	35.29	43.18	34.02	30.07	37.96	196	6.019	< 0.05
	Biomass	469.88	83.52	856.24	87.94	-30.19	206.07	6	2.499	< 0.05
	Density	116.46	29.53	203.39	32.43	0.55	64.31	6	2.613	< 0.05
	Total length	22.41	19.50	25.31	18.18	12.68	23.67	196	2.115	< 0.05
Acanthuridae	Biomass	77.56	-29.52	184.64	31.56	-18.22	81.33	6	1.105	0.311
	Density	26.96	-5.66	59.58	12.60	-5.13	30.34	6	1.155	0.292
	Total length	16.13	13.36	18.89	11.75	8.38	15.13	196	3.693	< 0.05
Lethrinidae	Biomass	5.64	1.22	10.05	3.55	0.77	6.34	6	1.823	0.118
	Density	3.27	1.81	4.74	1.95	1.08	2.82	6	2.972	< 0.05
	Total length	21.50	19.10	23.91	19.56	10.97	28.15	196	0.596	0.552
Lutjanidae	Biomass	11.78	4.12	19.44	8.15	2.85	13.45	6	1.273	0.250
	Density	4.60	2.86	6.33	2.69	1.68	3.71	6	2.976	< 0.05
	Total length	29.38	23.67	36.48	23.01	16.89	31.34	196	2.193	< 0.05
Scaridae	Biomass	155.46	38.64	272.29	42.46	10.55	74.38	6	3.017	< 0.05
	Density	39.48	17.41	61.55	15.53	6.85	24.20	6	3.760	< 0.05
	Total length	22.89	19.90	25.88	17.40	12.90	21.90	196	3.382	< 0.05
Serranidae	Biomass	2.99	0.98	4.99	3.27	1.07	5.46	6	-0.463	0.660
	Density	2.01	1.26	2.76	1.91	1.20	2.63	6	0.431	0.681
	Total length	21.23	17.48	24.99	18.14	14.38	21.89	196	3.805	< 0.05

Family	Variable	Special Management Area Treatment	LCL	UCL	Control mean	LCL	UCL	df	t score	p-value
Overall	Richness	34.89	29.23	40.56	33.77	28.99	38.55	274	0.436	0.663
	Biomass	143.54	13.30	273.78	185.64	3.73	367.54	6	-0.973	0.368
	Density	47.26	8.96	85.56	54.27	18.46	90.07	6	-0.711	0.504
	Total length	21.07	18.34	23.80	20.33	17.60	23.06	274	2.228	< 0.05
Acanthuridae	Biomass	19.61	-7.12	46.34	49.61	-22.53	121.75	6	-1.322	0.234
	Density	9.68	-2.18	21.53	18.09	-1.89	38.07	6	-1.542	0.174
	Total length	14.25	11.93	16.56	13.14	9.05	17.23	274	1.055	0.292
Lethrinidae	Biomass	2.49	1.51	3.46	3.62	2.20	5.04	6	-2.152	0.075
	Density	1.83	1.41	2.26	1.99	1.53	2.46	6	-0.773	0.469
	Total length	19.73	17.95	21.50	18.74	16.97	20.52	274	2.076	< 0.05
Lutjanidae	Biomass	2.65	0.74	4.56	9.29	2.59	16.00	6	-3.085	< 0.05
	Density	1.93	1.35	2.50	2.95	2.07	3.83	6	-3.745	< 0.05
	Total length	29.36	25.67	33.05	28.70	25.01	32.39	274	0.710	0.478
Scaridae	Biomass	82.44	20.98	143.91	101.38	43.83	158.94	6	-0.981	0.364
	Density	25.64	14.37	36.91	29.07	16.29	41.85	6	-1.288	0.245
	Total length	20.91	17.52	24.29	19.99	16.61	23.38	274	2.373	< 0.05
Serranidae	Biomass	2.12	0.52	3.72	3.70	0.91	6.48	6	-2.517	< 0.05
	Density	1.67	1.11	2.22	2.01	1.34	2.68	6	-2.068	0.084
	Total length	22.11	17.46	26.77	21.48	16.83	26.13	274	0.969	0.333

6.5 Discussion

This study demonstrates that the dual approach to community-based marine management in Tonga, including exclusive access areas and associated no-take reserves, can be scaled up to achieve meaningful impacts at a national level for both coastal fisheries resources and biodiversity conservation. The success of the no-take areas is likely linked to the incentive provided by exclusive access to and greater control over local resources. While there were few quantifiable impacts of exclusive access areas, overall the combination of having both types of management areas is positive. Our study provides one of the first full impact evaluations of a country's MPAnetwork that has incorporated counterfactual analyses and is quantifiably robust to contextual conditions (but see Gill et al. 2017). This approach can therefore be used as a template by which to structure future impact evaluations of MPAs. In addition, Smallhorn-West et al. (2019) also provides detailed recommendations for key conditions that should be in place for this approach to be useful. These results have important implications for management of reefs and for understanding how to balance the competing goals of improving coastal fisheries resources and biodiversity conservation in developing nations.

At the outset, it is important to acknowledge that, while positive ecological impacts are evident within the no-take FHRs, these represent only a small fraction, (3% as of October 2019), of Tonga's total coral reef area. Fish stocks and species richness will likely continue to increase within Tonga's network of FHRs as new areas are implemented and existing areas grow older. However, despite these improvements, it is unclear whether the FHRs and any potential spillover will be sufficient to meet food supply needs while maintaining coral reef ecosystem function. In addition, given the lack of visible ecological impacts within the SMAs it remains unclear the extent to which these areas are changing patterns of food consumption and nutrition within Tongan communities. Therefore, given the objective of "reviving the health and status of coastal fisheries resources for current and future generations" (Fisheries Division, Ministry of Agriculture & Food, 2010), additional management actions, such as changing fishing practices for the inshore commercial fisheries (Tonga Fisheries Sector Plan, Section 42), along with the continued expansion of the SMA program, might be necessary to achieve broad objectives. The SMA program therefore represents a platform from which to build in order to make progress towards many of the national and international biodiversity and sustainability targets.

Our findings can be used to improve our understanding of Tonga's progress towards achieving both national and international targets for marine biodiversity conservation and sustainability (Table S4). In the Tonga Fisheries Sector Plan, our results provide the first evidence of positive impacts of the SMA program under section 4.1 and 8.1 for Sustainable Community Fisheries. Likewise, under Tonga's National Strategic Biodiversity Action Plan, the SMA program is making progress towards Theme Area 2: Marine Ecosystems (Objectives 2.2, 2.3 & 2.4) and Theme Area 3: Species Conservation (Objectives 3.1, 3.2 & 3.4), as well as helping with barriers to effective reporting (Barrier 3: Monitoring) (Kingdom of Tonga's 5th CBD report, 2014). Under the Aichi targets, the SMA program is supporting progress under targets 6, 10, 11 and 15. However, while progress towards national and international targets is being made, caution is needed in using these targets to quantify success. For example, there is concern that too much focus on the area-based Aichi target 11, which aims to protect 10% of marine area by 2020, is encouraging minimal overlap between pressures and protection by favoring large, offshore reserves (Devillers et al. 2015). Care should therefore be taken in considering the conservation impacts of management, regardless of contributions to area protected or even representativeness (Pressey et al. 2017). Despite these caveats, it is clear that Tonga's SMA program represents a strong positive step for the country towards improving marine sustainability and biodiversity conservation.

A key principle in MPA design is that the size of no-take MPAs should be sufficient to incorporate the home ranges of the species they are intended to protect (Weeks, Green, Joseph, Peterson, & Terk, 2016). Numerous studies have also demonstrated that larger no-take MPAs are more likely to achieve positive results than smaller reserves (e.g. Edgar et al., 2014). However, the largest of Tonga's FHRs is only 2.6 km^2, and many are less than 1 km^2; yet they still consistently result in positive impacts, albeit across a limited total extent. While counterintuitive, these findings are consistent with other studies demonstrating that even small reserves (< 1 km^2) can produce significant biological responses (Bonaldo, Pires, Roberto, Hoey, & Hay, 2017; Russ and Alcala 1996; Russ et al. 2004). Given that the home ranges of many key target species are larger than the areas set aside for management, it is unclear by what mechanism or to what extent fishes are avoiding capture if they move beyond the boundaries of the FHR. Many protected areas globally are less than 1 km^2 in size (Costello & Ballantine, 2015) and further studies are necessary to investigate this effect; but there is evidence that some fishes, even wide-ranging species, may alter their behavior within a short timeframe to maximize the protection offered by no-take zones (Mee et al. 2017). In addition, the observed differences in recovery between reef fish families might also be due to the faster growth rates of scarids (Grandcourt 2002), combined with the relatively young age of the SMA program.

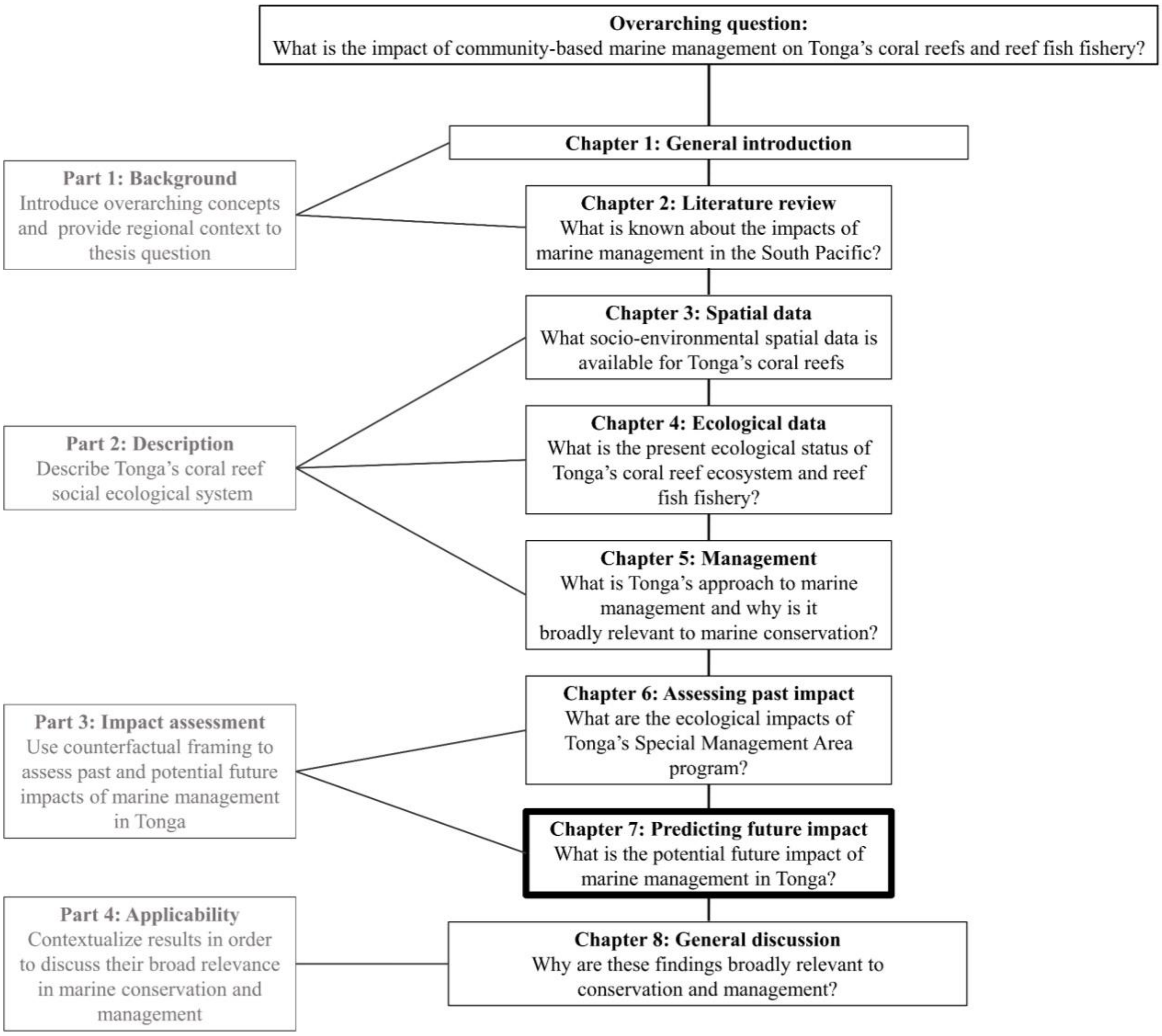
The results of this study provide little evidence for positive ecological impacts within SMA areas, where fishing still occurs. This result is consistent with a recent global meta-analysis of MPA effectiveness demonstrating that moderately protected MPAs rarely perform better than unprotected areas (Zupan et al., 2018). However, while FHRs were established to explicitly address conservation objectives, the goals of SMAs are primarily socioeconomic. Key management objectives for SMAs are to "raise community awareness on fisheries conservation and management, promote sustainable fishing practices and improve living standards within the community" (Fisheries Division, Ministry of Agriculture & Food, 2010). In addition, SMAs are generally seen as a way to re-establish customary tenure, which is common in many Pacific nations but was lost in Tonga, and to prevent large-scale commercial fishing activities from destroying local food security (Gillett, 2017). As such SMAs may still be achieving their desired objectives even if there is no observed ecological change. Furthermore, any recovery of target species likely to occur from reduced fishing pressure by sources outside the community may be counteracted by increased local fishing. That in most cases ecological impacts inside SMAs were not negative suggests that exclusive access is not increasing net fishing pressure, merely changing who fishes (Polunin, 1984), and therefore the net benefit of the dual system is positive. Ultimately the impacts of SMAs are be more likely to be found in people's nutrition and in their understanding of marine management than in the ecosystem itself.

Jupiter et al. (2014) outlined six management actions to achieve a broad range of objectives in community-based marine management: permanent closures, periodically harvested closures, species restrictions, gear restrictions, access restrictions, and alternative livelihood strategies. Within this framework, Tonga's SMA program represents a combination of access restrictions (i.e. SMAs) and permanent closures (i.e. FHRs). A key drawback suggested for access restrictions is that they might not be sufficient to maintain biomass or enhance sustainability, and that they "will not necessarily change the volume harvested, just who harvests it" (Jupiter et al., 2014; Polunin 1984). However, these authors also suggest that access restrictions might be necessary to facilitate other management actions. While these other actions, such as permanent closures, might have strong evidence to support their effectiveness, there was concern that, given they are not historically prevalent in the Pacific (Johannes, 1978), there could be social barriers to their effective implementation (Foale & Manele 2004). Tonga's management program builds on this hypothesis by utilizing SMAs (i.e. access restrictions) as necessary tools, despite no evident ecological impacts, to incentivize the implementation of FHRs (i.e. permanent closures). Given the open access history of Tonga's marine management (Gillett, 2017), FHRs might have had little support otherwise.

Tonga's SMA program represents a successfully vetted combination of management actions to add to the tool kit of marine managers aiming to achieve ecological impact in the community context. However, the success of this program has relied on reinventing customary tenure in a country with little historical management. While this approach has been successful in Tonga, other countries with stronger traditional access rights might have greater difficulty in providing incentives for permanent closures. A key consideration is therefore that support for this program will likely be greatest in areas where previous management is weakest. Determining the historical context of community priorities and using these to successfully incentivize conservation will be a key factor in the successful implementation of this framework in other regions.

Chapter 7: Predicting impact to assess the efficacy of community-based marine reserve design

P. Smallhorn-West, T. Bridge, S. Malimali, R. Pressey, G. Jones



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7.1 Abstract

The efficacy of different strategies to manage marine resources should ultimately be assessed by their impact, or ability to make a difference to ecological and social outcomes. While community-based and systematic approaches to establishing marine protected areas have their strengths and weaknesses, comparisons of their effectiveness often fail to explicitly address potential impact. Here we predict conservation impact to compare recently implemented community-based marine reserves in Tonga to a systematic configuration specifically aimed at maximizing impact. Boosted regression tree outputs indicated that fishing pressure accounted for ~24% of variation in target species biomass. We estimate that the community-based approach provides 84% of the recovery potential of the configuration with the greatest potential impact. This high potential impact results from community-based reserves being located close to villages, where fishing pressure is greatest. These results provide strong support for community-based marine management, with short-term benefits likely to accrue even where there is little scope for systematic reserve design.

7.2 Introduction

The prevailing combination of ongoing ecosystem exploitation and limited conservation resources highlights the critical need to develop rapid, cost-effective management actions. No-take marine protected areas (MPAs) are a key tool used in marine conservation and are suggested to enhance ecosystem resilience and reduce the decline of fisheries resources (Halpern and Warner 2002; Gaines et al. 2010; Mellin et al. 2016). The objectives for MPA implementation are also broad (Govan and Jupiter 2013; Jupiter et al. 2014), targeting both general (e.g. increasing biodiversity) and local (e.g. maintaining fish stocks) conservation priorities. In some cases, reserve systems have been systematically designed to meet particular objectives of species inclusion, based on the best available knowledge of ecosystems and species distributions (Pressey and Bottrill 2009). In other cases, reserves have been established at ad hoc locations by local communities (Mills et al. 2012). While each approach has its strengths and weaknesses, the degree to which the two differ in effectiveness depends upon their likely impact. The conservation impact of a reserve is the difference it makes to one or more intended or unintended outcomes relative to no intervention or a different intervention (Pressey et al. 2015; Pressey et al. 2017).

Community-based approaches to MPA management are common in developing nations and tend to involve the opportunistic establishment of reserves where there is a willingness of local resource owners to participate in marine management (Mills et al. 2012; Gaymer et al. 2014; Horigue et al. 2015). Here we define community-based conservation as natural resource or biodiversity protection by, for, and with the local community (Western and Wright 1994). This governance approach generally prioritises the goals of local communities, such as maintaining target fisheries, and responds to local constraints and opportunities (Ban et al. 2011), but does not focus explicitly on goals such as biodiversity conservation *per se*. Local engagement results in greater compliance, participation in enforcement and other management activities (Gurney et al. 2016), with a longer-term commitment to reserves (Gaymer et al. 2014). Community-based reserves can also be implemented effectively, even without the coordination and logistic support from a centralized government (Cox et al. 2010). However, conservation efforts implemented opportunistically and focused on local priorities might not meet biodiversity conservation objectives (Horigue et al. 2015).

Other approaches to reserve design include top-down central management, which we define as natural resource and biodiversity protection by a central governing authority. Central management can incorporate systematic conservation planning, which is characterized by explicit objectives and considerations of spatial context to guide the selection and management of conservation areas (Pressey and Bottrill 2009). The systematic approach theoretically has the capacity to target conservation actions in a way which maximizes impact, thereby being more effective at achieving national and international conservation objectives (Hansen et al 2011; Mills et al 2012). However, globally it is now well established that many protected areas are residual, in locations that are less than likely to be affected by extractive activities (Joppa and Pfaff 2011; Devillers et al. 2015). Residual MPAs might be more likely to arise from central management, with political agendas minimizing conflict with extractive uses while maximizing perceived gains for conservation, with gains often gauged by misleading measures such as MPA extent (Pressey et al. 2017).

While both central and systematic MPA planning can incorporate the interests of communities to varying degrees, the conservation actions they suggest are frequently at odds with the interests of communities, and often face strong opposition from stakeholders (Bennet and Dearden 2014). Local communities might not feel involved in these processes, so compliance can be low (Gaymer et al 2014). While, in theory, the ability of these top-down approaches to achieve target objectives will generally be greater than ad hoc community-based management, they often fall short in practice (Ban et al. 2011; Gaymer et al. 2014).

The most common method used to compare systematic and community-based conservation planning has been to rate their abilities to reach habitat representation targets (e.g. Ban et al 2011; Hansen et al. 2011; Mills et al 2012; Horigue et al. 2015; Bode et al. 2016). Generally, this approach suggests that community-based MPA designs either fail to reach national conservation targets for habitat representation or fall well below the systematic approach. However, the pervasive use of habitat representation as the sole basis for identifying conservation priorities risks failure to achieve impact (Pressey et al. 2017). Despite extensive literature on the relative pros and cons of community-based and systematic MPA design, the effectiveness of both methods in terms of conservation impact is unknown. Furthermore, while there is now an extensive body of literature measuring ecological outcomes of MPAs, few tools exist to predict the relative impact of alternative reserve designs during the planning phase.

Here, we predict the potential conservation impact, measured as the recovery of target species biomass, of alternative configurations of no-take MPAs in the Vava'u island group of Tonga. Tonga has recently expanded its marine conservation program to incorporate the widespread use of community-managed MPAs, of which 13 were implemented in Vava'u in 2016-17. In this program, the size and location of MPAs are determined by local communities rather than systematically by the government based on ecological and/or social factors. We set out to answer two main questions: 1. How much of the predicted optimal impact is achieved by community-based MPAs?, and 2. What is the potential impact of a secondary, theoretical configuration of MPAs designed to balance both impact and maximum total potential biomass in MPAs?

7.3 Methods

Potential impact was calculated using a two-step process incorporating both social and ecological data. First, social data on fishing effort across Tonga were obtained from questions regarding fishing practices in the 2016 Tongan national census (Statistics Department of Tonga; 2017) and key informant interviews (Chapter 3). To quantify the relationship between fishing pressure and target species biomass, a continuous spatial layer of fishing pressure derived from the social dataset and ground-truthed during key informant interviews was included as a predictor variable (Harborne et al. 2016; 2018). Fishing pressure was calculated as the weighted abundance of fishers in each village overlaid on the fishing grounds of Vava'u using separate decay kernels for subsistence and commercial practices, derived from the key informant interviews (Chapter 3) (Thiault et al. 2017). Fisher abundance was weighted by district-level data on fishing practices (commercial or subsistence), gear type (spear and handline), and frequency of fishing activities. This fishing pressure metric assumes that, all else being equal, fishers preferentially select sites closer to home and move further out as closer sites become exhausted or closed to fishing. While the model might therefore be decoupled from current fishing effort, it is nonetheless useful in constituting the long-term effects of fishing on fish assemblages throughout the island group.

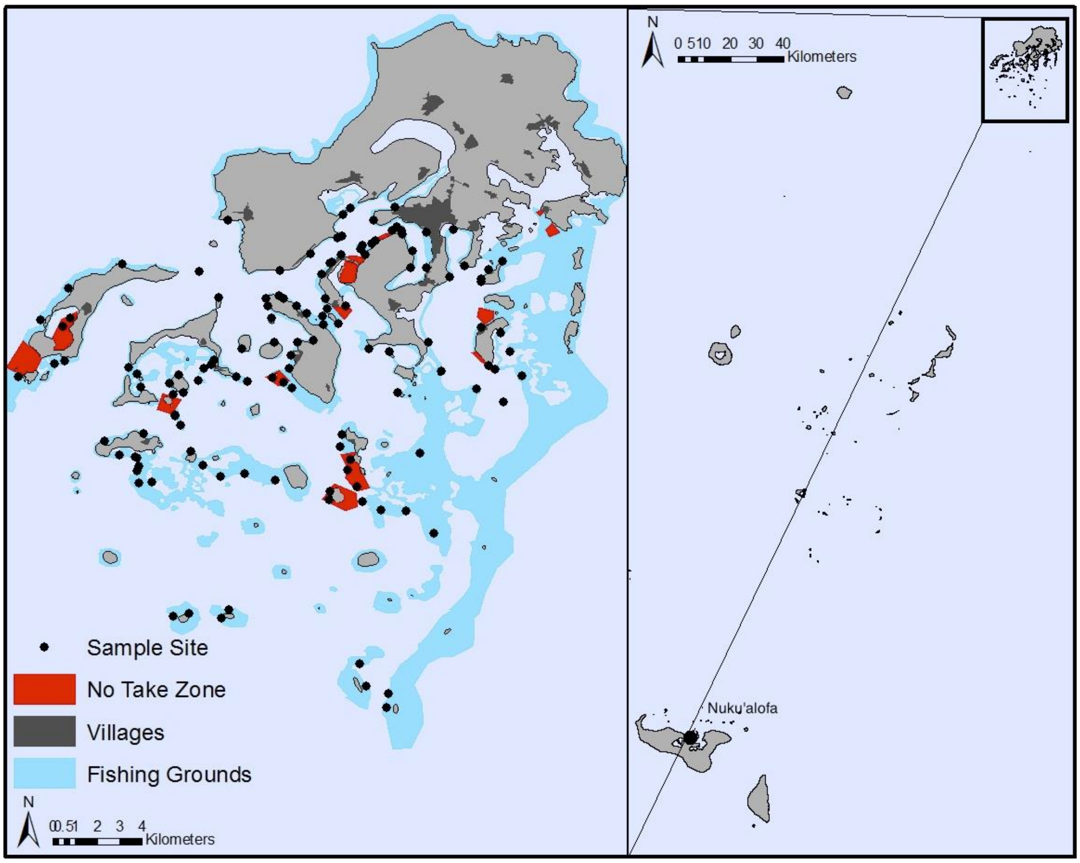


Figure 7.1. Map of Vava'u, Tonga, depicting the sample sites and new no-take reserves. Fishing grounds are defined as reef and patch reef environments at depths shallower than 10 m. Right-hand map shows the location of the Vava'u island group in the Tongan archipelago. Dark areas on land represent the outlines of villages.

Second, ecological surveys of coral reef fish community composition and biomass were conducted at 129 sites in Vava'u in 2016 to 2017 (Figure 7.1) (Chapter 4). At each site, the abundance and size of all target fish species was recorded in four 30 m x 5 m belt transects. Key target species were identified from survey questions in a baseline socioeconomic report for the new management areas (Parks 2017). The length and abundance of reef fish was converted to biomass following published length-weight relationships for each species (Kulbicki et al. 2005). We then used Boosted Regression Trees (Elith et al. 2008) and eight predictor variables (fishing pressure, habitat, wave energy, rugosity, slope, historic management status, district, surveyor) to create a spatial predictive model of the current biomass of target fish species across all reefs in the Vava'u group (Harborne et al. 2016; 2018). A random number variable was included and any predictor variables that explained less variability in the data than random were removed. Boosted Regression Trees are an additive regression model in which individual terms are simple trees, fitted in a forward, stagewise fashion (Elith et al. 2008). The model parameters (learning rate and tree complexity) were calculated across a series of values and the values that gave the best explanatory power were included in the final model. Confidence intervals were estimated around these fitted functions by taking 1000 bootstrap samples, to which we fitted the model. We used these samples to make separate predictions for the spatial data. All models were run using the 'gbm' package in R 3.X.

To assess the potential impact of the recently implemented no-take marine reserves in terms of recovery of target fish biomass, we re-inputted the data into the model with the same environmental variables, but with all fishing pressure values set to zero. Potential impact was calculated by subtracting, for each 50 m grid cell, current biomass from the potential biomass. The result was a layer continuous across the island group predicting the recovery of target species biomass for each 50 m grid cell.

The predicted impact of the current community-based configuration was then compared to two alternative systematic configurations with the same total area (8.8 km^2). The first comparison was made with the configuration that systematically protected an area equal to the community-based approach, but was configured to have the greatest impact. Impact is a measure of change and could therefore be equal in areas of both high and low predicted current biomass. Consequently, multiple configurations might exist with comparable impact, but with large differences in maximum recovered biomass. The community-based configuration was therefore also compared to a second systematic configuration, which aimed to maximize both potential impact and total biomass following recovery. This was done by preferentially selecting grid cells with high predicted biomass under no fishing when differences in impact between candidate cells were minimal.

A caveat to our estimation of impact is that it aimed to maximize the short-term benefit inside reserves only, without accounting for increased fishing pressure in non-reserve areas. However, because the relocated fishing pressure is spread over a large area, the fisheries squeeze effect is likely to be small. In addition, by maximizing the impact inside reserves, the recruitment subsidies from reserves will be greater than if reserves were situated in unfished areas.

7.4 Results

The predictor variables in the current biomass model explained 69% of the total variation in target fish biomass across Vava'u (Figure 7.2). The boosted regression tree learning rate was set to 0.001 and the interaction depth to 5, which resulted in a best iteration of 1720 trees. The greatest proportion of deviance (23.9%) was explained by fishing pressure (Figure 7.3a), with target species biomass declining rapidly as fishing pressure increased. However, the predictive power of fishing pressure decreased as fishing pressure increased, and this variable was unable to predict variation in target fish biomass at locations with values beyond ~40 fishers. The boosted regression tree models indicated that fish biomass increased rapidly with increasing distance from land (and decreasing population pressure), with biomass at the southernmost islands 2.5 times greater than around the inner islands (Figure 7.3b). The predictor variables district, historic management status and surveyor all explained less variability in the data than the random variable and were therefore removed.

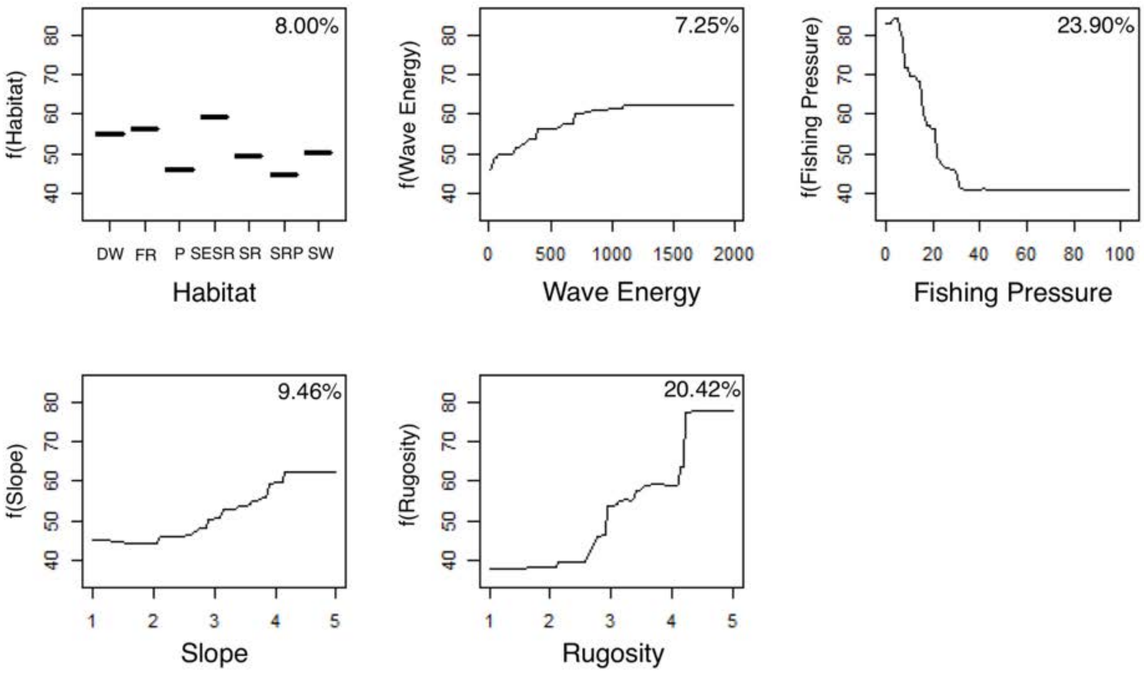


Figure 7.2. Boosted regression tree outputs. Relationships between each significant predictor variable and target species biomass (y-axes) after accounting for the average effects of all other variables in the model. Percent values represent how much of the deviance was explained by each variable. Habitat labels are: DW – deep wall, FR – exposed forereef, P – bare pavement, SESR – semi-exposed sloping reef, SR – sheltered reef, SRP – sandy rubble with patches, SW – shallow wall. Wave energy was calculated as joules per square meter. Fishing pressure is the abundance of fishers per grid cell fishing every two weeks or more frequently using a spear or handline. Slope and rugosity were both recorded on a five-point scale (supplementary materials section C).

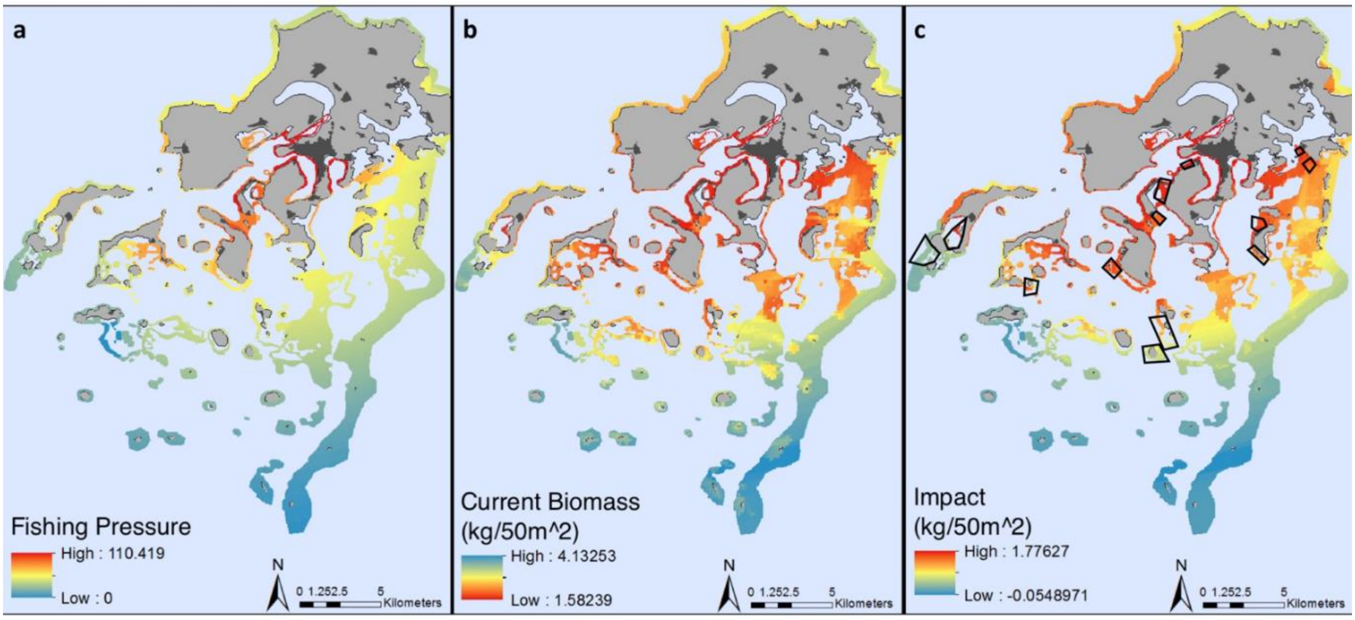


Figure 7.3. Vava'u fishing pressure, predicted current biomass, and potential impact. a) fishing pressure in Vava'u defined as the number of fishers capable of fishing an area fortnightly or more frequently; b) predicted current biomass of target species per 50 m grid cell; and c) potential impact, or change in target species biomass per 50 m cell, following the implementation of a no-take MPA. The black lines indicate the configuration of the 13 recently implemented MPAs.

The predicted total recovery of target species biomass (Figure 7.3c) across the 13 community-based MPAs was 84% of the systematic configuration with the greatest recovery potential (Figure 7.4). The second systematic configuration, which preferentially selected grid cells with high total biomass when differences in impact were minimal, achieved 8.8% greater total biomass than the first systematic configuration while only reducing predicted recovery by 2.3%. The systematic approach targeting high-impact areas focused protection on the central region of Vava'u where fishing pressure was highest (Figure 7.5a). The plateau of fishing pressure's effect on biomass corresponded spatially to the inner island group of Vava'u (Figure 7.5b). Within this region the second systematic configuration targeted areas with high-quality habitat and greater wave energy, and not those with the greatest fishing pressure (Figure 7.5c).

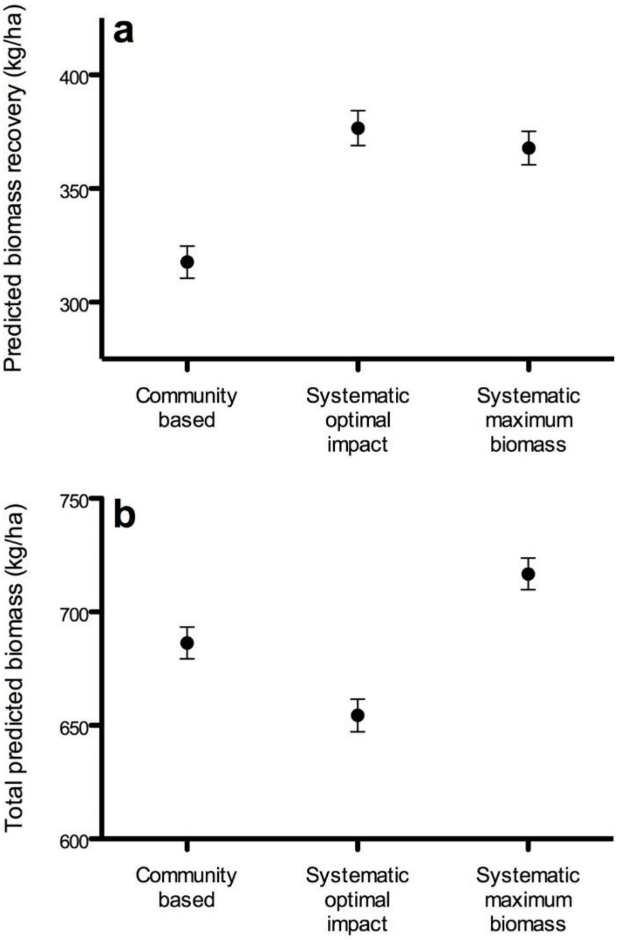


Figure 7.4 (above). Numerical comparisons of a community-based and two systematic MPA configurations. a) predicted impact as recovery of biomass; b) total predicted biomass. Error bars represent the 95% confidence intervals estimated from taking 1000 bootstrap samples of the input data, but selected randomly with replacement, and repeating the analysis for each sample.

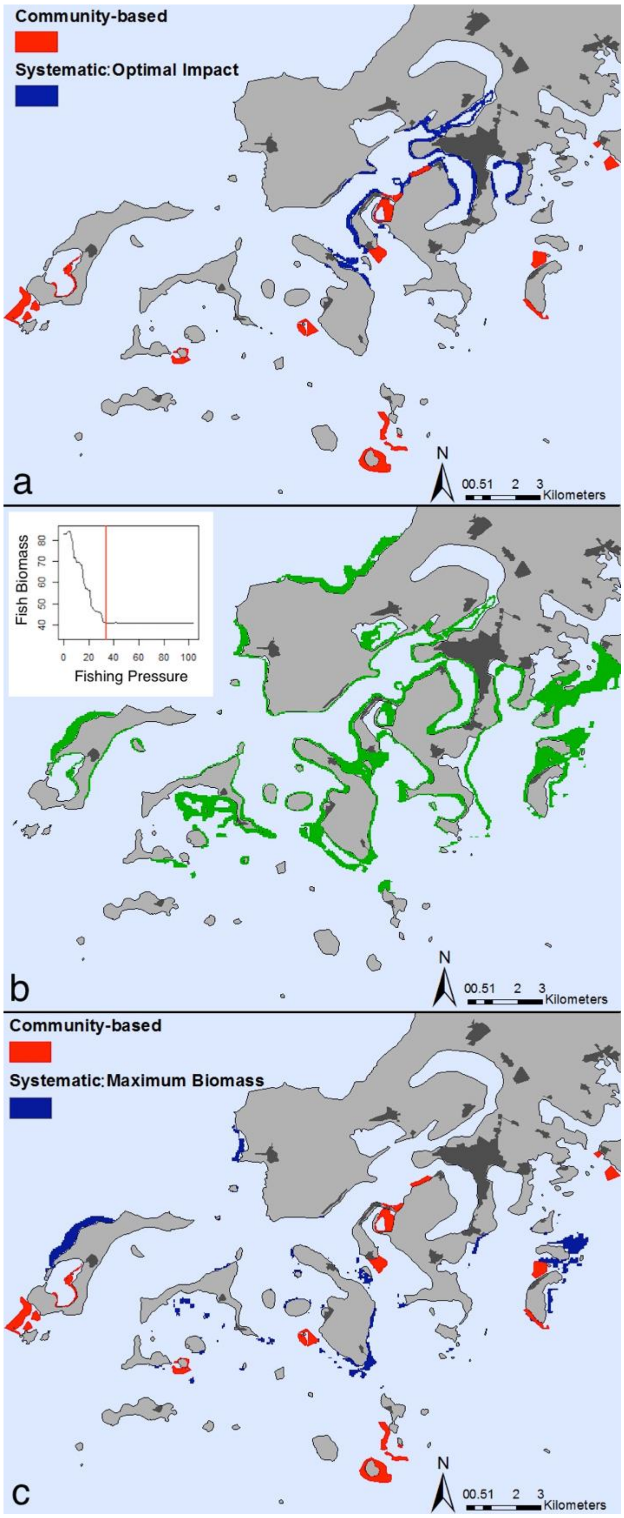


Figure 7.5 (right). Spatial comparison of community-based and alternative configurations of no-take MPAs. a) the current community-based configuration compared to a systematic configuration aimed at maximizing potential impact; b) the region in which fishing pressure's effect on fish biomass plateaus; within this region, alternative reserve configurations would have marginal differences in predicted recovery; c) the current community-based configuration compared to a second systematic configuration that maximizes total biomass when differences to potential impact are minimal.

7.5 Discussion

Our results indicate that local fishing pressure is reducing the biomass of target species close to villages, with fishing pressure accounting for ~24% of the variation in fish biomass. This suggests that community-managed no-take MPAs could have positive impacts on fish stocks, particularly in areas of high fishing pressure. The predicted impact of the community-based configuration of no-take reserves was 84% of the impact of the best-case systematic configuration. This result is important because it suggests that close-to-ideal benefits of MPAs can be achieved in situations where there is relatively little data for systematic placement of reserves or social/political constraints on applying systematic approaches.

This study confirms that fishing pressure can be a strong predictor of target species biomass. Other ecological metrics such as size distributions and community structure have also been demonstrated to vary along gradients of fishing pressure (Graham et al. 2005; Wilson et al. 2010). However, despite the high variance explained by fishing pressure, the model's predictive power decreased in areas of high fishing pressure. This result is indicative of the potential depletion or collapse of the inshore reef fishery in Vava'u. This is further corroborated by the ecological surveys, in which we observed that most of the inner island sites had small sizes and low abundances of reef fish.

Studies assessing the community-based approach to establishing MPAs have generally used habitat representation, and generally concluded that the resulting configurations of MPAs failed to reach 50% of their total capacity (Hansen et al. 2011; Mills et al. 2012; Horigue et al. 2015). However, by using predicted impact on target species as a metric of potential success, our results indicate the benefits could be much greater. The high impact predicted by our results is attributable to community-based MPAs generally being established close to villages where fishing pressure is likely to have been high. In contrast, systematic designs based on habitat representation are likely to include areas that are subject to little or no fishing pressure.

MPAs are often situated next to villages for social reasons, as a way to support local enforcement and maximize compliance (Cinner and Aswani 2007). While social and ecological strategies are not always aligned (Gaymer et al. 2014), the high potential impact of implementing reserves near villages in this study illustrates how ecological benefits can be achieved by emphasising social priorities. The systematic approach to reserve design is also not always feasible, especially in resource-limited nations, and a community-driven approach can therefore often be the most viable solution for marine management in the absence of well-supported centralized management (Ban et al. 2011). High compliance and marine stewardship by local communities are also critical to the success of MPAs (Mascia 2010), and the greater support of community-driven projects could potentially offset the difference in predicted impact between the systematic and community approach.

Furthermore, in practice, centralized planning is frequently not systematic, often resulting in residual MPAs situated to have minimum conflict with human activities and therefore low impact (Devillers et al. 2015).

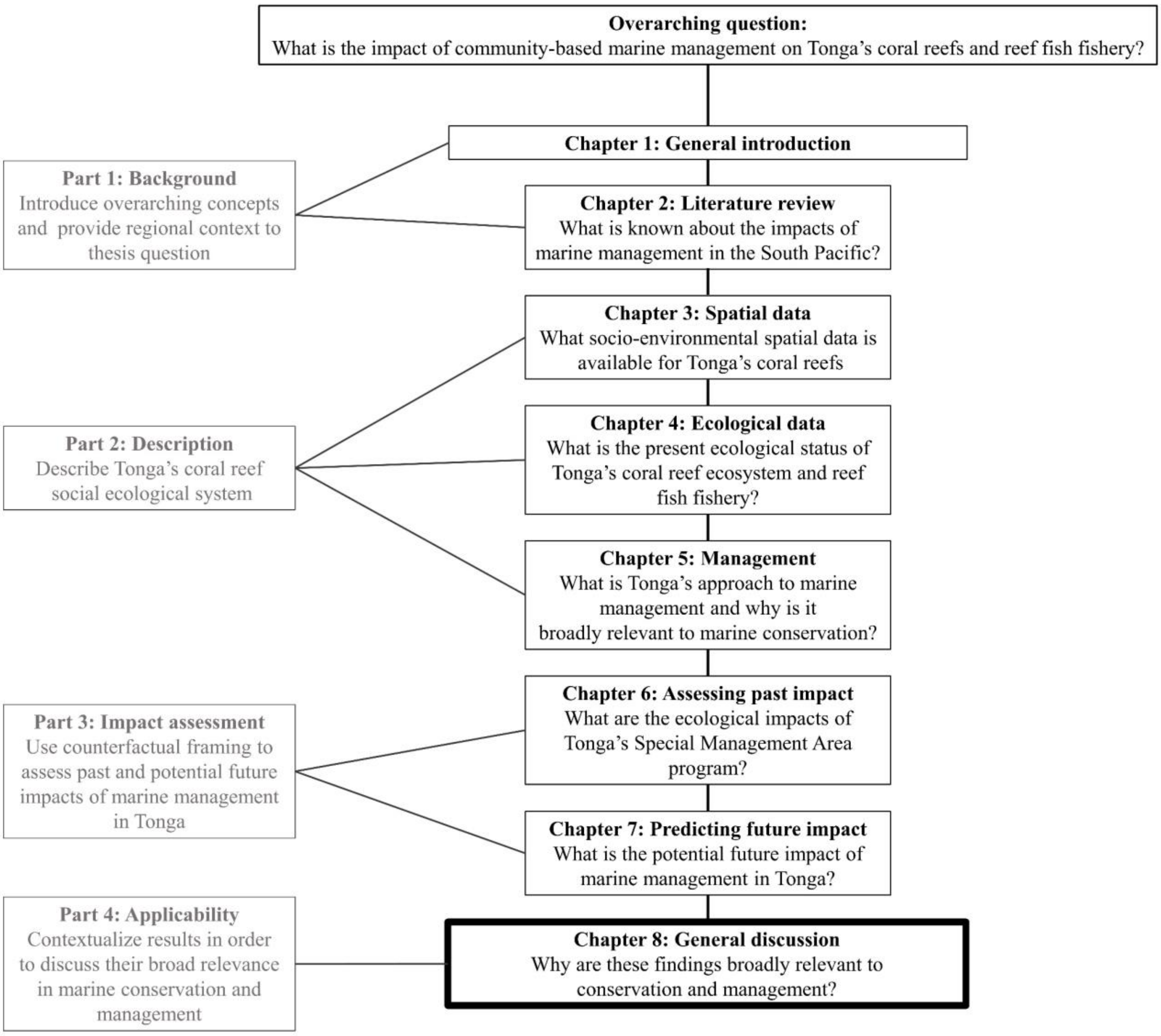
The similarity in potential impact between alternate MPA configurations suggest that within this system there can be flexibility when selecting areas using the predictive impact approach. Although the systematic configuration suggested placing reserves around the most populated part of the region, this might not be practical because compliance and enforcement around urban centres could be difficult. Our results demonstrate that alternative configurations can maintain high impact while also maximizing total biomass. This flexibility enables this approach to be incorporated into future management decisions both in Tonga and other small-island developing nations.

Given Tonga's remoteness, the net rate of stock depletion will likely remain constant following reserve establishment, potentially resulting in a fishery squeeze effect whereby fishing pressure is displaced rather than reduced (Halpern et al. 2004; Agardy et al. 2011). Although this study did not explicitly examine the potential loss of biomass in the absence of protection, this limitation was partially offset by factoring maximum biomass into the configuration as well as impact. Depletion of fish stocks might be exacerbated initially as fishers move to less harvested areas, with long-term benefits accruing only when MPAs build up standing populations of large, spawning fish (Agardy et al. 2011; Hopf et al. 2016). In addition, changes in fish biomass are not always predictable and the impact of no-take reserves on fish stocks can be limited by large-scale chronic impacts such as habitat degradation, pollution, and climate change (Green et al. 2014). However, many of these caveats are not isolated to our predictive method, but are limitations of MPA design in general. Various additional management strategies such as size limits and gear restrictions can be employed to help mitigate these impacts outside of existing MPAs (Lindquist and Granek 2005; Weeks et al. 2016).

Our model allowed us to estimate, based on local environmental parameters and changes to fishing pressure, a hypothetical carrying capacity representing the biomass an area might reach with the implementation of a well-managed no-take reserve after sufficient time has passed for fish stocks to recover. The time required for the biomass ceiling to be reached is beyond the scope of this study, encompassing many aspects of reef ecology. There is also a myriad of additional ecological factors that will affect the carrying capacity of a site, so our results are only indicative of which sites could have the greatest potential impact. Other factors such as coral cover, frequency of disturbances, and larval transport will also be important in establishing the final carrying capacity of each site (Jones et al. 2004; Hopf et al. 2016). In addition, other conservation targets such as fisheries yield are also important for fisheries management and could also be incorporated into estimates of the efficacy of alternative management strategies (McClanahan 2018).

There are various other approaches to the design and implementation of MPAs, each with their own merits and shortcomings (Botsford et al. 2003; Gaines et al. 2010). Our technique can be added to the existing toolset of marine conservation planners to highlight regions in which efforts should be focused and additional methods employed. While habitat representation is not a panacea to reserve design (Pressey et al. 2017), there are still significant ecological benefits to be accrued by protecting a range of habitats and conservation targets (Ward et al. 1999; Airame et al. 2003). Importantly these two approaches are not mutually exclusive, and future management should aim to incorporate both in conjunction when formulating decisions.

Chapter 8: General discussion



Effective management is critical in the global efforts to conserve coral reef social-ecological systems (SESs). Given both the limited resources and large scale of the crisis, management efforts should focus on pursuing actions that maximize impact, or the difference that management makes. However, determining which actions are most effective remains a fundamental challenge in conservation because questions are rarely framed to assess impact. This thesis outlined a clear and methodical approach by which to examine both existing and potential future ecological impacts of community-based coral reef management.

First, I demonstrated that, in the South Pacific, few studies of MPA efficacy have thus far incorporated counterfactual framing or robust impact evaluation techniques into their research designs (**Chapter 2**). In order to fill this gap, three chapters were used to build a national dataset on Tonga's coral reef SES. This involved developing a large dataset of socio-environmental variables that are relevant to Tonga's coral reefs (**Chapter 3**), mapping the current ecological status of Tonga's coral reefs and reef fish resources for the first time (**Chapter 4**), and providing a detailed description, developed from three years of collaboration with the Tongan Ministry of Fisheries, of the Tongan Special Management Area (SMA) program (**Chapter 5**). Using this information, I then conducted a formal impact evaluation to assess the difference the SMA program has made over and above estimated counterfactual conditions of their coral reefs and reef fish fishery (**Chapter 6**). Finally, I also developed a predictive approach to look ahead and assess the potential impact of the program, and then used this approach to compare both community-based and centralised approaches to management (**Chapter 7**). This thesis demonstrates that Tonga's Special Management Area (SMA) program has achieved positive impact for coral reef conservation and natural resource management, and has been able to avoid problems that plague many marine protected area (MPA) systems.

Rather than repeating discussion points from individual chapters, this section focuses primarily on building from each chapter's conclusions to target links between chapters and provide suggestions for developing research into conservation impact and community management. In the sections that follow I i) briefly discuss three core findings from this thesis and ii) suggest directions that the field of impact evaluation in conservation should be taken, with particular emphasis on coral reef SESs and management. These topics are followed by a short concluding remark.

Core findings

The first core finding of this thesis is that community-based marine management can achieve positive ecological impacts. Furthermore, an important mechanism driving this result is the non-residual nature of local management. While many studies exist that examine the effectiveness of MPAs and management, very few explicitly consider the causal links between MPAs and their impacts, and how these depend on socio-economic context (but see Ahmadia et al., 2015; Gill et al., 2017). Generally, studies assessing coral reef MPAs compare MPA and control sites with similar habitats, but there is little evidence of most studies considering any contextual variables beyond habitat. Counterfactual fishing pressure, or the expected fishing pressure if management had not occurred, should be a key consideration when developing impact evaluations because MPAs primarily function by changing patterns of fishing. In several instances, studies have attributed observed differences to management, despite selecting control sites with unequal counterfactual fishing pressure. By incorporating many contextual factors into impact evaluation designs, this thesis demonstrates that, once these variables are considered, it is possible to identify clear positive ecological impacts arising from community-based marine management.

The second core finding is the mechanism by which community-based marine management in Tonga has achieved the aforementioned results. Specifically, implementing no-take MPAs directly adjacent to communities is preventing residual conservation arising from local management in Tonga, and thereby maximizing the differences between reefs within and beyond MPA boundaries. Both chapters five and seven demonstrate that Fish Habitat Reserves (FHRs) are configured close to each SMA community. These reefs tend to have higher value in terms of accessibility, being close to villages and land, in areas of high historic fishing pressure and low wave energy. While these tendencies reflect decisions to enable enforcement and monitoring by communities, they have the benefit of providing a clear solution to the common problem of residual reservation. In Tonga it would not be possible to implement large, offshore community-based MPAs because communities do not have the mechanisms to manage these environments. Residual MPAs are generally driven by governing bodies aiming to achieve targets while minimizing opportunity costs (Adams et al., 2010; Devillers et al., 2015). Local management can therefore solve this fundamental problem, first by removing the perverse incentives that are placed on governments and second by involving the groups most likely to directly benefit from management.

Lastly, scaling local conservation programs to a national level requires identifying the best incentives by which to engage individuals in conservation (Mills et al. 2019). By using the incentive of exclusive access to community waters in exchange for implementing no-take MPAs, the SMA program has successfully incentivized communities and rapidly expanded its reach. This approach has been successful because the loss to fishing grounds that local groups would face from implementing a no-take MPA is offset by the benefit of exclusive access to other areas. While many people might value nature intrinsically, individuals must often act in their own self-interest at the cost of nature, particularly in regions with few economic choices (Hutton & Leader-williams, 2014). However, the SMA program has successfully demonstrated that it is possible to create policy interventions that align short-term self-interests with nature conservation. Rather than implementing policies that force conservation at the expense of communities, identifying policies that enable individuals to benefit from actions that also conserve nature should be a key priority in conservation research.

Future directions and limitations

This thesis has laid a foundation for framing marine management in terms of impact, which, given that conservationists and managers by definition seek to enact change, is hence the key metric that should be described (Ferraro, 2009). All other measurements of management efficacy can be considered proxies, and many of them poor, for the difference that has been achieved (Pressey et al. 2017). The field of conservation science should therefore actively seek methods by which to i) accurately determine the impact of existing actions and ii) maximize the impacts of potential future actions (Ferraro and Pressey, 2015; Pressey et al. 2015). Since management is fundamentally about changing human behaviour, the most direct way to maximize impact is therefore to capitalize on these changes. Indeed, effective, or impactful, conservation could even be defined as actions leading to the greatest changes in human behaviour for the conservation of nature that are deemed socially acceptable. This paradigm goes against the prevailing expectation in conservation planning of minimizing opportunity costs, instead suggesting that maximizing opportunity costs is an approach more likely to achieve high impact. At the outset it is important to clarify that this approach does not imply the mass displacement of society, such as turning all cities into parks, which would, by the definition used here, maximize impact (disregarding displacement of human pressures elsewhere). Rather, it suggests that, by framing questions in this context, we will more accurately understand the differences we make, while acknowledging the importance of balancing both human society and nature.

When framed in this context, the problem of residual conservation becomes much more apparent because, until changes in human behaviour have occurred, no impact is achieved. Residual MPAs are the result of configurations that minimize changes in present day actions (Devillers et al., 2015). Therefore, the only chance of achieving impact from a residual MPA is by its potential effects on future human actions if available resources outside managed areas become depleted. This is problematic in two ways: first, it lets business as usual continue and passes the requirement to change on to future generations; and second, there is always the possibility that future management practices could change, such as degazettement of MPAs once future societies require the resources within

(Mascia & Pailler, 2011). By using impact to frame conservation actions, it will be increasingly difficult for management authorities to suggest any benefits from protected areas that are residual (Pressey et al. 2015).

Formal impact evaluations that explicitly consider contextual variables should also increasingly replace traditional Control-Intervention or Before-After-Control-Intervention approaches that do not consider confounding variables (Adams et al. 2019; Frazer et al. 2019). However, acknowledging the potentially prohibitive costs and/or training associated with impact evaluations, an important starting point is to clearly consider context (McIntosh et al., 2017). This can be achieved by first describing potential confounding variables that could bias comparisons, and then identifying comparative areas that are likely to be most ecologically and socioeconomically similar to a predicted counterfactual state (Ferraro 2009). Care must be taken to manage two types of confounders: those that influence outcome variables (e.g. effects on target species biomass) and those that bias the configuration of MPAs (e.g. residual locations with low inherent fishing pressure) (Adams et al. 2019). Lastly, explicit discussions of any potential confounders and justification for site selection should be routine in any evaluation of marine management efficacy as well as the resulting publications.

Jupiter et al. (2014) outlined six strategies that could be used to manage marine resources in the community context. While there have been various approaches used to assess their efficacy, and the benefits and caveats of each have been discussed, they have yet to be compared through the lens of impact. An overall assumption for comparing marine management strategies using impact will be to assess how they change or reduce the net fishing pressure across the system. A key caveat of no-take MPAs in this context is that they might not reduce total fishing pressure, but only shift it. Likewise, access restrictions only change who fishes, not necessarily the volume harvested, and periodic closures might change only the timing of harvest events. Comparing the effects of management strategies on the entire system using changes in fishing pressure can provide novel insights into actions or combinations of actions that can be applied to maximize impact.

In general, local management is often viewed as a method by which developing countries can enable marine management when there is limited capacity or resources for centralized governments to do so (Ban et al., 2011; Gaymer et al., 2014; Mills et al., 2012). While the primary method by which to solve residual biases from centralized management is to ensure governments and managers are responsive to the concept of impact, developed countries could also gain additional benefits from local management. Residual conservation arises primarily from centralized governing bodies trying to balance multiple pressures, with extractive pressures generally dominating the need for MPAs with real ecological impact. Therefore, giving at least partial control of decisions to local groups, regardless of economic status, could help improve the impact of many protected areas. Local management can also generate a sense of ownership and pride in local resources, which is more likely to generate support and involvement in the conservation process (Gurney et al., 2014). While determining the best way to scale local conservation initiatives in more developed regions, particularly those with high population densities, remains a challenge, doing so will likely improve the long-term efficacy of their management systems.

Predictive approaches to conservation planning that incorporate impact are still in their infancy (Law et al., 2017; Pressey et al., 2017) . Few other studies have thus far used predictive tools to assess impact (but see Fulton et al. 2015; Harborne et al. 2018; Sacre, 2019; Visconti et al. 2015), instead focusing on methods such as habitat representation. While the long-term changes expected from management might be more difficult to quantify than known metrics such as the current locations of species, habitats and bioregions, an impact-directed approach will be crucial to developing meaningful conservation actions, and ultimately making a difference (Pressey et al. 2017). Incorporating, and accepting, uncertainty in our estimates of potential impact can be viewed as a caveat of this approach, particularly when metrics not necessarily related to impact can be measured accurately (Pressey et al., 2017). However, developing reliable techniques that can manage the uncertainty inherent in conservation will, in the long-term, produce greater positive impacts than substituting them for unreliable metrics that might bear no relationship to change (Sacre et al. 2019a; 2019b).

While chapter 7 developed a model that incorporated impact into planning MPA configurations, there are several ways that this model could be improved. First, predicting impact not only involves quantifying potential recovery within no-take MPAs, but also ongoing loss outside MPAs, particularly if net fishing pressure shifts instead of being reduced (Sacre, 2019). Chapter 7 therefore considered only half the story – potential recovery - and future research should investigate potential timescales of decline from fishing effort outside MPAs in order to combine both components of impact (but see Sacre, 2019). Likewise, we were not able to account for differences in ecosystem health, whereby some areas might be too depleted to enable recovery, despite being non-residual (Cinner et al., 2018). Incorporating metrics of ecosystem health and the trajectory of recovery into predictive models should therefore also be a priority. In addition, it should be understood that, ultimately, spatial prioritization tools are for decision support and not decision making, which requires human experience and the consideration of many more criteria than are typically modelled in software systems (McClanahan et al., 2016).

This thesis did not demonstrate any positive ecological impacts from the SMA areas in Tonga, where communities can still fish. One caveat to this approach is that we were not able to incorporate actual catch data into the study design. While there was no difference in the state of the reef fish fishery between SMA and open areas, there could have been differences in catch. For example, catch inside the SMA area might have been greater than in open areas, which would be a positive impact of the program. Likewise, over time, if catch within the SMAs remains stable but declines beyond their boundaries, this would also constitute positive impacts for the program. Lastly, if the benefits of the fishery are better controlled by local communities then despite decline this may be seen as a positive impact. While SMA communities are meant to collect catch data and deliver it to the Ministry of Fisheries, during the time of this thesis, these data were either not available or of questionable quality. Therefore, future studies examining the impacts of Tonga's SMA program should focus on combining ecological data with socioeconomic data explicitly focused on catch.

A final caveat of this thesis and of management in general is that, regardless of management impact, many threats to coral reefs are of a scale and severity that local actions, even scaled to a regional level, are failing to prevent ecosystem decline (Bellwood et al. 2019). Climate change is a global issue that requires the immediate reduction in fossil fuel consumption (Hughes et al., 2017, 2019). Small island nations like Tonga are not only some of the most vulnerable to the effects of climate change, but, given their small populations, some of the least equipped to prevent it, regardless of their climate policies. While various management actions might boost ecosystem resilience (Game et al. 2009; McCleod et al. 2009), or improve food security (Mascia et al., 2010), in the long-term these ecosystems could continue to decline unless more populated countries act now to reduce their carbon footprint. This caveat remains a key topic of global research.

Concluding remarks

Overall, the broad implication of this thesis is that local management matters, and can drive positive differences towards nature conservation and natural resource management. We are often faced with trade-offs between conservation and human need, but these two concepts need not be dichotomous. Rather, there are techniques that can be used to promote simultaneous benefits to both. Developing existing, and discovering new, solutions that are able to strike this balance are key to progressing the field of conservation in the 21st century.

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Dataset methodology

Environmental variables

Bathymetry

Depth is both a crucial determinant of marine community structure (Huston 1985, Brokovich et al. 2008), as well as a mitigation factor from anthropogenic activities (Bridge et al. 2013). The bathymetric profile of Tonga was therefore included as a layer in this dataset. Data describing bathymetry between 0 and -20 m at a resolution of 2 m^2 was obtained from Land Information New Zealand (LINZ) for all island groups of Tonga (Hartmann *et al.* 2018). For the island groups of Vava'u and Ha'apai (excluding the Nomuka group) deeper bathymetric data (0 to -60 m) was available from the Khaled bin Sultan Living Ocean Foundation Global Reef Expedition (KSLOF-GRE, Purkis *et al.* 2019) and was therefore used in preference for these areas.

Bathymetry data created by Purkis *et al.* (2019) was derived via spectral derivation of water depth from WorldView-2 (WV2) satellite imagery. Authors used empirical algorithms described by Stumpf et al. (2003) and Kerr and Purkis (2018) to extract bathymetry data from multispectral WV2 imagery and followed methodology by Kerr and Purkis (2018) to map water depth (see Purkis *et al.* 2019 for more details). For full details of LINZ methods see Hartmann *et al.* (2019).

Original layers from both Linz and Purkis *et al.* (2019) were combined and the resolution reduced to 10 m^2 to limit file size. Pixel resolution reduction was completed using the *Resample* tool with the *cubic* function, before applying a smoother to reduce the effects of rogue pixels.

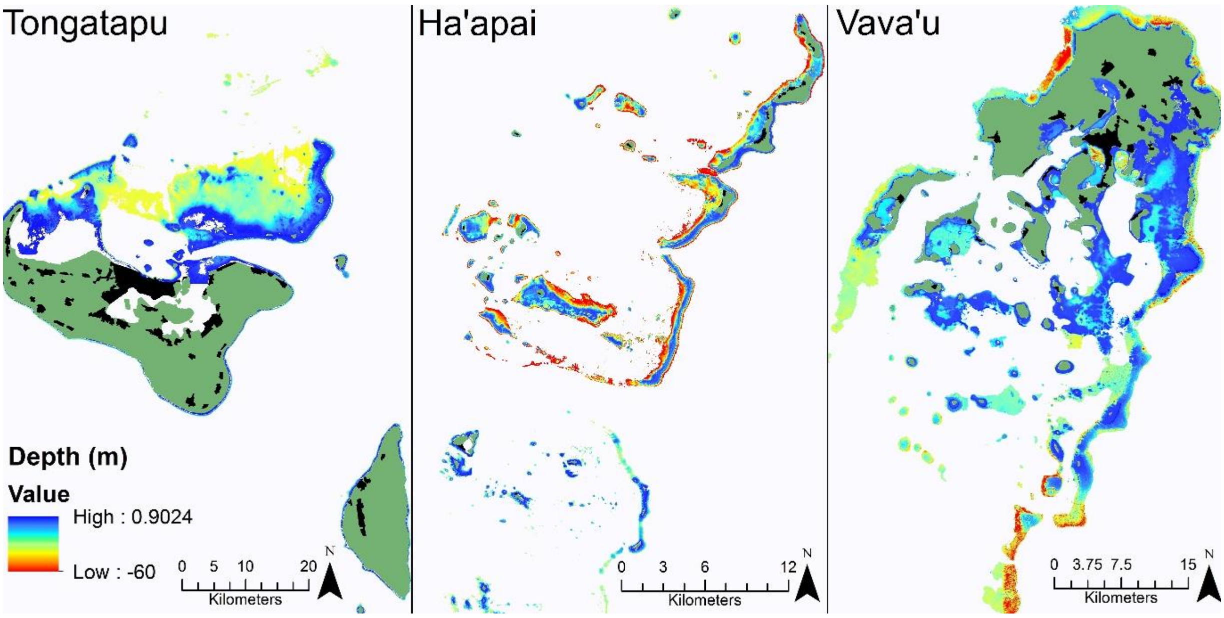


Figure S1. Bathymetric profile of Tonga's shallow water marine environment. Satellite derived bathymetry (SDB) data for Vava'u and Northern Ha'apai were collected by the Khaled bin Sultan Living Ocean Foundation (KSLOF) from 0 – 60 m. Satellite derived bathymetry data for Tongatapu and southern Ha'apai (Nomuka group) were collected by Land Information New Zealand (LINZ) from 0 – 20 m. Green areas represent land and black areas represent villages.

Coral reef density

Coral reef density was calculated as the total area (m^2) of coral reef habitat within a radius from each 10 m^2 pixel defined by a buffer distance of both 5 and 15 km. These distances were selected because they represent the lower and upper range of larval dispersal distances for most reef fish (Green *et al*. 2015) (Fig. S2). Coral reef habitat classification by Purkis et al. (2019) consisted of 36 habitat classes at a resolution of 2 m^2 but was not available for the island groups of Tongatapu or Nomuka (within Ha'apai). For these island groups habitat classification by Andrefouet *et al.* (2006) was used (24 classes, 30 m^2 resolution). While determining the most accurate degree of connectivity between reefs in Tonga will depend on both biophysical modeling of dispersal patterns and genetic parentage analysis, it was beyond the scope of this study to complete a comprehensive assessment of connectivity at this level within Tonga's >15,000 km^2 of reef habitat (Bode *et al.* 2019). These reef density layers therefore represent a first approximation of potential patterns of connectivity.

Reef habitat for Vava'u and Ha'apai was defined from Purkis *et al.* (2019) habitat classification and included the following habitats: shallow fore reef terrace, shallow fore reef slope, reef crest, lagoon pinnacle reefs (massive coral dominated and calcareous red algae conglomerate), lagoon floor bommies, lagoon patch reefs, lagoon fringing reefs, deep forereef slope, back reef pavement, back reef coral framework, and back reef coral bommies. Reef habitat for Tongatapu and Nomuka was defined from Andrefouet *et al.* (2006) habitat classification, and included the following habitats: subtidal reef flat, shallow terrace with constructions, reef flat, forereef on terrace, and fore reef. A raster layer with all included reef layers was generated by assigning a value of 1 to each 10 m^2 pixel containing reef habitat, and a value of 0 for pixels containing non-reef habitat. The *focal statistic* tool was then used to calculate the sum of the number of pixels within a 5 or 15 km radius of each 10 m^2 pixel of reef area in Tonga. The resulting value was then converted to units of m^2.

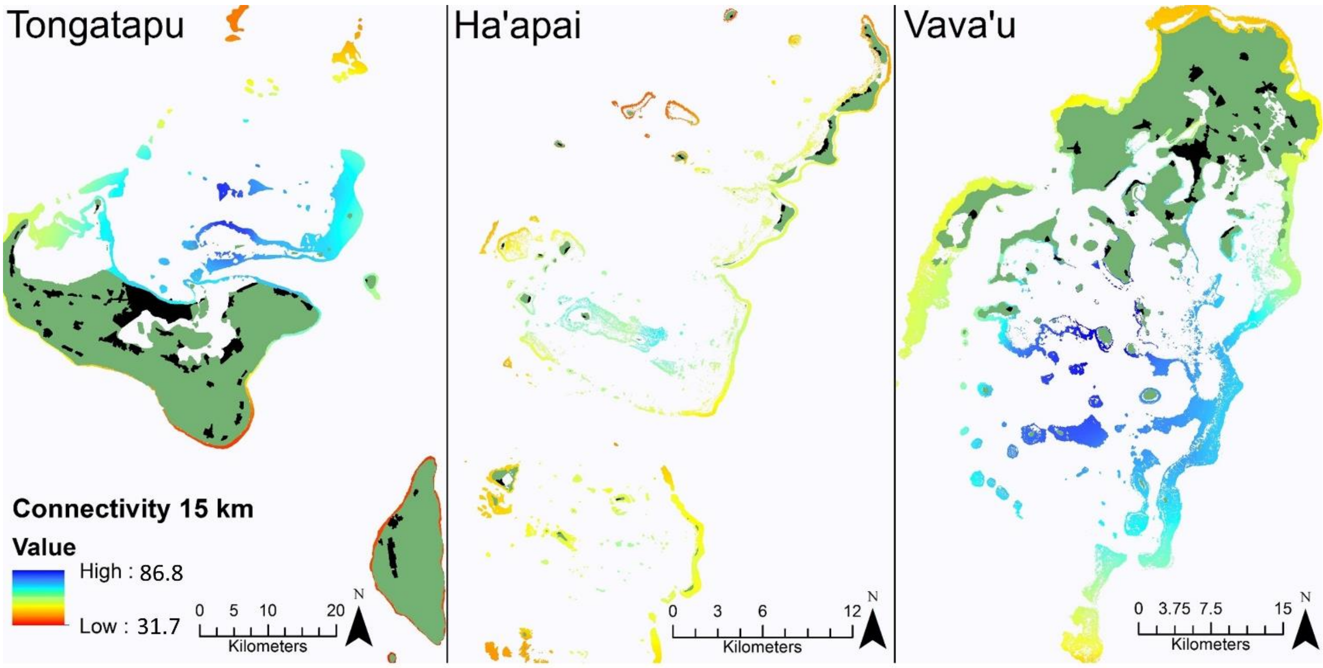


Figure S2. Coral reef density in Tonga measured as the amount of reef habitat in within a 15 km radius of each 10 m^2 reef pixel. An additional layer with a 5 km buffer is also provided in the online data source. Green areas represent land and black areas represent villages.

Distance to deep water

Differences in coral reef community structure can be driven not only by depth, but also the overall depth of the reef system in question (Bak 1977). For example, benthic and reef fish communities at a depth of 4 m in a shallow lagoon may be remarkably different than at the same depth on a deep wall system. Two spatial layers were therefore created describing the distance of each pixel to the 10 and 20 m depth contours respectively.

First, to minimize the influence of erroneous pixels, the *resample* function (ArcMap V10.4.1) was used to resize bathymetry layers to a resolution of 10 m^2 using the cubic function. A *smoothing* filter was then twice applied twice to further minimize the influence of erroneous pixels on the dataset. The *raster calculator* and *extract by attribute* functions were used to split the bathymetry layers into two layers corresponding to all values shallower and deeper than the specified depth (10 or 20 m). The *Euclidean distance* tool was then used to calculate the distance to the 10 and 20 m depth contour for each pixel shallower than the specified depth. All pixels deeper than the specified depth were designated a value of zero. Lastly, the two resulting layers were merged using the *mosaic to new raster* function. This resulted in a continuous layer with a value of the distance to each depth contour for all pixels shallower than the specified depth, and a value of zero for all pixels deeper than the specified depth.

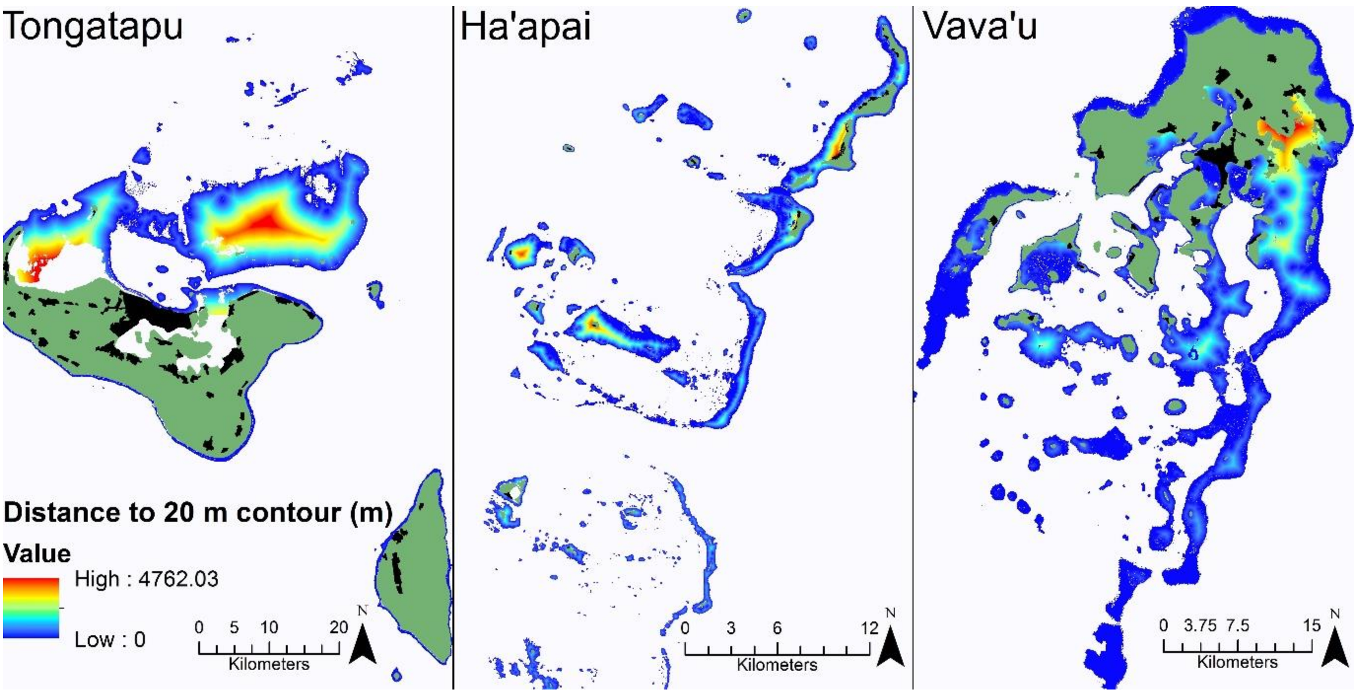


Figure S3. Distance to the 20 m depth contour. Distance to the 10 m depth contour is also provided as an additional layer. Green areas represent land and black areas represent villages.

Distance from land

Distance from land may be an ecologically relevant variable for both environmental and anthropogenic reasons. Firstly, environmental factors such as terrestrial runoff are important factors in marine processes (Fabricius 2005). In addition, anthropogenic factors may decrease with distance from shore. For example, while most fishing occurs close to villages, fishers in Tonga occasionally set up fishing camps on remote islands. Therefore distance to land, including small islands, could act as a proxy for additional anthropogenic pressures unable to be accounted for by other metrics such as distance from villages or population centres. Distance from land may also be an important consideration for other industries, such as aquaculture, where distance from land may be a more important consideration than distance from village. The distance to the nearest landmass, including small, uninhabited islands was therefore calculated for each 10 m^2 pixel using the *Euclidean distance* function (Fig. S4).

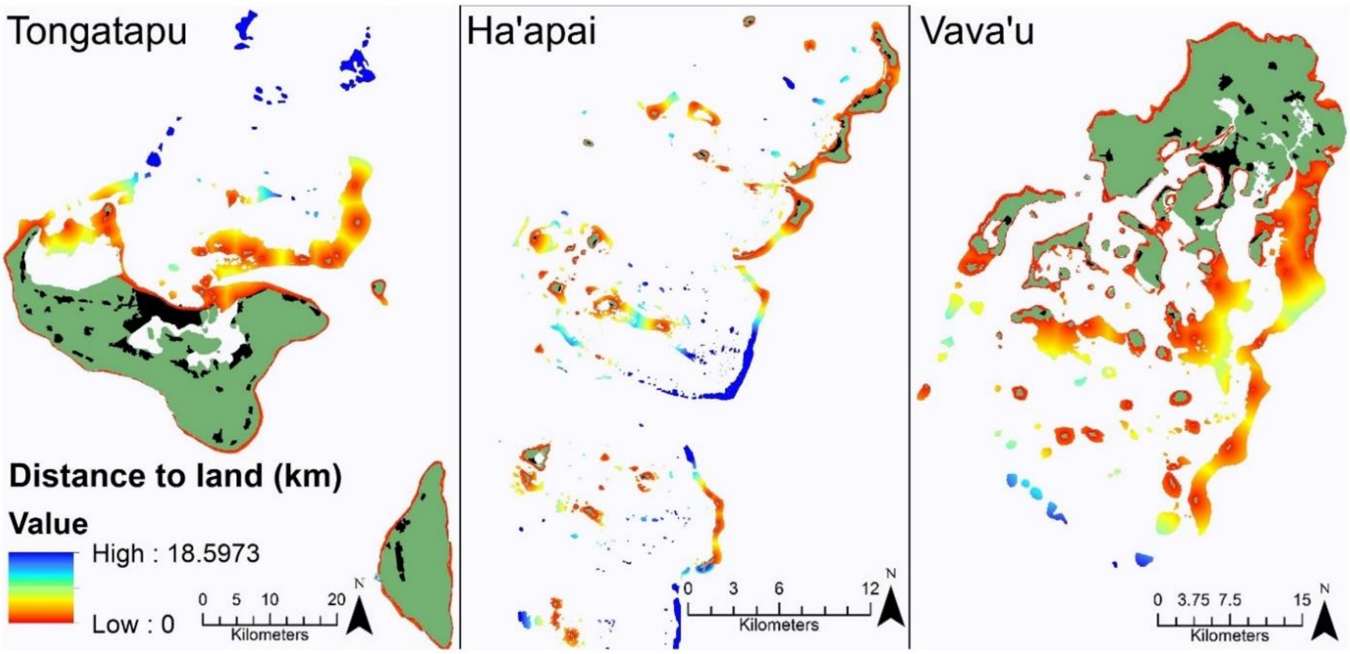


Figure S4. Distance from land for every 10 m^2 pixel of Tonga's near-shore marine environment. Green areas represent land and black areas represent villages.

Distance from major terrestrial inputs

Terrestrial runoff is a well-established stressor to the marine ecosystem, affecting growth, survival, reproduction, recruitment, and species interactions of a variety marine organisms (Fabricius 2005). Nutrient inputs from land derived sources are commonly detectable in primary producers up to 15 km from shore (Lapointe and Clark 1992; Yeager et al. 2017) and terrestrial-derived dissolved organic nutrients may be detectable 50 km or more from the coast (Delvin and Brodie 2005)

There are five major sources of terrestrial inputs in Tonga. Three large lagoon areas with strong tidal flow occur in Vava'u: near the villages of Taoa, Makave and Koloa, respectively. Two occur in Tongatapu: the main lagoon of Fanga'uta and the tidal flat between villages Puke and Ha'atafu. These five locations are the main sources of terrestrial inputs into the marine environment of Tonga, and likely also sources of both pollution and raw effluent (Aholahi *et al.* 2017). Distance to the nearest major terrestrial input sources was therefore calculated for each 10 m^2 pixel using the *Euclidean distance* function (Fig. S5).

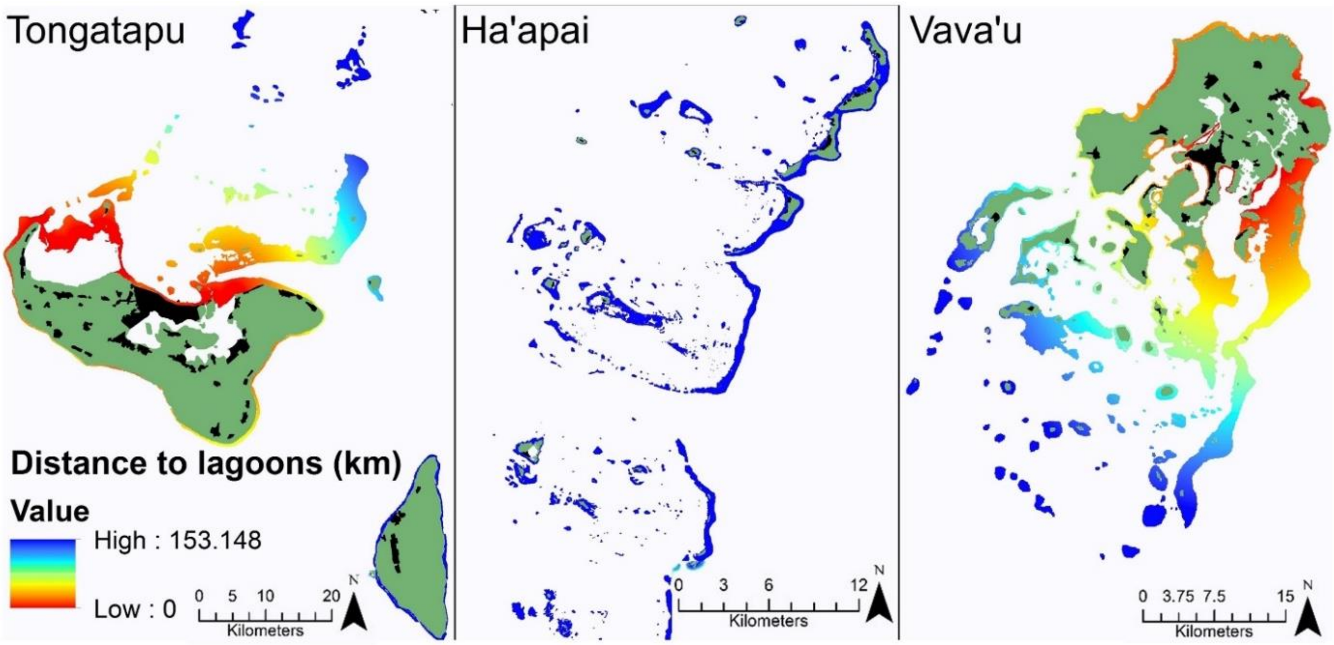


Figure S5. Distance from major terrestrial input sources in meters for every 10 m^2 pixel of Tonga's near-shore marine ecosystem. Green areas represent land and black areas represent villages.

Habitat

Habitat is a crucial determinant of marine ecosystem structure (Coker *et al.* 2014). Marine habitat classification was therefore obtained from Purkis et al. (2019) and Andrefouet *et al.* (2006) (Fig. S6). Purkis et al. (2019) consisted of 36 aggregated map classes at a resolution of 2 m^2 but was not available for the island groups of Tongatapu or Nomuka (within Ha'apai). For these island groups habitat classification by Andrefouet *et al.* (2006) was used (24 classes, 30 m^2 resolution). In addition to the two habitat layers included in this dataset, as of March 2020 the Allen Coral Atlas has also completed habitat maps for Tonga, available to download at: https://www.allencoralatlas.org/atlas

Habitat classification data created by Purkis *et al.* (2019) used eCognition software (v. 5.2, Trimble Inc.) to segment WorldView-2 (WV2) satellite imagery into polygons labelled by zone, structure, and ultimately habitat class. Habitat classification was then calibrated by field observations. Habitat classification by Andrefouet *et al.* (2006) used Landsat 7 ETM+ satellite imagery and habitat classification was determined using image-based criteria to determine geomorphological classes. For a detailed methodology of image acquisition and habitat classification schemes, see Purkis *et al.* (2019) and Andrefouet *et al.* (2006).

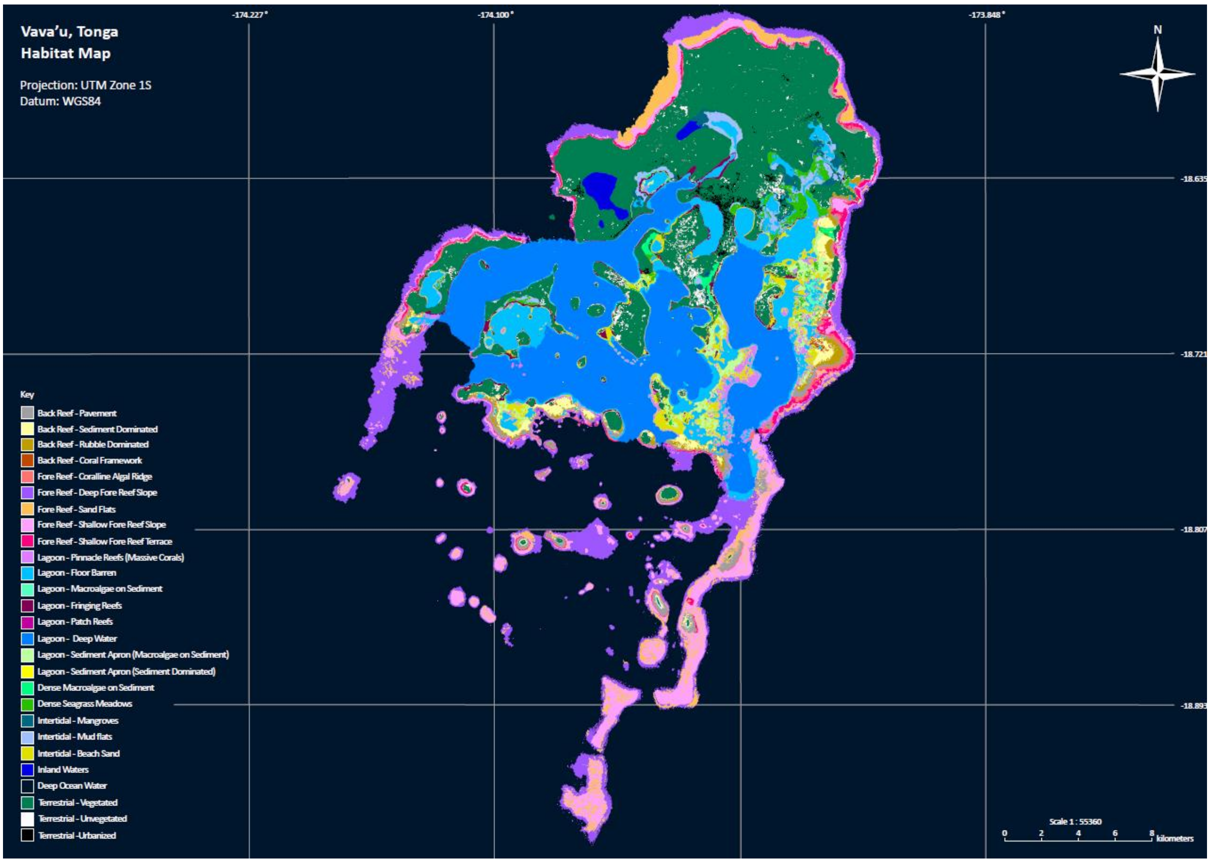


Figure S6. Habitat classification by Purkis *et al.* (2019) for Vava'u. Green areas represent land and black areas represent villages.

Land area

Local marine community structure and productivity may be influenced by terrestrial nutrients and runoff into the marine ecosystem (Fabricius 2005). Total land area, as well as distance from land may therefore also act as a useful metric for the degree of terrestrial influence on near-shore marine ecosystems. The total land area within a 5 and 15 km buffer zone of each 10 m^2 pixel was calculated as an additional proxy for terrestrial influence. Five and 15 km buffers were selected as previous studies found that nutrient inputs from terrestrial sources are commonly detectable in primary producers up to 15 km from shore (Lapointe and Clark 1992). While Yeager et al. (2017) acknowledge that riverine plumes may affect the marine environment up to 50 km from the coast (Delvin and Brodie 2005), in most cases the effects are limited to within ~10 km of shore (Fabricius 2005). A raster layer was generated by assigning values of 1 for all land pixels and values of 0 for all marine pixels. The *focal statistics* tool was then used to calculate the sum of pixel values within a 5 and 15 km radius. Lastly, the *extract by mask* function was used to clip the large resulting layer by the extent of Tonga's near-shore marine ecosystem (Fig. S7).

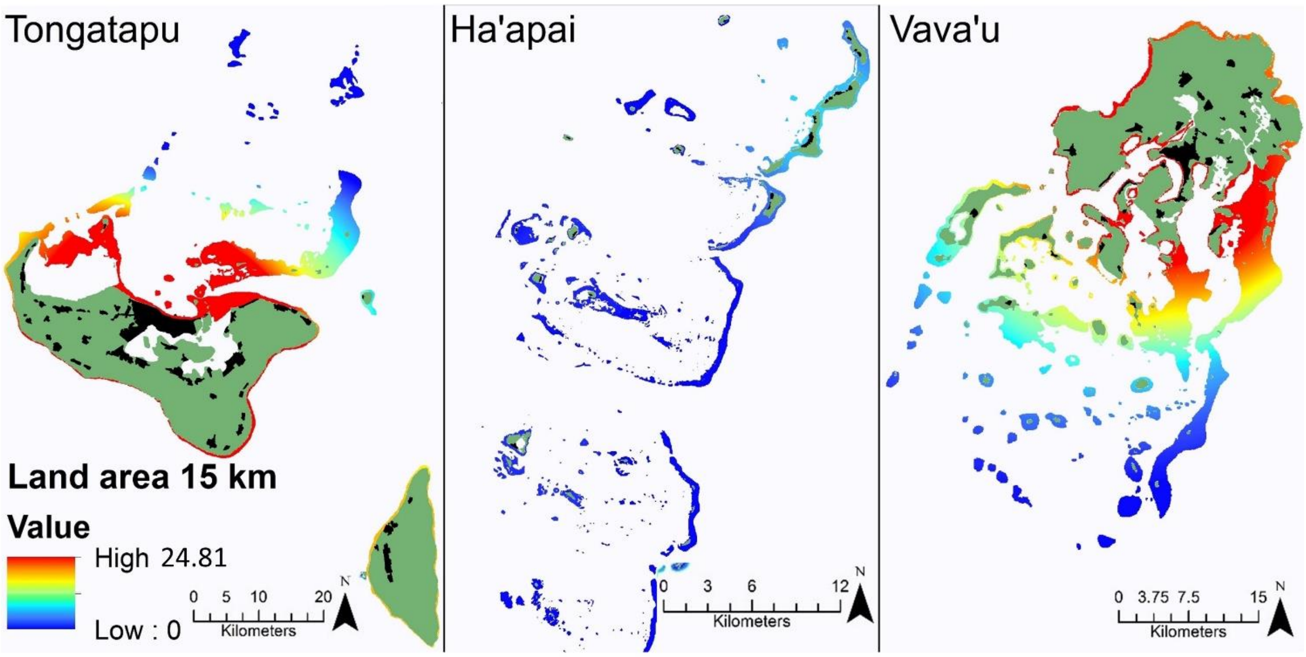


Figure S7. Total land area (km^2) within 15 km of the near-shore marine ecosystem of Tonga. An additional layer with total land area within 5 km is also provided. Green areas represent land and black areas represent villages.

Net primary productivity

Variation in primary productivity can affect the assemblage structure of herbivorous fishes (Mumby et al. 2013) and the total biomass of reef fishes (Williams et al. 2015; Harborne 2016). An oceanic primary productivity layer was therefore extracted from a global layer developed by Yeager *et al.* (2017) to describe the marine ecosystem of Tonga.

Yeager et al. (2017) global layers were developed from 8-day composite layers from 2003-2013 produced by NOAA Coast Watch (http://coastwatch.pfeg.noaa.gov/erddap/griddap/erdPPbfp28day.graph?productivity). The NPP layer was modelled on a 2.5 arcmin grid based on satellite measurements of photosynthetically available radiation (NASA's SeaWiFS), SST (NOAA's National Climactic Data Center Reynolds Optimally-Interpolated SST), and chlorophyll a concentrations (NASA's Aqua MODIS; http://coastwatch.pfeg.noaa.gov/erddap/griddap/erdPPbfp28day.html) (Behrenfeld and Falkowski, 1997). Remotely sensed estimates of productivity over shallow water are confounded by bottom reflectance, so grid cells with a minimum depth of <30 m were filtered out based on the STRM30 plus bathymetry layer (0.5 arcmin resolution, http://topex.ucsd.edu/WWW_html/srtm30_plus.html) following Gove et al. (2013). The values for cells with missing data following filtering were interpolated from the three closest surrounding cells within a 125 km search radius.

The primary productivity of benthic communities can vary at small scales because of differences in wave exposure, light intensity and nutrient concentrations (Harborne 2016). High resolution NPP of reef habitat is therefore not possible from remotely sensed data. However, the Yeager et al. (2017) NPP layer captures larger-scale patterns in productivity across the region and this layer is therefore supplied at a coarse resolution and covers Tonga's nearby oceanic system (approximately 220 km east-west by 330 km north-south (Fig. S8).

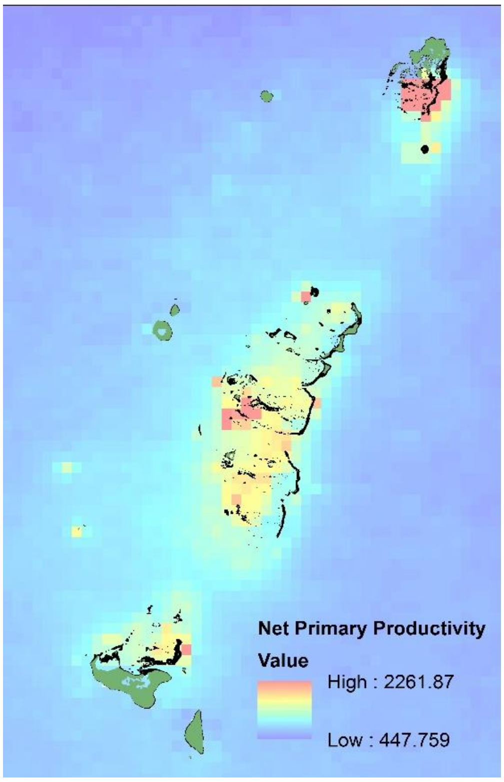


Figure S8. Net Primary Productivity (NPP) of Tonga's marine environment.

Salinity

Environmental fluctuations in salinity strongly affect the physiological functions of marine organisms (e.g. marine bivalves, Lucas 2008) and may structure species assemblages (Barletta et al. 2005). The global layer of sea surface salinity developed by Sbrocco and Barber (2013) shows a small increase in salinity from north to south across Tonga's waters. Despite the minor difference in salinity, this layer was still included to address potential needs of end users. Due to the coarse resolution, the extent of this layer therefore details a broader marine area than previous layers and also includes Tonga's nearby oceanic system (approximately 220 km East-West by 330 km North-South) (Fig. S9).

Measurements of salinity were extracted from the Sbrocco and Barber (2013) global layer of mean sea surface salinity. These values were obtained by Sbrocco and Barber (2013) from in situ oceanographic observations compiled by NOAA's World Ocean Atlas 2009 (WOA09; Antonov et al. 2010). The authors calculated monthly means (measured in practical salinity units) by averaging five "decadal" climatologies at 1 arc-degree resolution for the time periods from 1955 to 2006. These were subsequently smoothed by Sbrocco and Barber (2013) in ArcMap to 30 arc-second grids. The final MARSPEC layer included was the mean annual sea surface salinity in psu at 1 km resolution. Further details can be found in both Sbrocco and Barber (2013) and (Antonov et al. 2010).

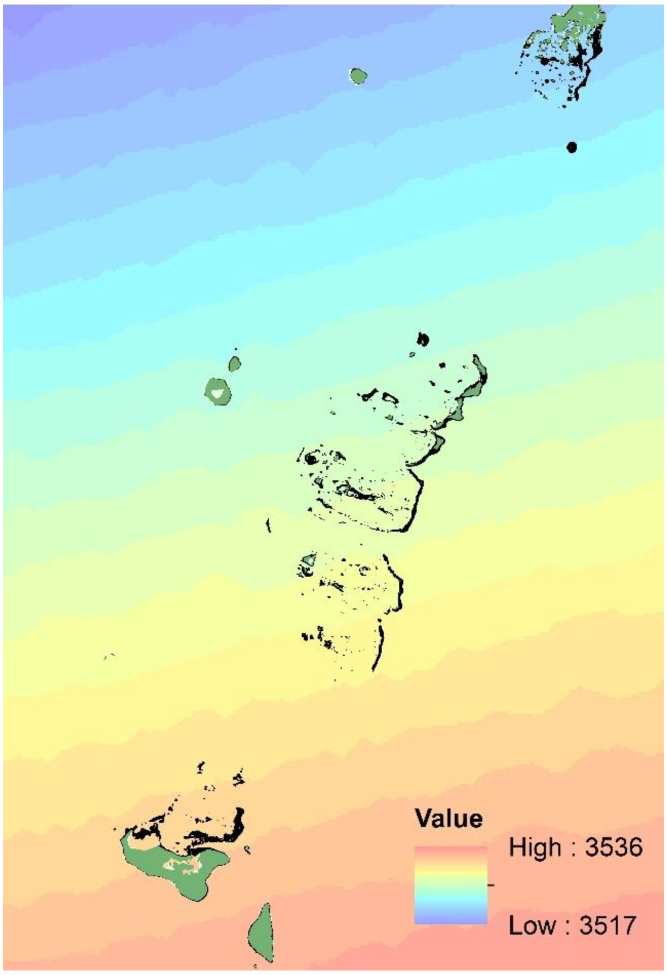


Figure S9. Mean annual salinity of Tonga's marine environment measured in practical salinity units (psu).

Sea surface temperature

Temperature is a primary abiotic factor affecting the physiology of marine organisms (Brett 1971; Harborne 2016), including algal productivity (Hatcher 1990) and thus potentially the demographics of herbivorous fishes (Harborne 2016). The recurrent mass bleaching of coral reefs globally is also directly linked to sea surface temperature (SST) (Hughes et al. 2017). Coral bleaching events are primarily associated with variability in sea surface temperature, and the metric Degree Heating Weeks (DHW) is commonly used as a proxy for heat stress events. Nine SST variability layers are available from the NOAA coral reef watch website (https://coralreefwatch.noaa.gov/product/thermal_history/index.php), including average time between stress events and number of stress events since 1985, at DHW0, DHW4 and DHW8 respectively. However, the resolution of these layers (5 km) was too coarse for use in the current study. DHW4 and DHW8 layers were included in initial models, but were unable to explain patterns of observed coral bleaching. Given the observations of clear recent bleaching events at many sites, we suspect that patterns of bleaching within Tonga may be too fine scale for these layers to be of use. Consequently, general patterns in SST across Tonga were also included in this dataset, with the hypothesis that corals living closer to their thermal threshold may be more likely to have bleached in the past. (Fig. S10).

Mean annual sea surface temperature (SST) was extracted from Sbrocco and Barber (2013) MARSPEC global ocean layers. Sbrocco and Barber (2013)obtained satellite measurements of SST at 2.5 arc-minute resolution (approximately 4 km^2) from Aqua-MODIS 4-micron night-time SST level 3 standard mapped image products, downloaded from NASA's Ocean color website (*http://oceancolor.gsfc.nasa.gov/*). Monthly climatological means from September 2002 to August 2010 were used to calculate mean annual SST. As with NPP and salinity, global layers were clipped by the extent of Tonga's nearshore oceanic environment. Temperature is presented in degrees Celsius.

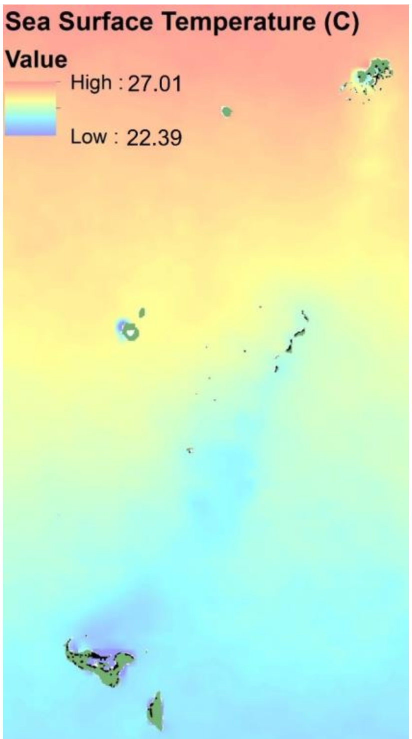


Figure S10. Mean annual sea surface temperature (SST) of Tonga's marine environment in degrees Celsius.

Wave energy

Wave exposure is an important variable structuring coral reef communities (Fulton et al. 2005) and can have significant effects on both fish assemblages and benthic habitat types. Mean wave energy, calculated as joules per square meter, was calculated using the University of Guam Marine Lab (UOGML) Wave Energy Tool (Fig. S11). A detailed description of methodology is provided in Jenness and Houk (2014) and Ekebom et al. (2003). Mean wind speed and direction were calculated from weekly wind speed and direction obtained from QuikSCAT satellite scatterometer data. Land and reef flat habitat layers from Andrefoeut *et al.* (2006) were then used to calculate fetch to the nearest landmass, reef flat or reef crest. Mean wave energy was then calculated using wind speed, direction, fetch and linear wave equations (Ekebom et al. 2003). While this data only accounts for surface wave exposure, it is likely to be a good estimate of the exposure experienced in each cell, since this project is designed for use in shallow-water, near-shore habitats. Due to extended processing times, grid cell size was set to 200 m^2, then outputs smoothed twice using the *filter* function and *resampled* to 10 m with binary weighting to produce a 10 m^2 resolution.

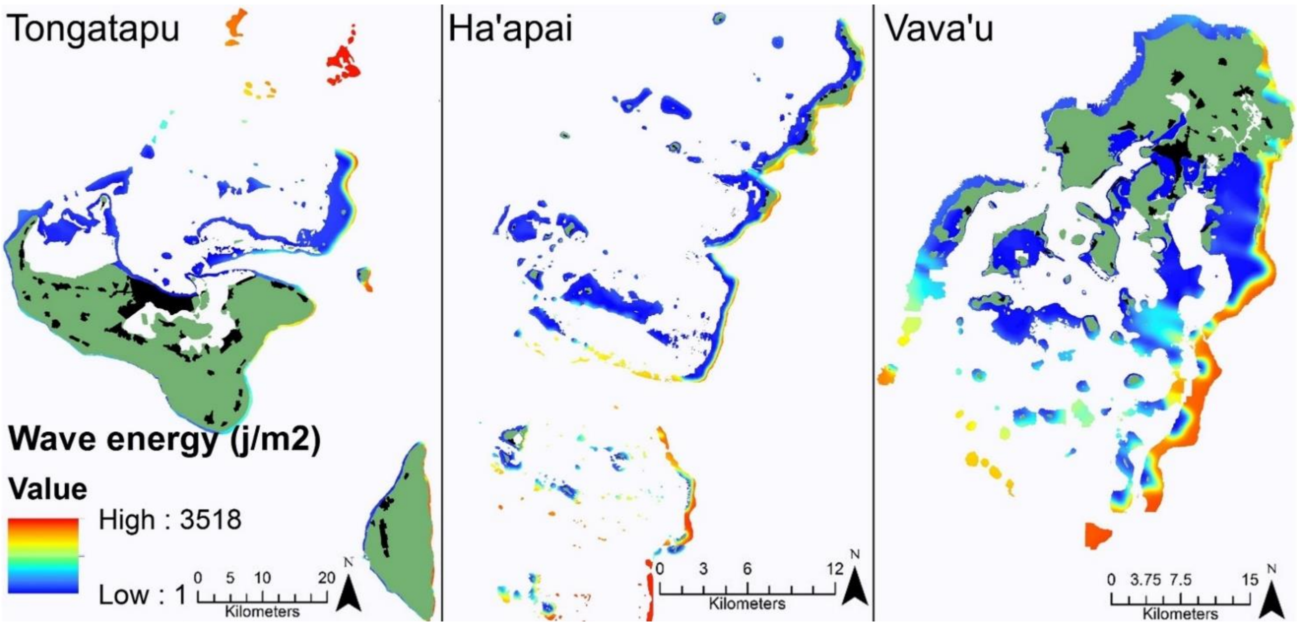


Figure S11. Mean wave energy, calculated as joules per m^2, for each 10 m^2 pixel of Tonga's near-shore shallow marine environment. Green areas represent land and black areas represent villages.

Anthropogenic variables

Distance from markets

Globally, distance to fish markets has a strong explanatory role in the structure of reef fish biomass (Brewer et al. 2012; Cinner et al. 2013). Market access can also be a better predictor of the condition of reef fish fisheries than the density of local human populations alone (Cinner and McClanahan 2006). Three main fish markets exist in Tonga, associated with the capital of each island group. The Tongatapu fish market is located at the small boats harbor near the Nuku'alofa wharf. The Vava'u fish market is situated at the main commercial wharf in Neiafu. While not permanent, in Ha'apai most reef fish are sold commercially at the Pangai wharf. The distance from the nearest of these three locations to each 10 m^2 pixel (marine extent defined by Andrefouet *et al.* (2006), see extent description in section 2.1.6 Habitat) was calculated using the *Euclidean distance* function (Fig. S12).

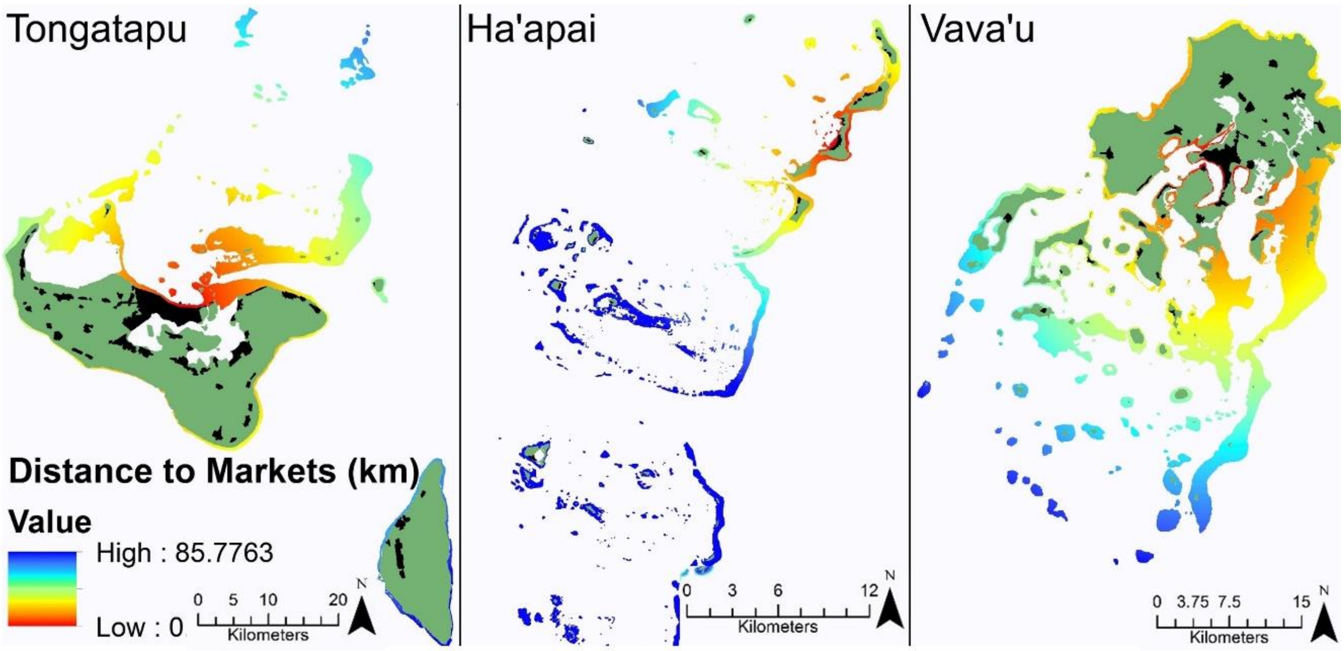


Figure S12. Distance to the three main fish markets for each 10 m^2 pixel of Tonga's near-shore marine environment. Green areas represent land and black areas represent villages.

Fishing pressure

Reef fish fisheries in Polynesia are critical for maintaining livelihoods and food security (Kronen 2004). Fishing pressure is a strong determinant of many metrics of reef health, and one of the most direct ways that humans interact with coral reefs (Cinner et al. 2018). Metrics of fishing pressure are often calculated using fisheries-dependent data (e.g. catch data). However, while some catch data are available from Tonga, they lack the spatial and temporal resolution, wide spread coverage and detail required to build an accurate model of fishing pressure for the entire region. Furthermore, current fishing activities may not be an accurate reflection of long-term trends, as fishers will likely change fishing grounds as stocks become depleted (Ochiewo 2004).The current study therefore used a combination of census data and key informant interviews to build a historical model of relative fishing effort across the reef fish fishing grounds of Tonga (Fig. S13, S14). This model represents a unit-less value of relative fishing effort that assumes fishers minimize travel time and only extend their range as closer stocks become depleted.

The reef fish fishery in Tonga can be broadly divided into commercial and subsistence fishing, each with different patterns of resource use and behavior (Kronen 2004). Key informant interviews were used to ascertain the specific details of both fishing practices, and took place during regular training meetings between Ministry of Fisheries staff and communities implementing new management areas. Interviews were conducted with both Ministry of Fisheries staff as well as local fishers (who classified themselves as either mostly commercial or mostly subsistence fishers). Twelve fishers from four villages agreed to participate in short informal interviews to discuss their fishing practices (Smallhorn-West et al. 2018). Fishers were asked the type of fishing they engage in, the methods employed and if willing, to outline on a map their fishing grounds.

The 2016 national census reported 2301 individuals in Tonga who identify as fishers. Of these, 1868 fish mainly for subsistence, while the remaining 433 reported fishing predominantly for commercial purposes (Statistics Department Tonga, 2016). Commercial fishing in Tonga is an organized profession, in which groups of fishers go out in boats at night time to fish an area of reef (Kronen 2004, Smallhorn-West *et al.* 2018). Following a night of fishing commercial fishers generally travel to the main fish markets and sell their catch to middlemen who run stalls in town and on roadsides. While commercial fishers also often engage in subsistence fishing, it is rare for subsistence fishers to fish commercially (Kronen 2004, Smallhorn-West *et al.* 2018). Subsistence fishing is here defined as 'fishing mainly for personal consumption or for that of family or gifts.' In contrast, subsistence fishing is much more opportunistic. Subsistence fishing is generally shore based and practiced close to the villages, with fishers swimming out from shore (Kronen 2004, Smallhorn-West *et al.* 2018).

Census data and key informant interviews were used to build a model of fishing pressure for Tonga, using similar methodology to Smallhorn-West *et al.* (2018). While village level population data was available from the 2016 national census, only district level data was available on fishing practices. Therefore, the village level abundance of commercial and subsistent fishers targeting reef fish was calculated by: 1) dividing the district level population of commercial and subsistence fishers by the population of each village; 2) multiplying the resulting value by the district level proportion of fishers who target reef fish, and 3) multiplying each value by a constant representing the proportional difference in total catch for each type of fishing, to account for differences in total catch between commercial and subsistent fishers.

An economic assessment of fisheries types in Tonga by Kronen (2004, Table S2) suggested that there was no clear economic distinction between commercial and subsistence coastal fisheries, however both national census data and key informant interviews suggested that fishers consistently identify themselves according to these categories. We therefore categorized Kronen (2004) Group 1 individuals as 'subsistence' and Group 3 as 'commercial' (Table 2). Group 1 individuals are predominantly shore based and align with subsistence practices. Group 3 fishers are exclusively spear fishers, fishing predominantly at night, which align with key informant interview findings of commercial practices. The proportional difference in catch between groups was calculated using total catch week^{-1} (kg) values of 40 and 75 kg respectively (Kronen 2004, Table S2). The abundance of commercial and subsistence fishers in each village was then multiplied by the proportional difference between these values, centered around 1 (1.30 commercial, 0.695 subsistence). The values for each village therefore represent the number of commercial or subsistence fishers who target reef fish, weighted by proportional differences in total catch (kg week^{-1}).

Table S2. Major characteristics of four Tonga fishery systems groups from Kronen (2004).

	Group I (simple coastal fishery system)	Group II (single-to multi-geared)	Group III (exclusive spear fishers)	Group IV (single- to multi-geared fishery systems with market choice)
Type of fishing	Simple	Variable	Specialised	Choice of markets
Boat transport	No, rarely or non-motorised	Owner and/or regular user of motorised	Always using rented motorised	Owner and/or regular user of motorised
Purchase of ice	No	Yes/no	Yes	Yes/no
Purchase of bait	No	Yes/no	No	Yes/no
Fishing gear	Restricted, mainly handline	Exclusive handline or multi-geared	Exclusive night time spear diving	Single- to multi-geared
Productivity CPUE kg	Low ≤ 3	Variable from ≥ 3.3 to mostly ≥ 6	Low 2.8	Variable from ≥ 3.3 to mostly ≥ 6
Average catch/trip kg	10–12	20–60	25	35–50
Total catch/week kg	40–50	60–180	75	105–160
Total hours fished/week	16–20	8–24	27	18–48
Major variables	Low investment, low productivity	Catch variation, compensation for motorised boat transport	High input cost prior to catches sold	Significant price variations according to markets served

To extrapolate fisher abundance across the reef fish fishing grounds of Tonga, polygons of each village (142 total) were created and converted to points. The fishing grounds for reef fish in Tonga are defined as all reef habitat from Andrefouet *et al.* (2006) and Purkis *et al.* (2019). The *heatmap* function (QGIS V.2.14) was then used to create separate decay kernels that extrapolated the weighted abundance of commercial and subsistence fishers across the reef habitat of Tonga. Key informant interviews established that commercial fishers fish every part of their island group, from inner to outer islands. The decay kernel extent was therefore set to 30 km, corresponding to the outer extent of each island group. Subsistence fishing is generally limited to the waters close by each village, and therefore the kernel extent was set with a cut-off of 3 km around each village. This distance is based on the maximum distance identified as fishing grounds by subsistence fishers during key informant interviews. All values of fishing pressure in Fish Habitat Reserves (FHRs) were set to 0, and Special Management Areas (SMA) values set to the sum of commercial and subsistence fishers from each corresponding SMA. This model therefore assumes full compliance by fishers. One caveat in this model is that many SMAs and FHRs have only been implemented recently and therefore values created might not represent accurate long term trends in fishing effort. The current study therefore also created two additional fishing pressure layers: 1) raw fishing pressure, values without any adjustments for management practices (Null model), and; 2) a layer only including SMAs/FHRs implemented more than five years previously (Old model).

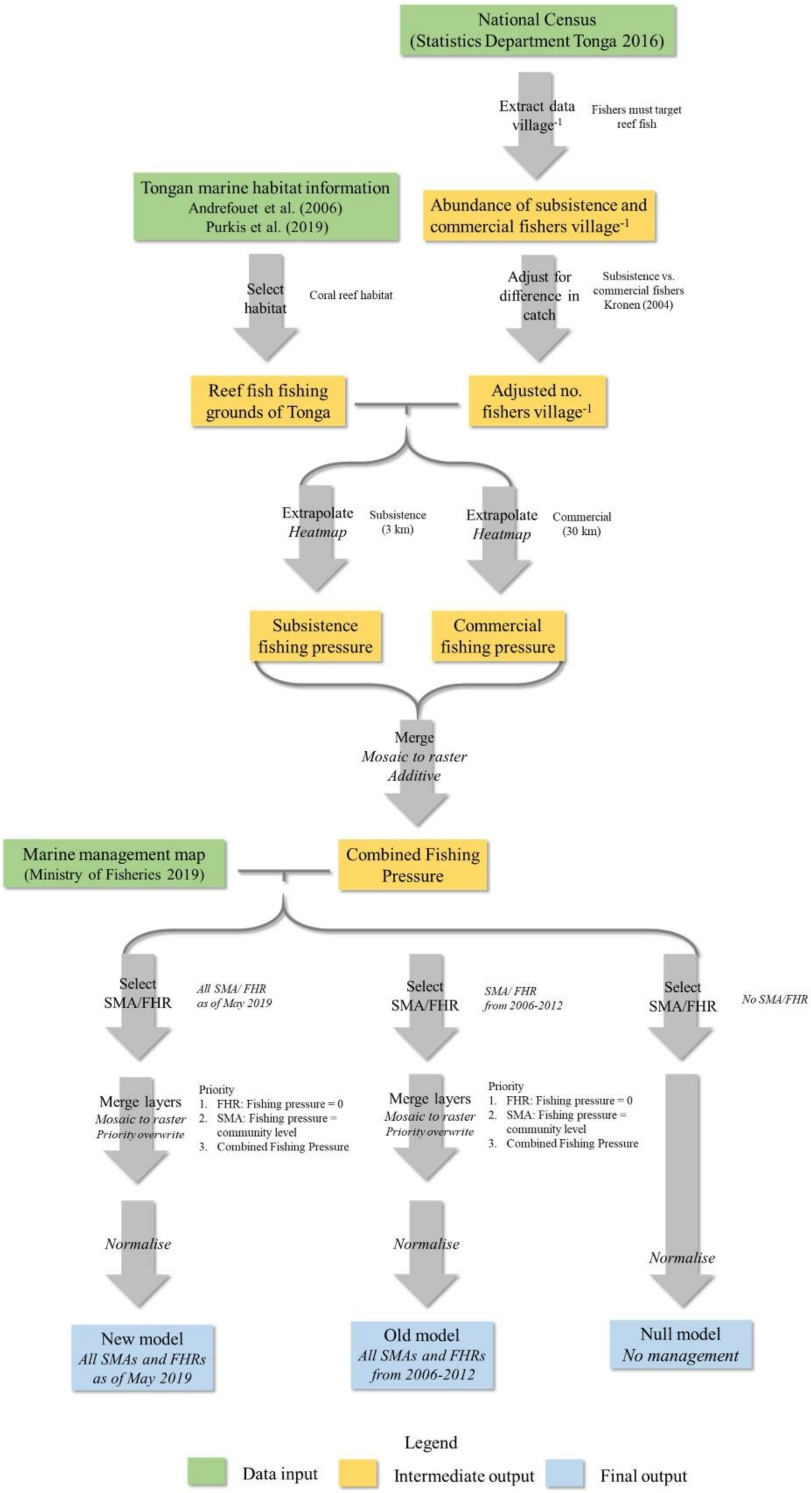


Figure S14. Flow chart representing the steps used to build three fishing pressure models for Tonga.

Commercial and subsistence fishing pressure heatmaps, as well as specific fishing pressure values for each FHR and SMA were merged using the *mosaic to new raster* function (ArcMap V10.4.1). This function added commercial and subsistence values together, but overruled them if the area corresponded to an SMA and/or FHR. This raster layer was subsequently clipped by the coral reef habitat of Tonga using the *extract by mask* function. Lastly, these values were normalized to provide values ranging between 0 and 100.

The final fishing pressure metric represents a unit-less value of relative fishing effort throughout the region. This metric assumes that, all else being equal, fishers preferentially select sites closer to home and extend their range as close locations become exhausted. While the model is therefore likely decoupled from current fishing effort, it is nonetheless useful in that it constitutes the historical impact of fishing on reef fish assemblages in Tonga.

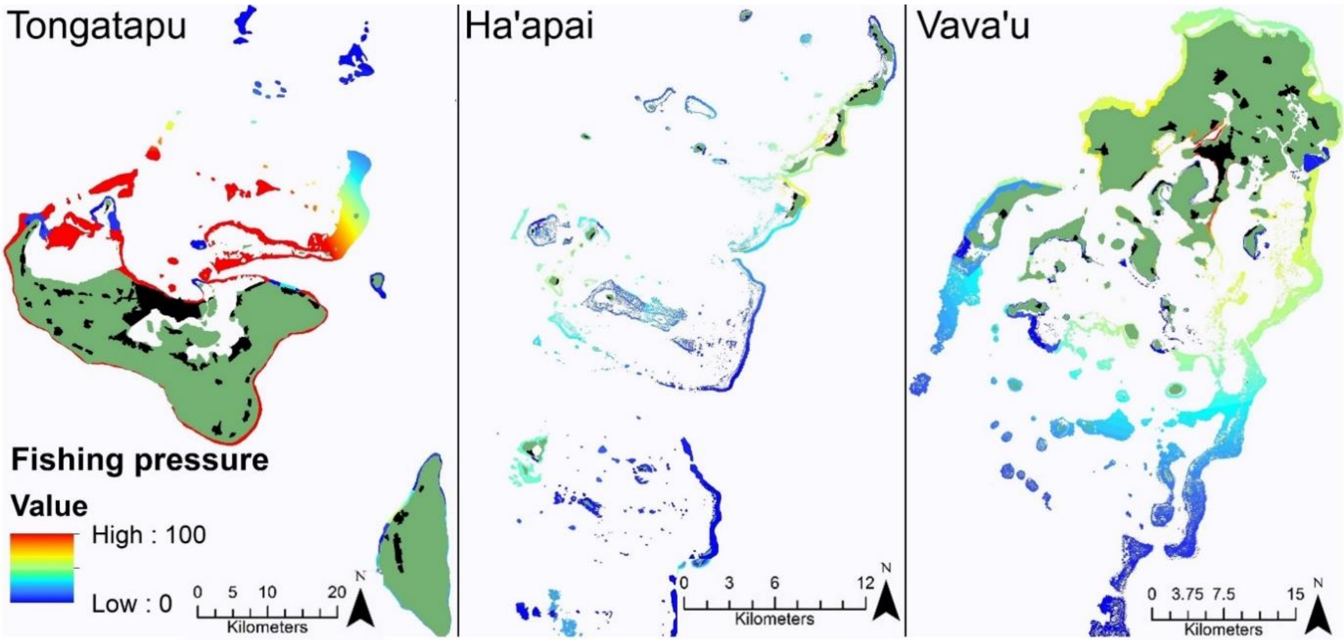


Figure S14. Relative fishing pressure for Tonga's coral reef ecosystem measured as the catch adjusted village level abundance of commercial and subsistence fishers extrapolated across the fishing grounds of Tonga. This figure represents the Current model, which includes all SMAs and FHRs as of 2019. Green areas represent land and black areas represent villages.

Management status

Extensive literature now demonstrates the global importance of marine protected areas as a way to reduce fishing pressure and change coral reef community structure (Lester et al. 2009; Edgar et al. 2014). Historically fishing in Tonga has been open access. In 2002, amid concerns over the depletion of the reef fish fishery, the Tongan Ministry of Fisheries implemented the Special Management Area (SMA) program (Gillett 2017). Special management areas are locally managed marine protected areas comprised of two management components: 1) an exclusive access zone in which only members of the SMA community can fish, and; 2) a permanent no-take Fish Habitat Reserve (FHR) in which no one can fish. While the extent of each SMA is defined by the Ministry of Fisheries, the size and location of the FHR within is determined by the community itself (represented by the SMA committee). It is the responsibility of each community to manage and enforce compliance of fishers within their SMA and FHR. While between 2006 and 2014 only seven SMAs were implemented, recently community demand has increased rapidly, with over 40 new SMAs gazetted in the past five years. Separate polygon layers were created to define the location of SMAs and FHRs, with the area (km^2), perimeter length (km) and year established of each SMA and FHR embedded in the spatial layer (Fig. S15; Table S3).

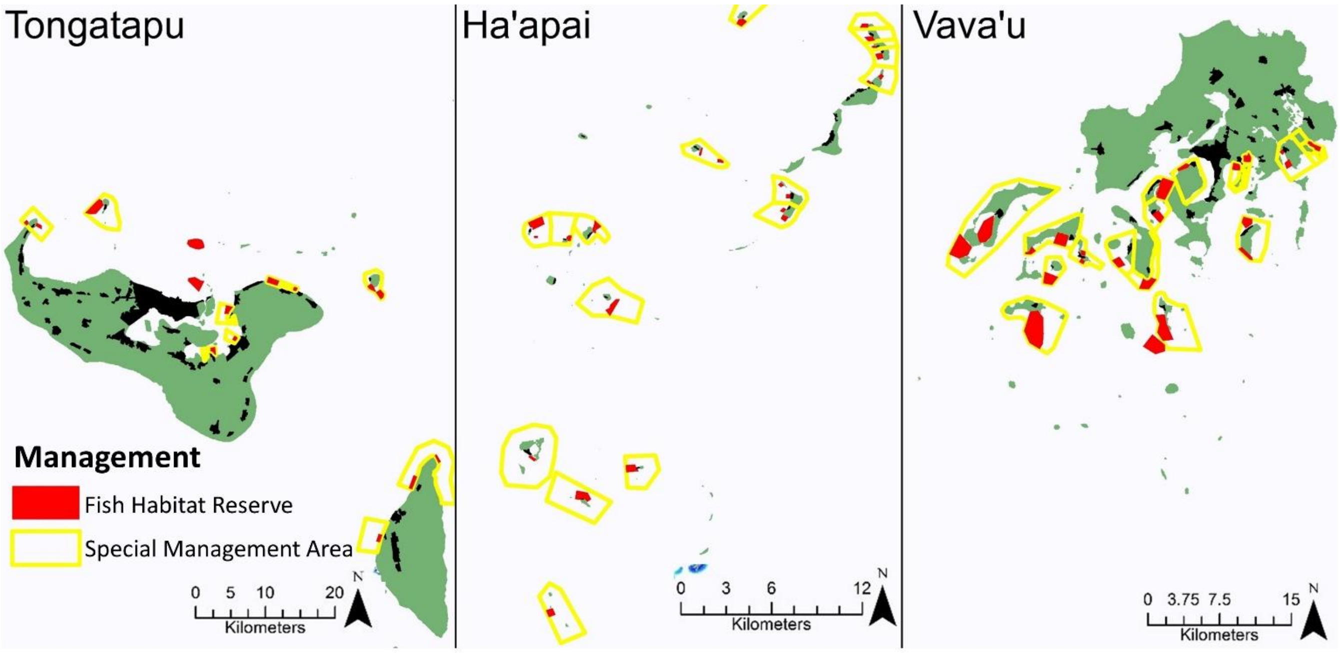


Figure S15. Configuration of Special Management Areas and Fish Habitat Reserves in Tonga. Green areas represent land and black areas represent villages.

Table S3. All special management areas and fish habitat reserves in Tonga as of May 2019.

Name	Island group	Year Established	Area (km2)	Perimeter (km)	Name	Island Group	Year Established	Area (km2)	Perimeter (km)
Atata FHR	Tongatapu	2008	1.54	5.48	Lapaha SMA	Tongatapu	2016	1.10	4.41
Atata SMA	Tongatapu	2008	8.40	11.46	Lape FHR	Vava'u	2017	0.58	3.15
Eueiki FHR Vavau	Vava'u	2017	1.19	4.37	Lape SMA	Vava'u	2017	1.98	5.57
Euiki FHR 1 Tongatapu	Tongatapu	2008	0.50	2.80	Lofanga FHR 1	Ha'apai	2018	0.36	3.08
Euiki FHR 2 Tongatapu	Tongatapu	2008	0.37	2.73	Lofanga FHR 2	Ha'apai	2018	0.45	2.77
Euiki SMA	Tongatapu	2008	3.75	8.36	Lofanga SMA	Ha'apai	2018	14.83	18.20
Fafa FHR	Tongatapu	2014	1.59	4.99	Makave FHR 1	Vava'u	2019	0.23	1.91
Fakakakai FHR	Ha'apai	2018	0.94	3.93	Makave FHR 2	Vava'u	2019	0.25	2.12
Fakakakai SMA	Ha'apai	2018	10.74	13.96	Makave SMA	Vava'u	2019	1.68	11.62
Faleloa FHR 1	Ha'apai	2018	0.45	2.72	Mango FHR	Ha'apai	2017	2.78	7.53
Faleloa FHR 2	Ha'apai	2018	0.25	2.22	Mango SMA	Ha'apai	2017	39.75	27.51
Faleloa SMA	Ha'apai	2018	15.83	16.50	Matamaka FHR 1	Vava'u	2019	0.10	1.30
Falevai FHR	Vava'u	2017	0.36	2.49	Matamaka FHR 2	Vava'u	2019	0.09	1.29
Falevai SMA	Vava'u	2017	3.98	8.06	Matamaka SMA	Vava'u	2019	2.09	7.11
Felemea FHR 1	Ha'apai	2008	0.44	2.78	Matuku FHR	Ha'apai	2017	0.55	3.08
Felemea FHR 2	Ha'apai	2008	0.74	3.51	Matuku SMA	Ha'apai	2017	16.89	17.00
Felemea SMA	Ha'apai	2008	17.10	17.99	Muitoa FHR	Ha'apai	2018	0.72	3.71
Fonoi FHR	Ha'apai	2017	1.91	6.20	Muitoa SMA	Ha'apai	2018	10.81	16.04
Fonoi SMA	Ha'apai	2017	22.33	18.44	Nomuka FHR	Ha'apai	2011	0.53	3.26
Ha'afeva FHR 1	Ha'apai	2007	0.44	2.75	Nomuka SMA	Ha'apai	2011	68.20	30.40
Ha'afeva FHR 2	Ha'apai	2007	0.95	4.12	Nuapapu FHR 1	Vava'u	2019	0.19	2.09
Ha'afeva SMA	Ha'apai	2007	14.30	16.69	Nuapapu FHR 2	Vava'u	2019	0.69	3.34
Ha'ano FHR	Ha'apai	2018	0.87	4.23	Nuapapu SMA	Vava'u	2019	5.83	11.62
Ha'ano SMA	Ha'apai	2018	11.96	17.27	Nukuleka FHR	Tongatapu	2016	0.51	3.04
Ha'atafu FHR 1	Tongatapu	2017	0.17	1.62	Nukuleka SMA	Tongatapu	2016	2.63	9.16
Ha'atafu FHR 2	Tongatapu	2017	0.24	2.14	Ofolanga FHR 1	Ha'apai	2018	1.80	5.35
Ha'atafu SMA	Tongatapu	2017	5.35	9.58	Ofolanga FHR 2	Ha'apai	2018	1.20	4.47
Holoeva FHR	Vava'u	2019	0.25	2.55	Ofolanga SMA	Ha'apai	2018	40.70	26.88
Holoeva SMA	Vava'u	2019	1.50	10.13	Ofu FHR 1	Vava'u	2017	0.29	2.82
Holonga FHR	Tongatapu	2017	0.30	2.42	Ofu FHR 2	Vava'u	2017	0.38	2.41
Holonga SMA	Tongatapu	2017	0.93	5.74	Ofu SMA	Vava'u	2017	4.93	8.55
Houma FHR 1	Eua	2019	0.58	3.62	Oua FHR	Ha'apai	2006	2.16	7.32
Houma FHR 2	Eua	2019	0.23	2.27	Oua SMA	Ha'apai	2006	41.68	27.26
Houma SMA	Eua	2019	17.48	26.75	Ovaka FHR	Vava'u	2008	2.60	6.38
Hunga FHR 1	Vava'u	2017	1.46	4.84	Ovaka SMA	Vava'u	2008	9.21	13.31
Hunga FHR 2	Vava'u	2017	1.32	4.77	Pangaimotu FHR	Tongatapu	2017	1.40	5.06
Hunga SMA	Vava'u	2017	20.73	21.40	Pukotala FHR	Ha'apai	2018	0.23	2.33
Kapa FHR	Vava'u	2019	0.58	3.49	Pukotala SMA	Ha'apai	2018	5.68	12.47
Kapa SMA	Vava'u	2019	2.33	11.60	Talihau FHR	Vava'u	2017	0.36	2.47
Kelefesia FHR	Ha'apai	2018	1.31	4.61	Talihau SMA	Vava'u	2017	2.52	6.16
Kelefesia SMA	Ha'apai	2018	32.72	24.31	Taunga FHR	Vava'u	2013	1.21	5.10
Koloa FHR 1	Vava'u	2017	0.06	0.99	Taunga SMA	Vava'u	2013	7.74	11.78
Koloa FHR 2	Vava'u	2017	0.20	1.82	Tufuva FHR	Eua	2018	0.33	2.45
Koloa SMA	Vava'u	2017	4.52	8.42	Tufuva SMA	Eua	2019	7.24	11.16
Kolonga FHR 1	Tongatapu	2015	0.15	1.57	Uiha FHR 1	Ha'apai	2018	0.37	2.51
Kolonga FHR 2	Tongatapu	2015	0.70	3.65	Uiha FHR 2	Ha'apai	2018	0.46	2.74
Kolonga SMA	Tongatapu	2015	1.64	7.96	Uiha SMA	Ha'apai	2018	17.09	17.45
Kotu FHR 1	Ha'apai	2015	3.02	7.54	Utulei FHR	Vava'u	2017	0.21	2.18
Kotu FHR 2	Ha'apai	2015	0.19	1.92	Utulei SMA	Vava'u	2017	4.16	7.99
Kotu SMA	Ha'apai	2015	16.86	15.73	Utungake FHR	Vava'u	2017	1.08	4.22
Lapaha FHR	Tongatapu	2016	0.19	1.68	Utungake SMA	Vava'u	2017	2.34	6.54

Population density

Globally, human population pressure is one of the strongest drivers of ecological and anthropogenic patterns on coral reefs (Cinner et al. 2018), driving changes in fishing, pollution and other destructive practices. While spatial layers describing metrics of fishing pressure and pollution were supplied in the current dataset, raw human population pressure may also be a useful metric required by end users. Human population density within 5, 15 and 30 km of all $10m^2$ pixels of near-shore marine habitat was therefore calculated using uniform kernel heatmaps (QGIS V.2.14) and village level population data from the 2016 census.Resulting heatmaps were subsequently clipped by the extent of the near-shore marine environment of Tonga (as defined by Andrefouet *et al.* (2006) using the *extract by mask* function (ArcMap V10.4.1) (Fig. S16). Distance cut-offs for population pressure followed that of previous studies utilizing radiuses of 5 km (Stallings 2009, Cinner et al. 2013), 15 km (Williams et al. 2008), and 30 km (Halpern et al. 2008, Mora et al. 2011).

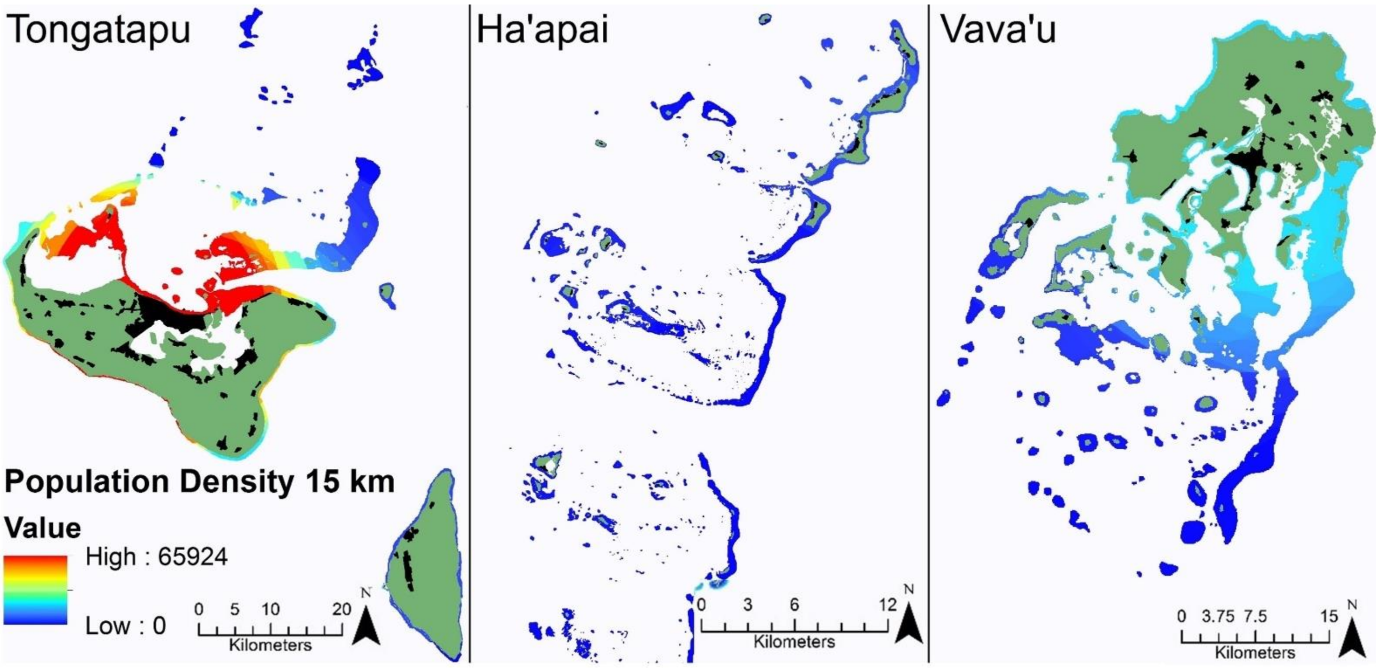


Figure S16. Population within 15 kilometers of each 10 m^2 pixel of Tonga's near-shore marine ecosystem. Additional layers with population density within 5 km and 30 km are also provided. Green areas represent land and black areas represent villages.

Socioeconomic development index

The level of socioeconomic development of a region may affect the marine environment in a variety of ways. There exists the potential for a both an increase (e.g. greater effluent runoff associated with higher population density) and a decrease (e.g. reduced rubbish dumping associated with increased access to waste management services)) in some harmful activities in areas with higher levels of socioeconomic development (Brewer et al. 2012, Harborne et al. 2016). Data from the 2016 national census (Statistics Department Tonga, 2016) was used to calculate the population density, population growth rate, mean age, education level and level of unemployment for each village in Tonga. Rather than using each variable separately, these data were combined using multivariate analysis to create a composite index of socioeconomic development for each village in Tonga (following Harborne 2016) (Fig. S17).

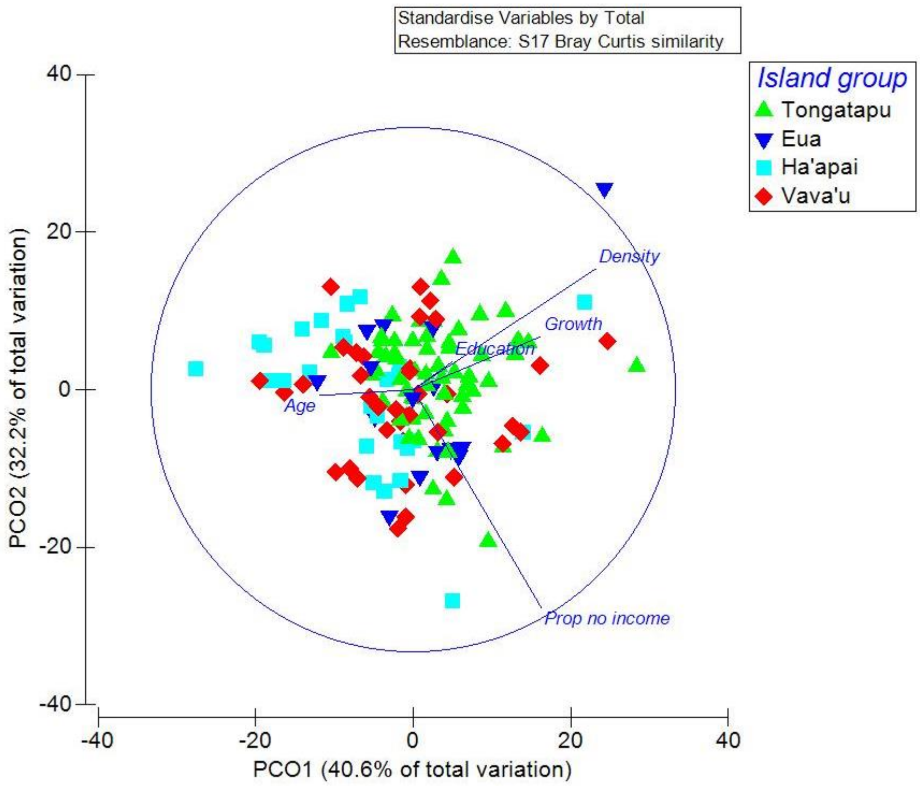


Figure S17. Principal component ordination of five indicators of socioeconomic development for all 142 villages in Tonga.

Population density was calculated by: 1) using satellite images to create polygons for each village in Tonga, and; 2) dividing each villages' population by the area of the polygon. Population growth rate was calculated using the yearly difference in population between the 2016 and 2011 census. Highest level of education was divided into six categories (preschool, primary, lower and higher secondary, technical and tertiary), which were classified on a 12 point scale, to calculate mean education level for each village. The proportion of each village not engaged in work as their main income source was defined as categories 'no income' and 'remittance' from the occupation section of the 2016 national census (Statistics Department Tonga, 2016). All values were weighted equally prior to analysis. Principal component ordination (PCO) was used to calculate the distance between villages relative to the axis accounting for the greatest amount of data variability. Axis 1 explained 40.6% of variation between villages, with higher values on this axis representing villages with higher population density, faster growth, greater levels of education and a younger mean age.

Values from the primary axis were used as a metric of socioeconomic development for each village. The subsequent socioeconomic indices were then extrapolated across 10 m^2 pixels of the near-shore marine ecosystem of Tonga (Fig. S18). Heatmaps using a uniform kernel shape and a radius of 2, 5 and 10 km were generated (QGIS V.2.14) and subsequently clipped by the Andrefouet *et al.* (2006) defined habitat extent. All raster cells that exceeded the specified radius (e.g 2, 5 or 10 km) were left blank (no data). Pixels with positive values represent areas within the sphere of influence of communities with a high socioeconomic development indices, while negative values represent areas influenced by communities with low socioeconomic development.

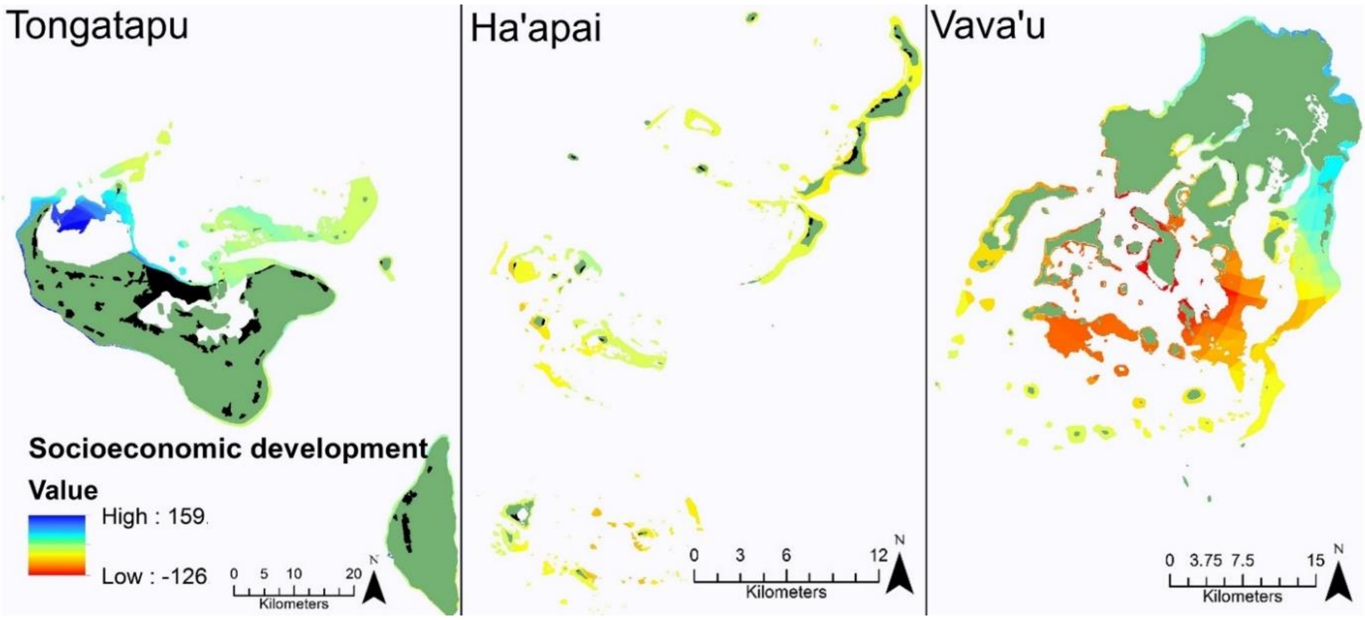


Figure S18. Socioeconomic development axis 1, explaining 40.6% of total variation between villages. Larger values represent $10m^2$ pixels within the sphere of influence of communities with higher population densities, growth and mean education level and younger mean age. This layer represents values extrapolated to 10 km, but additional layers with socioeconomic development within 2 km and 5 km are also provided. Green areas represent land and black areas represent villages.

Distance from village

The distance from each 10 m^2 pixel to the nearest village was also included as a layer within this dataset (Fig. S19). While other factors such as population pressure and fishing pressure may be stronger drivers of ecological processes, distance from village may be useful for other applications by end users. For example, distance from village may be an important determinant of marine traffic intensity or may aid in identify the location of a new marine industrial project. . The distance from each 10 m^2 pixel of near-shore marine environment to the nearest village was therefore calculated using the *Euclidean distance* function (ArcMap V.10.4.1) and subsequently clipped by the habitat extent defined by Andrefouet *et al.* (2006).

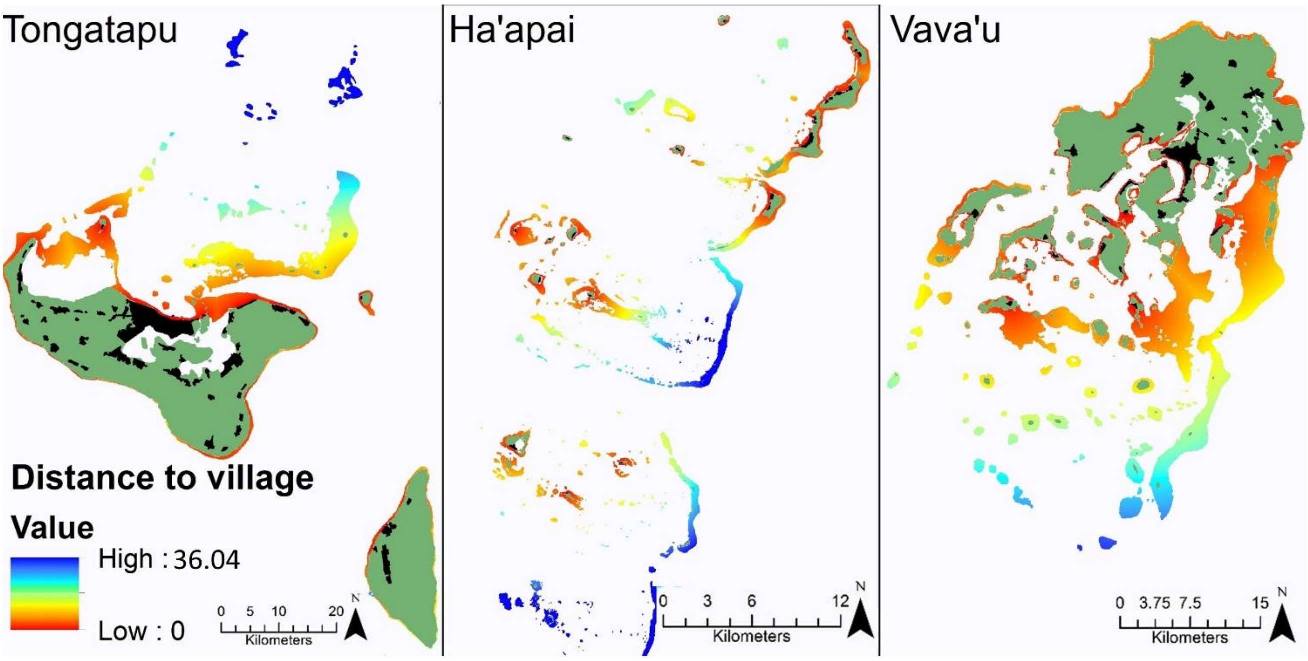


Figure S19. Distance from the nearest village for each 10 m^2 pixel of Tonga's near-shore marine environment. Green areas represent land and black areas represent villages.

Village

Polygons were created from outlines of each village (142) in Tonga using satellite imagery. The subsequent layer is supplied with the village-associated data from the 2016 national census (Statistics Department Tonga, 2016) embedded within the file. This village-associated data was used as inputs to generate fishing pressure, population density and socioeconomic development index spatial layers in the current dataset. Data included within the attribute table are: village name, area, population, population density, weighted number of commercial fishers, weighted number of subsistence fishers, socioeconomic development score, education score, population growth, mean age, proportion of population not engaged in work, island group, district, and village block.

Supplementary Materials: Chapter 4

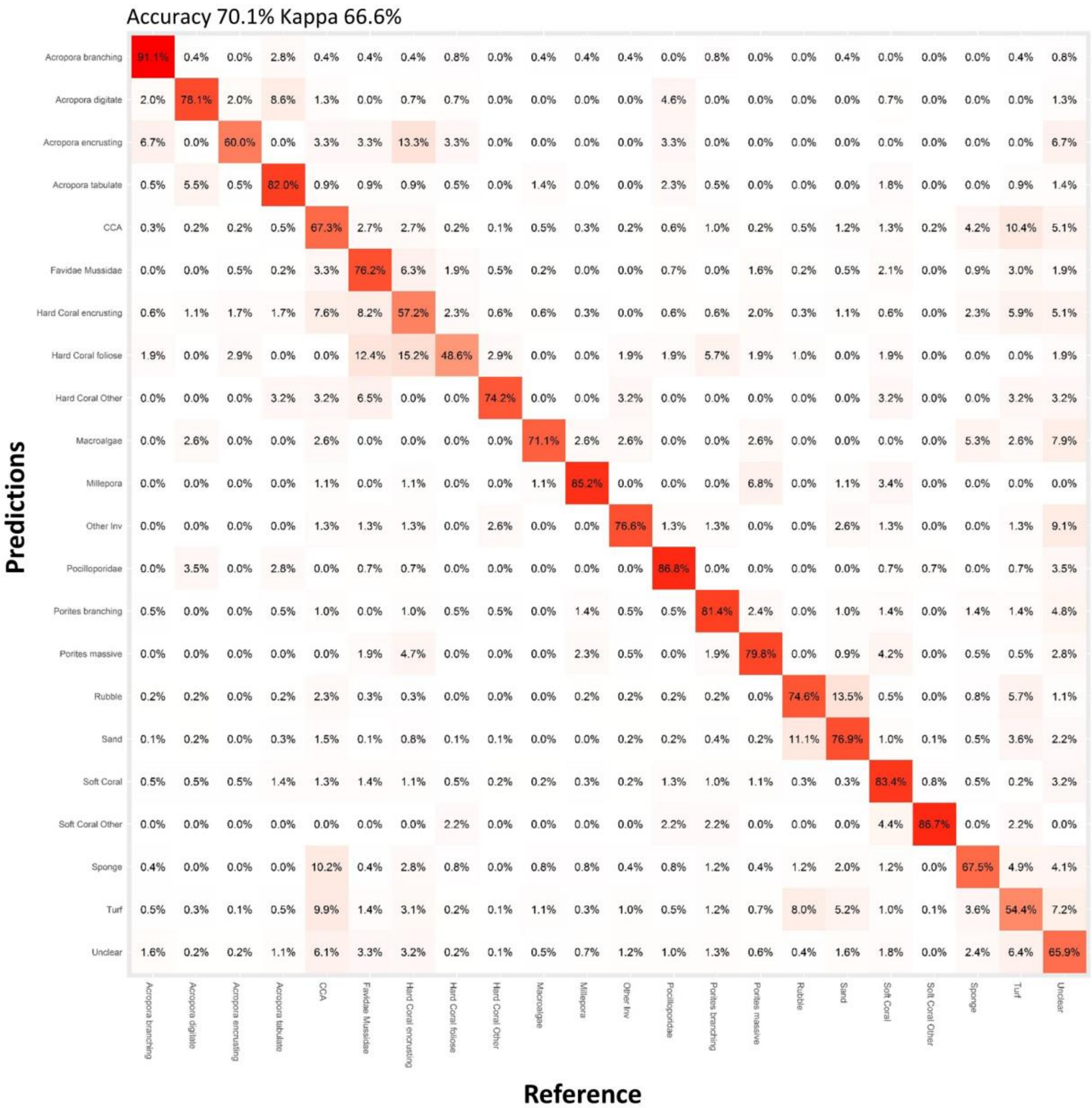
Benthic Composition

Benthic community composition was estimated using image analysis of ten 1 x 1 m benthic photoquadrats per transect with 15 points randomly overlaid across each image (total 150 points per transect). Given the large number of images and points required for annotation (images = 11020; points = 165300), we used the machine learning software BenthoBox (www.benthobox.com) to assist with the benthic annotations. BenthoBox automatically classifies points into benthic substrate categories from images based on training provided by a human annotator.

A label set of 22 benthic categories from four functional groups was established based on their functional relevance to coral reef ecosystems and their ability to be reliably identified from images by human and automated annotators (Beijbom et al., 2015a; González-Rivero et al., 2016). Four broad functional groups represent the main benthic components of coral reefs in Tonga: "Algae", "Hard Coral", "Other Invertebrates" and "Other". The main algal groups were categorized based on their functional relevance: Crustose Coralline Algae (CCA), Macroalgae and Turf algae. Turf algae are considered a grazed assemblage of algal species up to 1 cm in height (González-Rivero et al., 2016). Hard corals comprise 12 groups classified based on a combination of taxonomy (*i.e.* family) and functional morphology. Fire corals of the family Milleporidae (Class Hydrozoa) were included in the hard coral category because they fulfil a similar functional role i.e. the provision of three-dimensional habitat complexity. Soft corals were divided into Alcyoniidae soft corals (class "Soft Corals") and Gorgoniidae soft corals (class "Other Soft Corals").

The aim of the automated annotation method is to learn from human annotations and automatically analyse the remaining images to within an acceptable margin of error (Beijbom, 2015a,b). While automation typically captures similar trends but with higher variability than among human annotators (Beijbom et al. 2015; González-Rivero et al., 2016), the impact of this error on interpretation depends on the relative abundance of organisms, taxonomic resolution and ecological relevance of the variables in question. Typically, the noise around automated annotations may lead to misinterpretations of rare categories (<5 % total cover) for which the average abundance is similar to the error in quantification. However, the impact of automated analysis error on more dominant benthic groups (>5 % total cover) is less pronounced, and usually has marginal effects on derived cover estimates (González-Rivero et al., 2016). For the purposes of this study we therefore included four common benthic categories each with mean cover greater than five percent: Hard Coral, Soft Coral, CCA and Turf Algae.

A total of 4880 images were drawn from the overall pool of images and manually annotated to use as training and validation sets for the automated annotator. This consisted of all 3880 images annotated from the Vava'u island group, and an additional 1000 images (10% of total) randomly selected from the Ha'apai and Tongatapu island groups. The Vava'u annotations had been completed as part of a previous study (Smallhorn-West et al. 2019) but were included as extra training data for the model. The validation set consisted of 10,000 random manual annotations withheld from the training dataset and instead used to compare machine predictions with human annotations. The overall error of each benthic category in percent cover was then calculated and used to determine whether the machine's accuracy fell within acceptable bounds. Variability in the error of percent cover was calculated by randomly subsetting the holdout set into ten subcategories to generate a mean and 95% confidence intervals (Fig S1-S4)



Predictions \ Reference	Acropora branching	Acropora digitate	Acropora encrusting	Acropora tabulate	CCA	Favidae Mussidae	Hard Coral encrusting	Hard Coral foliose	Hard Coral Other	Macroalgae	Millepora	Other Inv	Pocilloporidae	Porites branching	Porites massive	Rubble	Sand	Soft Coral	Soft Coral Other	Sponge	Turf	Unclear
Acropora branching	91.1%	0.4%	0.0%	2.8%	0.4%	0.4%	0.4%	0.8%	0.0%	0.4%	0.4%	0.4%	0.0%	0.8%	0.0%	0.0%	0.4%	0.0%	0.0%	0.0%	0.4%	0.8%
Acropora digitate	2.0%	78.1%	2.0%	8.6%	1.3%	0.0%	0.7%	0.7%	0.0%	0.0%	0.0%	0.0%	4.6%	0.0%	0.0%	0.0%	0.0%	0.7%	0.0%	0.0%	0.0%	1.3%
Acropora encrusting	6.7%	0.0%	60.0%	0.0%	3.3%	3.3%	13.3%	3.3%	0.0%	0.0%	0.0%	0.0%	3.3%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	6.7%
Acropora tabulate	0.5%	5.5%	0.5%	82.0%	0.9%	0.9%	0.9%	0.5%	0.0%	1.4%	0.0%	0.0%	2.3%	0.5%	0.0%	0.0%	0.0%	1.8%	0.0%	0.0%	0.9%	1.4%
CCA	0.3%	0.2%	0.2%	0.5%	67.3%	2.7%	2.7%	0.2%	0.1%	0.5%	0.3%	0.2%	0.6%	1.0%	0.2%	0.5%	1.2%	1.3%	0.2%	4.2%	10.4%	5.1%
Favidae Mussidae	0.0%	0.0%	0.5%	0.2%	3.3%	76.2%	6.3%	1.9%	0.5%	0.2%	0.0%	0.0%	0.7%	0.0%	1.6%	0.2%	0.5%	2.1%	0.0%	0.9%	3.0%	1.9%
Hard Coral encrusting	0.6%	1.1%	1.7%	1.7%	7.6%	8.2%	57.2%	2.3%	0.6%	0.6%	0.3%	0.0%	0.6%	0.6%	2.0%	0.3%	1.1%	0.6%	0.0%	2.3%	5.9%	5.1%
Hard Coral foliose	1.9%	0.0%	2.9%	0.0%	0.0%	12.4%	15.2%	48.6%	2.9%	0.0%	0.0%	1.9%	1.9%	5.7%	1.9%	1.0%	0.0%	1.9%	0.0%	0.0%	0.0%	1.9%
Hard Coral Other	0.0%	0.0%	0.0%	3.2%	3.2%	6.5%	0.0%	0.0%	74.2%	0.0%	0.0%	3.2%	0.0%	0.0%	0.0%	0.0%	0.0%	3.2%	0.0%	0.0%	3.2%	3.2%
Macroalgae	0.0%	2.6%	0.0%	0.0%	2.6%	0.0%	0.0%	0.0%	0.0%	71.1%	2.6%	2.6%	0.0%	0.0%	2.6%	0.0%	0.0%	0.0%	0.0%	5.3%	2.6%	7.9%
Millepora	0.0%	0.0%	0.0%	0.0%	1.1%	0.0%	1.1%	0.0%	0.0%	1.1%	85.2%	0.0%	0.0%	0.0%	6.8%	0.0%	1.1%	3.4%	0.0%	0.0%	0.0%	0.0%
Other Inv	0.0%	0.0%	0.0%	0.0%	1.3%	1.3%	1.3%	0.0%	2.6%	0.0%	0.0%	76.6%	1.3%	1.3%	0.0%	0.0%	2.6%	1.3%	0.0%	0.0%	1.3%	9.1%
Pocilloporidae	0.0%	3.5%	0.0%	2.8%	0.0%	0.7%	0.7%	0.0%	0.0%	0.0%	0.0%	0.0%	86.8%	0.0%	0.0%	0.0%	0.0%	0.7%	0.7%	0.0%	0.7%	3.5%
Porites branching	0.5%	0.0%	0.0%	0.5%	1.0%	0.0%	1.0%	0.5%	0.5%	0.0%	1.4%	0.5%	0.5%	81.4%	2.4%	0.0%	1.0%	1.4%	0.0%	1.4%	1.4%	4.8%
Porites massive	0.0%	0.0%	0.0%	0.0%	0.0%	1.9%	4.7%	0.0%	0.0%	0.0%	2.3%	0.5%	0.0%	1.9%	79.8%	0.0%	0.9%	4.2%	0.0%	0.5%	0.5%	2.8%
Rubble	0.2%	0.2%	0.0%	0.2%	2.3%	0.3%	0.3%	0.0%	0.0%	0.0%	0.2%	0.2%	0.2%	0.2%	0.0%	74.6%	13.5%	0.5%	0.0%	0.8%	5.7%	1.1%
Sand	0.1%	0.2%	0.0%	0.3%	1.5%	0.1%	0.8%	0.1%	0.1%	0.0%	0.0%	0.2%	0.2%	0.4%	0.2%	11.1%	76.9%	1.0%	0.1%	0.5%	3.6%	2.2%
Soft Coral	0.5%	0.5%	0.5%	1.4%	1.3%	1.4%	1.1%	0.5%	0.2%	0.2%	0.3%	0.2%	1.3%	1.0%	1.1%	0.3%	0.3%	83.4%	0.8%	0.5%	0.2%	3.2%
Soft Coral Other	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	2.2%	0.0%	0.0%	0.0%	0.0%	2.2%	2.2%	0.0%	0.0%	0.0%	4.4%	86.7%	0.0%	2.2%	0.0%
Sponge	0.4%	0.0%	0.0%	0.0%	10.2%	0.4%	2.8%	0.8%	0.0%	0.8%	0.8%	0.4%	0.8%	1.2%	0.4%	1.2%	2.0%	1.2%	0.0%	67.5%	4.9%	4.1%
Turf	0.5%	0.3%	0.1%	0.5%	9.9%	1.4%	3.1%	0.2%	0.1%	1.1%	0.3%	1.0%	0.5%	1.2%	0.7%	8.0%	5.2%	1.0%	0.1%	3.6%	54.4%	7.2%
Unclear	1.6%	0.2%	0.2%	1.1%	6.1%	3.3%	3.2%	0.2%	0.1%	0.5%	0.7%	1.2%	1.0%	1.3%	0.6%	0.4%	1.6%	1.8%	0.0%	2.4%	6.4%	65.9%

Fig S1. Confusion matrix of individual benthic categories used for automated image annotation.

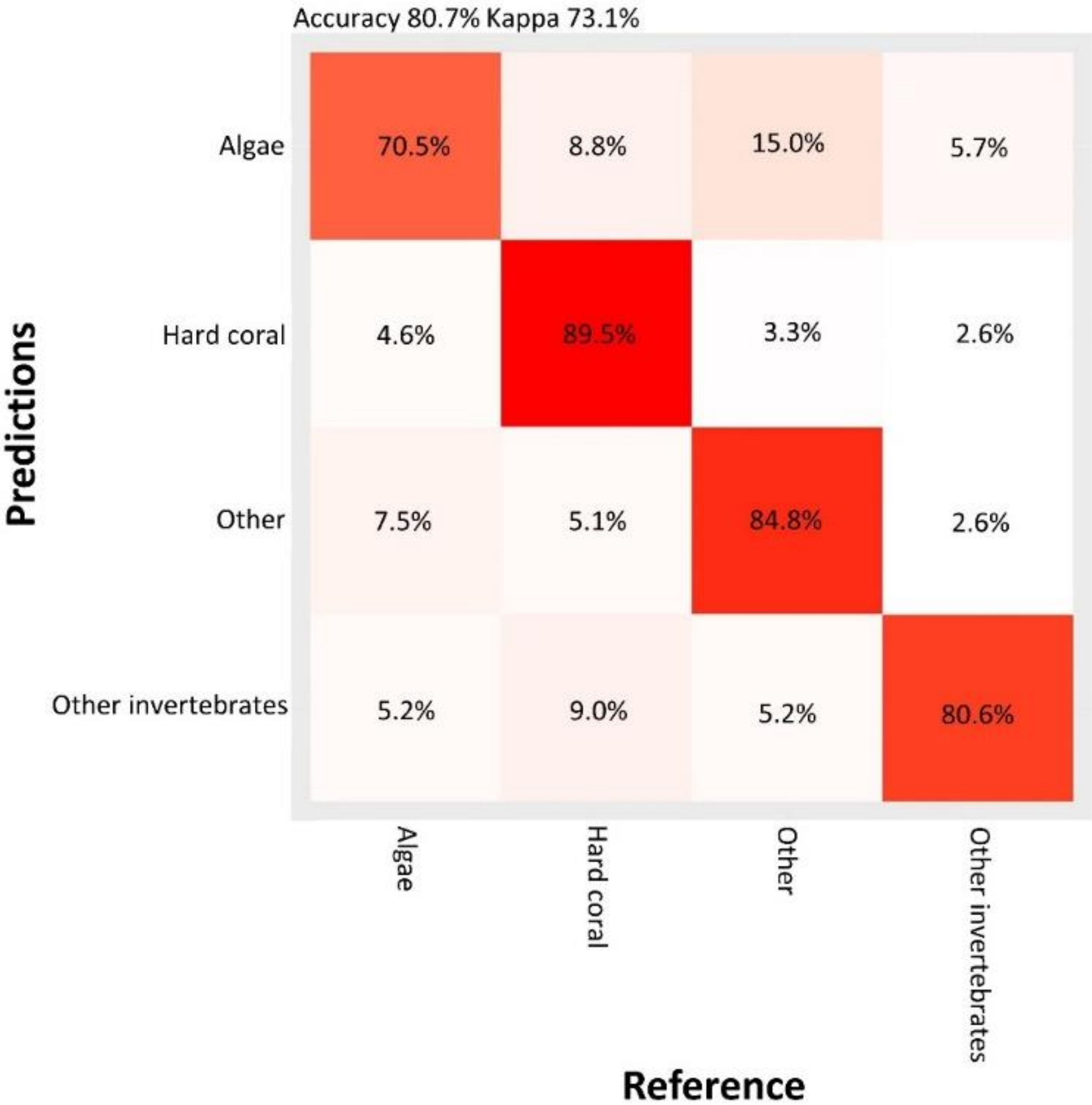


Fig S2. Funcitonal level confusion matrix of benthic categories used for automated image annotation.

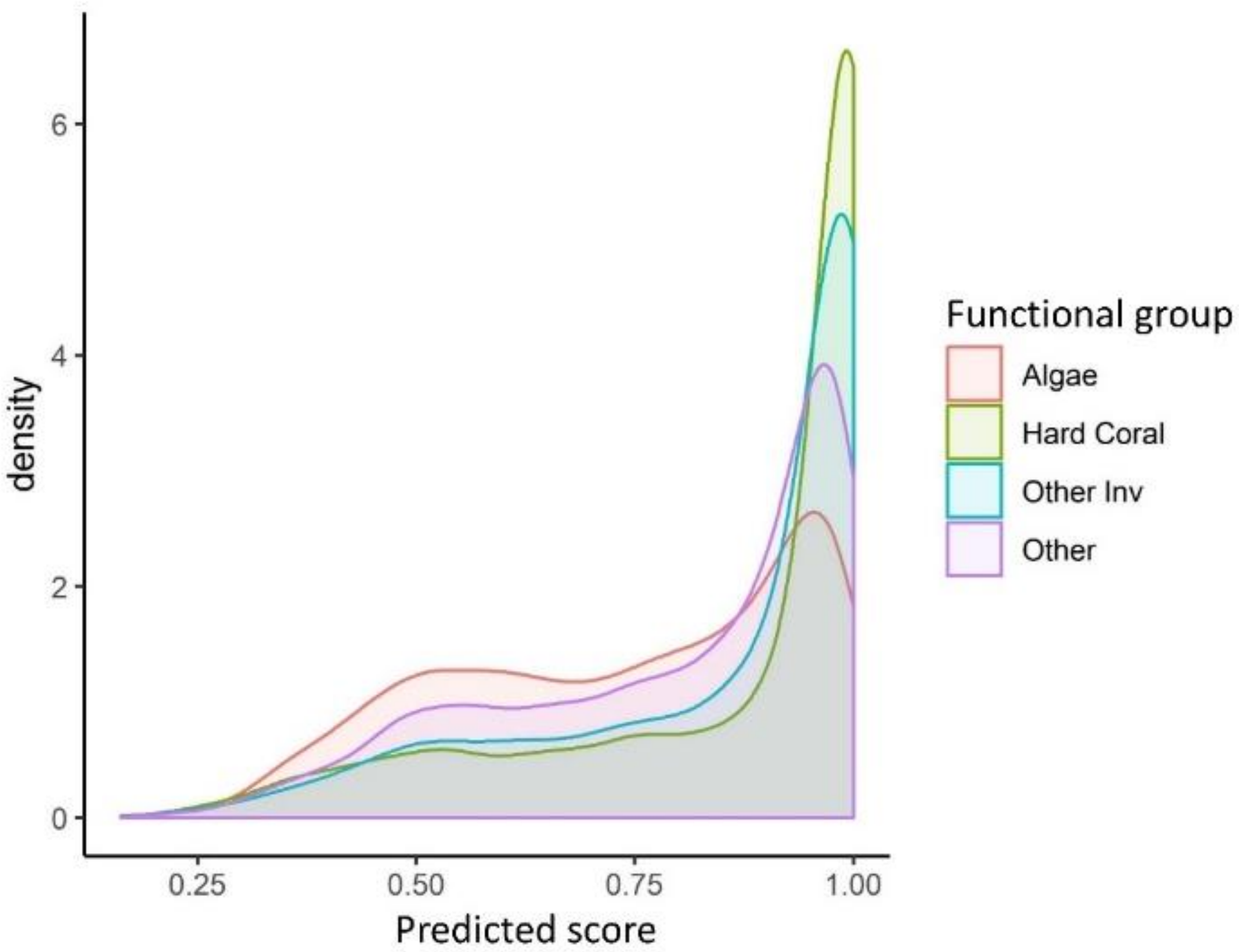


Fig S3. Density plot of machine confidence in functional level annotations. X-axis represents the confidence (%) that a given annotation is correct.

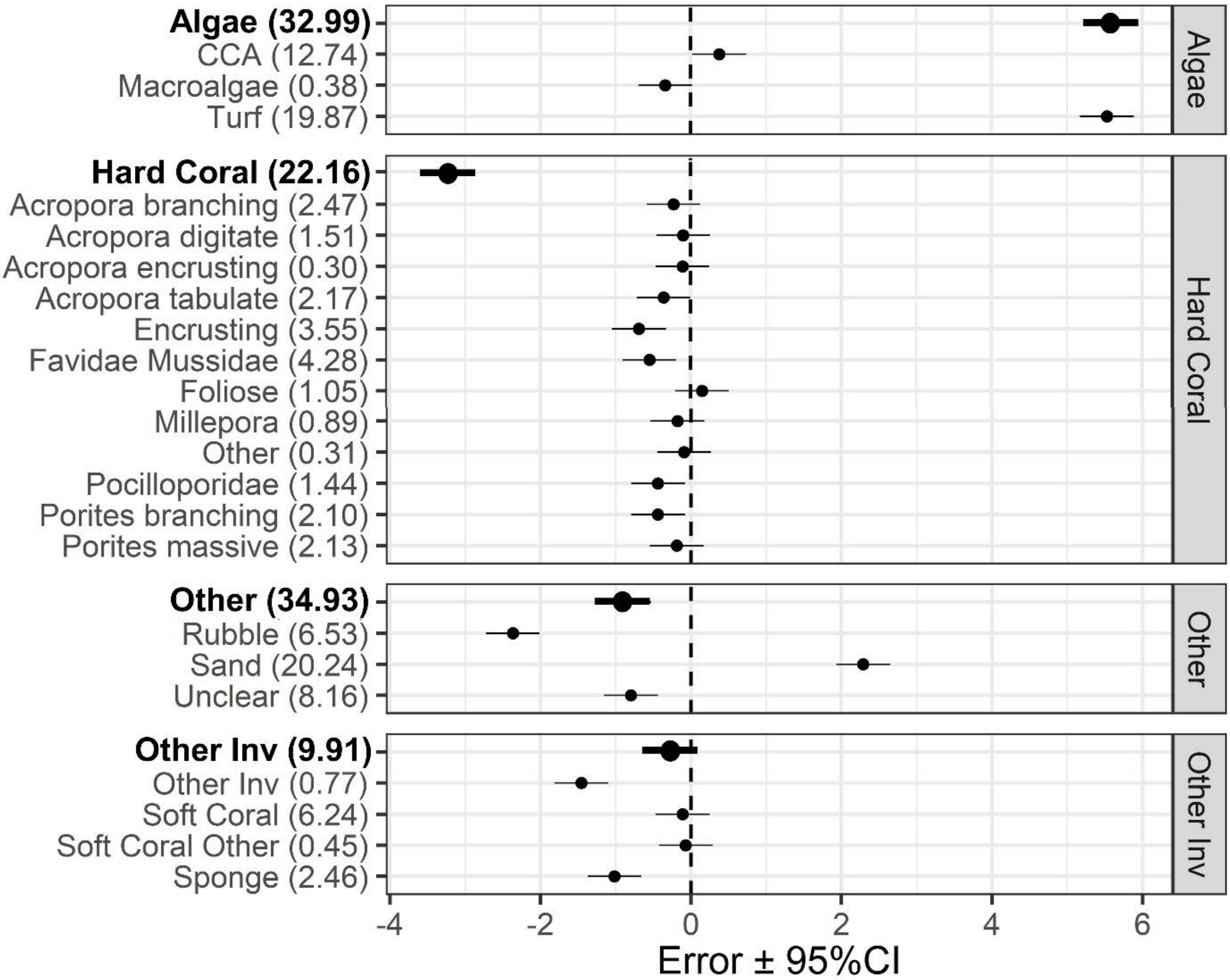


Fig S4. Error from automated benthic estimations of benthic cover in Tonga. Errors are presented for each one of the 22 variables and aggregated by functional groups (“Algae”, “Hard Coral”, “Other” and “Other Inv”). Points represent the mean machine error and error bars indicate the 95% confidence limits. Overall machine accuracy and kappa were 70.0% and 66.6% at the specific category level, respectively, and 80.7% and 73.1% at the functional group level, respectively.

Table S1. Generalized linear model outputs testing for differences in the percent cover of four key benthic categories between the main island groups of Tonga. Data transformations are listed in italics.

Hard coral
Log(x+1)

Variable	Estimate	std. error	t value	p value
Intercept	2.65	0.07	37.99	<0.05
Northern Ha'apai	0.04	0.95	0.45	0.65
Southern Ha'apai	0.64	0.10	6.35	<0.05
Tongatapu	0.39	0.09	4.27	<0.05
Vavau	-0.79	0.08	-10.01	<0.05

Soft Coral
Log(x+1)

Variable	Estimate	std. error	t value	p value
Intercept	2.05	0.07	29.33	<0.05
Northern Ha'apai	-0.51	0.09	-5.36	<0.05
Southern Ha'apai	0.60	0.10	5.96	<0.05
Tongatapu	-0.32	0.09	-3.50	<0.05
Vavau	-1.49	0.08	-18.69	<0.05

CCA
Negative binomial

Variable	Estimate	std. error	t value	p value
Intercept	2.94	0.93	31.71	<0.05
Northern Ha'apai	-0.23	0.13	-1.85	0.06
Southern Ha'apai	0.11	0.13	0.83	0.41
Tongatapu	-0.12	0.12	-0.97	0.33
Vavau	-0.01	0.11	-0.08	0.93

Turf
Log(x+1)

Variable	Estimate	std. error	t value	p value
Intercept	45.96	1.90	24.21	<0.05
Northern Ha'apai	10.20	2.59	3.94	<0.05
Southern Ha'apai	-16.80	2.75	-6.11	<0.05
Tongatapu	1.84	2.52	0.73	0.47
Vavau	16.19	2.17	7.45	<0.05

Table S2. Generalized linear model outputs testing for differences in key reef fish metrics between the main island groups of Tonga.

Richness

Raw data

Variable	Estimate	std. error	t value	p value
Intercept	33.91	0.77	44.25	<0.05
Northern Ha'apai	2.38	1.05	2.26	<0.05
Southern Ha'apai	0.21	1.10	0.19	0.85
Tongatapu	0.21	1.02	0.21	0.83
Vavau	-9.58	0.85	-11.28	<0.05

Density

Log(x)

Variable	Estimate	std. error	t value	p value
Intercept	7.55	0.06	128.63	<0.05
Northern Ha'apai	0.08	0.08	0.97	0.33
Southern Ha'apai	0.01	0.08	0.10	0.92
Tongatapu	-0.03	0.08	-0.34	0.735
Vavau	-0.44	0.07	-6.73	<0.05

Target biomass

Negative binomial

Variable	Estimate	std. error	t value	p value
Intercept	6.36	0.12	54.12	<0.05
Northern Ha'apai	0.45	0.16	2.81	<0.05
Southern Ha'apai	0.33	0.17	1.95	0.051
Tongatapu	0.09	0.16	0.58	0.56
Vavau	-0.53	0.13	-4.09	<0.05

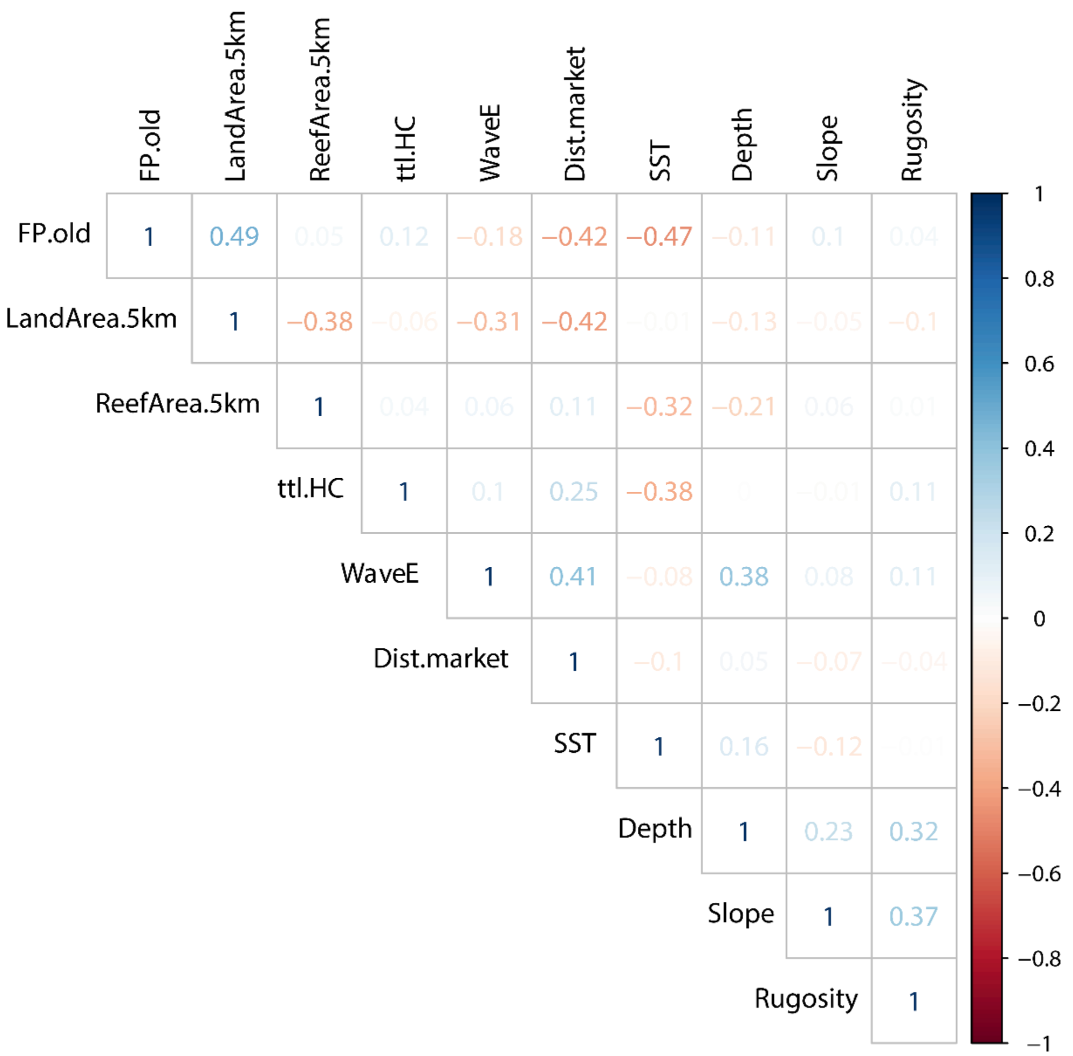


Fig S5. Correlation matrix of socio-environmental variables used to examine patterns of reef condition across Tonga.

Table S3. Boosted regression tree (BRT) parameters used to determine optimal tree complexity, bag fraction and learning rate for each of the four benthic response variables. The model with the greatest explained deviance while containing a minimum of 1000 trees was selected for the subsequent analysis (bold).

Variable	Tree complexity	Learning rate	Bag fraction	CV % Explained deviance	# Trees
Coral cover (poisson distribution)	3	0.01	0.25	36.7	2050
	5	**0.01**	**0.25**	**39.46**	**1150**
	7	0.01	0.25	36.45	1200
	3	0.01	0.5	34.57	1100
	5	0.01	0.5	36.71	600
	7	0.01	0.5	36.46	650
	3	0.01	0.75	34.18	1100
	5	0.01	0.75	37.35	1000
	7	0.01	0.75	40.15	550
	3	0.001	0.25	36.6	9350
	5	0.001	0.25	34.8	9250
	7	0.001	0.25	35.76	8200
	3	0.001	0.5	35.48	7750
	5	0.001	0.5	39.27	6900
	7	0.001	0.5	36.64	4650
	3	0.001	0.75	37.37	9450
	5	0.001	0.75	38.28	7000
	7	0.001	0.75	37.09	6050
CCA (poisson distribution)	3	0.01	0.25	46.67	1850
	5	0.01	0.25	49.09	1450
	7	0.01	0.25	48.92	1700
	3	0.01	0.5	48.89	1900
	5	0.01	0.5	51.79	1100
	7	0.01	0.5	50.02	700
	3	0.01	0.75	50.68	2050
	5	**0.01**	**0.75**	**52.84**	**1250**
	7	0.01	0.75	48.83	700
	3	0.001	0.25	48.27	9650
	5	0.001	0.25	48.28	9600
	7	0.001	0.25	48.46	8750
	3	0.001	0.5	46.11	8650
	5	0.001	0.5	51.06	8350
	7	0.001	0.5	50.17	6450
	3	0.001	0.75	48.63	9450
	5	0.001	0.75	48.47	6950
	7	0.001	0.75	50.82	7650
Soft coral (poisson distribution)	3	0.01	0.25	56.69	1350
	5	0.01	0.25	56.62	600
	7	0.01	0.25	53.92	550

	3	0.01	0.5	57.05	1750
	5	0.01	0.5	59.1	950
	7	0.01	0.5	58.32	750
	3	0.01	0.75	55.04	1150
	5	0.01	0.75	56.68	650
	7	0.01	0.75	57.38	700
	3	0.001	0.25	55.99	9450
	5	0.001	0.25	55.47	6250
	7	0.001	0.25	54.07	4650
	3	0.001	0.5	56.47	8600
	5	**0.001**	**0.5**	**58.22**	**7400**
	7	0.001	0.5	56.91	5900
	3	0.001	0.75	54.75	8450
	5	0.001	0.75	56.74	6050
	7	0.001	0.75	55.55	6150
Turf algae (gaussian distribution)	3	0.01	0.25	51.74	550
	5	0.01	0.25	50.94	550
	7	0.01	0.25	51.13	400
	3	0.01	0.5	52.73	550
	5	0.01	0.5	51.51	400
	7	0.01	0.5	51.83	350
	3	**0.01**	**0.75**	**53.84**	**1000**
	5	0.01	0.75	51.99	600
	7	0.01	0.75	50.09	350
	3	0.001	0.25	51.93	5450
	5	0.001	0.25	51.85	5000
	7	0.001	0.25	52.42	4650
	3	0.001	0.5	50.55	4650
	5	0.001	0.5	52.53	4300
	7	0.001	0.5	51.79	3600
	3	0.001	0.75	51.19	4750
	5	0.001	0.75	51.73	4900
	7	0.001	0.75	49.6	3350

Table S4. Boosted regression tree (BRT) parameters used to determine optimal tree complexity, bag fraction and learning rate for each of the three fish response variables. The model with the greatest explained deviance while containing a minimum of 1000 trees was selected for the subsequent analysis (bold).

Variable	Tree complexity	Learning rate	Bag fraction	CV % Explained deviance	# Trees
Reef fish density (poisson distribution)	3	0.01	0.25	27.41	900
	5	0.01	0.25	24.42	500
	7	0.01	0.25	25.59	700
	3	0.01	0.5	27.47	650
	5	0.01	0.5	29.86	450
	7	0.01	0.5	27.92	350
	3	0.01	0.75	27.93	1859
	5	0.01	0.75	29.17	600
	7	0.01	0.75	29.57	450
	3	0.001	0.25	25.43	5450
	5	0.001	0.25	26.29	4100
	7	0.001	0.25	23.73	3800
	3	0.001	0.5	25.19	5550
	5	0.001	0.5	29.15	4150
	7	0.001	0.5	29.22	3450
	3	0.001	0.75	28.22	5800
	5	0.001	0.75	27.17	3300
	7	**0.001**	**0.75**	**29.55**	**3650**
Reef fish species richness (gaussian distribution)	3	0.01	0.25	66.98	1500
	5	0.01	0.25	66.86	1900
	7	0.01	0.25	67.82	1600
	3	0.01	0.5	68.17	2300
	5	0.01	0.5	69.2	1350
	7	**0.01**	**0.5**	**69.46**	**1100**
	3	0.01	0.75	67.13	1750
	5	0.01	0.75	68.44	1250
	7	0.01	0.75	68.88	900
	3	0.001	0.25	Did not converge	10000+
	5	0.001	0.25	66.9	7700
	7	0.001	0.25	66.84	7350
	3	0.001	0.5	66.25	9150
	5	0.001	0.5	67.41	7300
	7	0.001	0.5	68.97	6500
	3	0.001	0.75	Did not converge	10000+
	5	0.001	0.75	68.39	7850
	7	0.001	0.75	67.36	6050
Target biomass (poisson distribution)	3	0.01	0.25	43.1	450
	5	0.01	0.25	45.13	700

7	0.01	0.25	47.35	650
3	0.01	0.5	44.13	1750
5	0.01	0.5	44.23	400
7	0.01	0.5	45.83	500
3	0.01	0.75	41.04	1150
5	0.01	0.75	41.12	350
7	0.01	0.75	42.53	300
3	0.001	0.25	43.73	4400
5	**0.001**	**0.25**	**46.27**	**4400**
7	0.001	0.25	45.17	3950
3	0.001	0.5	44.58	7000
5	0.001	0.5	44.26	4050
7	0.001	0.5	44.71	3250
3	0.001	0.75	40.41	4150
5	0.001	0.75	43.35	4650
7	0.001	0.75	42.44	3600

Table S5. Summary of BRT model performance and spatial autocorrelation. As in Jouffray et al. (2019), model performances assessed on training data (used for model fitting) were higher than when assessed on the left out data (cross-validated). The cross-validated performance indicated how good the model is at predicting new data (Buston and Elith 2011). Moran's I measures spatial autocorrelation and ranges from -1 to 1, with values close to 0 indicating no spatial autocorrelation.

	Coral	CCA	Soft	Turf	Dens	Rich	BM
Total.Deviance	11.31	16.73	9.25	674.28	766.88	102.91	651.64
Residual.Deviance	2.89	3.03	1.89	206.10	281.71	7.31	231.01
Correlation	0.87	0.94	0.93	0.84	0.77	0.97	0.71
Per.Expl	74.47	81.92	79.55	69.43	63.27	92.90	64.55
cvDeviance	7.12	8.19	3.90	332.07	540.29	32.57	353.61
cvCorrelation	0.58	0.69	0.78	0.71	0.56	0.84	0.63
cvPer.Expl	37.04	51.06	57.83	53.84	29.55	68.35	45.73
Morans I	0.01	-0.01	0.07	-0.01	0.01	-0.04	0.01

Table S6. Reef fish species list for Tonga from the 375 sites surveyed between 2016 and 2019. Species in bold were included as target species in the analysis based on Parks et al. (2017)

Abudefduf sexfasciatus
Abudefduf sordidus
Abudefduf vaigiensis
Acanthurus achilles
Acanthurus albipectoralis
Acanthurus blochii
Acanthurus grammoptilus
Acanthurus guttatus
Acanthurus lineatus
Acanthurus maculiceps
Acanthurus nigricans
Acanthurus nigricauda
Acanthurus nigrofuscus
Acanthurus nigroris
Acanthurus olivaceus
Acanthurus pyroferus
Acanthurus thompsoni
Acanthurus triostegus
Acanthurus xanthopterus
Acanthurus dussumieri
Aetobatus ocellatus
Aluterus scriptus
Amanses scopas
Amblyeleotris fasciata
Amblyeleotris guttata
Amblyeleotris periophthalma
Amblyeleotris steinitzi
Amblyglyphidodon aureus
Amblyglyphidodon melanopterus
Amblygobius nocturnus
Amblygobius phalaena
Amblygobius rainfordi
Amphiprion chrysopterus
Amphiprion clarkii
Amphiprion melanopus
Amphiprion pacificus
Amphiprion perideraion
Anampses caeruleopunctatus
Anampses geographicus
Anampses melanurus
Anampses meleagrides
Anampses neoguinaicus
Anampses twistii
Anyperodon leucogrammicus
Aphareus furca
Apogon cyanosoma
Apogon hyallosoma
Apogon leptacanthus
Apogon luteus
Apogon nigrofasciatus
Aprion virescens
Arothron caeruleopunctatus
Arothron hispidus
Arothron manilensis
Arothron meleagris
Arothron nigropunctatus
Arothron stellatus
Aspidontus taeniatus
Asterropteryx semipunctata
Aulostomus chinensis
Balistapus undulatus
Balistoides conspiculum
Balistoides viridescens
Blenniella chrysospilos
Bodianus axillaris
Bodianus dictynna
Bodianus loxozonus
Bodianus mesothorax
Bothus mancus
Bryaninops yongei
Caesio caerulaurea
Caesio lunaris
Caesio teres
Calostomus coralinus
Cantherhines dumerilii
Cantherhines fronticinctus
Cantherhines pardalis
Canthigaster ambionensis
Canthigaster axiologa
Canthigaster bennetti
Canthigaster solandri
Canthigaster valentini
Caracanthus maculatus
Carangoides ferdau
Caranx ignobilis
Caranx melampygus
Caranx sexfasciatus
Carcharhinus amblyrhynchos
Centropyge bicolor
Centropyge bispinosus
Centropyge flavissima
Centropyge heraldi
Cephalopholis argus
Cephalopholis leopardus
Cephalopholis miniata
Cephalopholis sonnerati
Cephalopholis urodeta
Cetoscarus bicolor
Cetoscarus ocellatus
Chaetodon auriga
Chaetodon baronessa
Chaetodon bennetti
Chaetodon citrinellus
Chaetodon ephippium
Chaetodon flavirostris
Chaetodon kleinii
Chaetodon lineolatus
Chaetodon lunula
Chaetodon lunulatus
Chaetodon melanotus
Chaetodon mertensii
Chaetodon meyeri
Chaetodon ornatissimus
Chaetodon pelewensis
Chaetodon plebeius
Chaetodon quadrimaculatus
Chaetodon rafflesi
Chaetodon reticulatus
Chaetodon semeion
Chaetodon trifascialis
Chaetodon ulietensis
Chaetodon unimaculatus
Chaetodon vagabundus
Cheilinus chlorourus
Cheilinus fasciatus
Cheilinus oxycephalus
Cheilinus trilobatus
Cheilinus undulatus
Cheilio inermis
Cheiliodipterus artus
Cheilodipterus artus
Cheilodipterus isostigmus
Cheilodipterus macrodon
Cheilodipterus quinquelineatus
Chlorurus bleekeri
Chlorurus frontalis
Chlorurus japanensis
Chlorurus microrhinus
Chlorurus sordidus
Choerodon jordani
Chromis acares
Chromis agilis
Chromis alpha
Chromis amboinensis
Chromis analis
Chromis atripectoralis
Chromis atripes
Chromis bami
Chromis chrysura
Chromis flavapicis
Chromis flavomaculata
Chromis iomelas
Chromis lepidolepis
Chromis margaritifer
Chromis opercularis
Chromis retrofasciata
Chromis ternatensis
Chromis tricincta
Chromis vanderbilti
Chromis viridis
Chromis weberi
Chromis xanthura
Chrysiptera biocellata
Chrysiptera brownriggii
Chrysiptera rollandi
Chrysiptera starcki
Chrysiptera talboti
Chrysiptera taupou
Chrysiptera unimaculata
Cirrhilabrus punctatus
Cirrhitichthys falco
Cirripectes chelomatus
Cirripectes fuscoguttatus
Cirripectes polyzona
Cirripectes stigmaticus
Coris aygula
Coris batuensis
Coris dorsomacula
Coris gaimard
Corythoichthys intestinalis
Ctenochaetus binotatus
Ctenochaetus cyanocheilus
Ctenochaetus flavicauda
Ctenochaetus hawaiiensis

Ctenochaetus striatus
Ctenogobiops aurocingulus
Dascyllus aruanus
Dascyllus reticulatus
Dascyllus trimaculatus
Dasyatis kuhlii
Diodon holocanthus
Diodon hystrix
Echidna nebulosa
Ecsenius bicolor
Ecsenius flavus
Ecsenius midas
Elagatis bipinnulata
Epibulus insidiator
Epinephelus fasciatus
Epinephelus hexagonatus
Epinephelus howlandi
Epinephelus macrospilos
Epinephelus maculatus
Epinephelus malabaricus
Epinephelus merra
Epinephelus ongus
Epinephelus pasciatus
Epinephelus polyphekadion
Epinephelus spilotoceps
Exallias brevis
Exyrias bellisimus
Fistularia commersonii
Forcipiger flavissimus
Forcipiger longirostris
Fusigobius signipinnis
Gnathanodon speciosus
Gnathodentex aureolineatus
Gobiodon citrinus
Gomphosus varius
Grammistes sexlineatus
Gymnocranius euanus
Gymnocranius microdon
Gymnosarda unicolor
Gymnothorax buroensis
Gymnothorax flavimarginatus
Gymnothorax javanicus
Gymnothorax meleagris
Gymnothorax nudivomer
Halichoeres argus
Halichoeres biocellatus
Halichoeres chrysus
Halichoeres hortulanus
Halichoeres margaritaceus
Halichoeres marginatus
Halichoeres melanochir
Halichoeres melanurus
Halichoeres nebulosus
Halichoeres ornatissimus
Halichoeres prosopeion
Halichoeres trimaculatus
Hemigymnus fasciatus
Hemigymnus melapterus
Hemitaurichthys polylepis
Heniochus acuminatus
Heniochus chrysostomus
Heniochus monoceros
Heniochus singularis
Heniochus varius
Hipposcarus longiceps
Hologymnosus annulatus
Hologymnosus doliatus
Istigobius goldmanni
Istigobius rigilius
Koumansetta rainfordi
Kyphosus cinerascens
Kyphosus vaigiensis
Labrichthys unilineatus
Labroides bicolor
Labroides dimidiatus
Labroides pectoralis
Labroides rubrolabiatus
Labropsis australis
Leptoscarus vaigiensis
Lethrinus harak
Lethrinus nebulosus
Lethrinus obsoletus
Lethrinus olivaceus
Lutjanus biguttatus
Lutjanus bohar
Lutjanus carponotatus
Lutjanus ehrenbergii
Lutjanus fulviflamma
Lutjanus fulvus
Lutjanus gibbus
Lutjanus kasmira
Lutjanus malabaricus
Lutjanus monostigma
Lutjanus quinquelineatus
Macolor macularis
Macolor niger
Macropharyngodon meleagris
Macropharyngodon negrosensis
Malacanthus brevirostris
Malacanthus latovittatus
Meiacanthus atrodorsalis
Meiacanthus bundoon
Meiacanthus ditrema
Meiacanthus procne
Meiacanthus tongaensis
Melichthys niger
Melichthys vidua
Monotaxis grandoculis
Monotaxis heterodon
Mulloidichthys flavolineatus
Mulloidichthys pflugeri
Mulloidichthys vanicolensis
Myripristis adusta
Myripristis berndti
Myripristis kuntee
Myripristis murdjan
Myripristis violacea
Myripristis vittata
Naso brachycentron
Naso brevirostris
Naso caesius
Naso hexacanthus
Naso lituratus
Naso lopezi
Naso tonganus
Naso unicornis
Nectamia fusca
Nemateleotris magnifica
Neocirrhitus armatus
Neoglyphidodon carlsoni
Neoniphon argenteus
Neoniphon opercularis
Neoniphon sammara
Neopomacentrus azyron
Neopomacentrus metalicus
Novaculichthys taeniourus
Novaculoides macrolepidotus
Odonus niger
Ostorhinchus aureus
Ostorhinchus cookii
Ostorhinchus cyanosoma
Ostorhinchus aureus
Ostracion cubicus
Ostracion meleagris
Oxycheilinus arenatus
Oxycheilinus celebicus
Oxycheilinus digramma
Oxycheilinus nigromarginatus
Oxycheilinus orientalis
Oxycheilinus rhodochrous
Oxycheilinus unifasciatus
Oxymonacanthus longirostris
Paracanthurus hepatus
Paracirrhites arcatus
Paracirrhites forsteri
Paracirrhites hemistictus
Paraluteres prionurus
Parapercis australis
Parapercis clathrata
Parapercis cylindrica
Parapercis hexophthalma
Parapercis millepunctata
Parupeneus barberinoides
Parupeneus barberinus
Parupeneus bifasciatus
Parupeneus ciliatus
Parupeneus crassilabris
Parupeneus cyclostomus
Parupeneus indicus
Parupeneus multifasciatus
Parupeneus pleurostigma
Parupeneus spilurus
Pempheris oualensis
Pervagor alternans
Pervagor aspricaudus
Pervagor janthinosoma
Pervagor melanocephalus
Petroscirtes mitratus
Plagiotremus flavus
Plagiotremus laudandus
Plagiotremus rhinorhynchos
Plagiotremus tapeinosoma
Platax boersii
Platax teira

Plectorhinchus chaetodonoides
Plectorhinchus lineatus
Plectorhinchus picus
Plectroglyphidodon dickii
Plectroglyphidodon imparipennis
Plectroglyphidodon johnstonianus
Plectroglyphidodon lacrymatus
Plectroglyphidodon leucozonus
Plectropomus laevis
Plectropomus leopardus
Plotosus lineatus
Pomacanthus imperator
Pomacentrus adelus
Pomacentrus amboinensis
Pomacentrus brachialis
Pomacentrus callainus
Pomacentrus chrysurus
Pomacentrus coelestis
Pomacentrus imitator
Pomacentrus maafu
Pomacentrus margaritifer
Pomacentrus microspilus
Pomacentrus moluccensis
Pomacentrus pavo
Pomacentrus philippinus
Pomacentrus spilotoceps
Pomacentrus vaiuli
Pomacentrus wardi
Pomachromis richardsoni
Priacanthus arenatus
Priacanthus blochii
Priacanthus hamrur
Pristiapogon exostigma
Pristiapogon fraenatus
Pristiapogon kallopterus
Pseudanthias dispar
Pseudanthias pleurotaenia
Pseudanthias squamipinnis
Pseudobalistes flavimarginatus
Pseudobalistes fuscus
Pseudocheilinus evanidus
Pseudocheilinus hexataenia
Pseudocheilinus octotaenia
Pseudocoris heteroptera
Pseudocoris yamashiroi
Pseudojuloides cerasinus
Pteragogus cryptus
Ptereleotris evides
Ptereleotris hanae
Ptereleotris heteroptera
Ptereleotris microlepis
Ptereleotris monoptera
Pterocaesio digramma
Pterocaesio marri
Pterocaesio tile
Pterocaesio trilineata
Pterois radiata
Pterois volitans
Pygoplites diacanthus
Rastrelliger kanagurta
Rhinecanthus aculeatus
Rhinecanthus rectangulus
Salarias fasciatus
Salarias nigrocinctus
Sargocentron caudimaculatum
Sargocentron diadema
Sargocentron ittodai
Sargocentron melanospilos
Sargocentron spiniferum
Sargocentron tiere
Sargocentron violaceum
Saurida gracillis
Scarus altipinnis
Scarus chamaeleon
Scarus dimidiatus
Scarus flavipectoralis
Scarus forsteni
Scarus frenatus
Scarus ghobban
Scarus globiceps
Scarus longipinnis
Scarus niger
Scarus oviceps
Scarus psittacus
Scarus rivulatus
Scarus rubrovialaceous
Scarus schlegeli
Scarus spinus
Scarus tricolor
Scolopsis bilineatus
Scolopsis lineatus
Scolopsis trilineata
Scomberoides lysan
Scorpaenopsis macrochir
Sebastapistes cyanostigma
Siderea thyrsoidea
Siganus argenteus
Siganus doliatus
Siganus niger
Siganus punctatus
Siganus spinus
Siganus stellatus
Siganus vulpinus
Siphamia jebbi
Sphyraena barracuda
Sphyraena helleri
Stegastes albifasciatus
Stegastes fasciolatus
Stegastes lividus
Stegastes nigricans
Stegastes punctatus
Stethojulis bandanensis
Stethojulis notialis
Stethojulis strigiventer
Sufflamen bursa
Sufflamen chrysopterum
Synchiropus splendidus
Synodus binotatus
Synodus dermatogenys
Synodus variegatus
Taeniamia fucata
Thalassoma amblycephalum
Thalassoma hardwicke
Thalassoma jansenii
Thalassoma lunare
Thalassoma lutescens
Thalassoma nigrofasciatum
Thalassoma quinquevittatum
Thalossoma purpureum
Triaenodon obesus
Valenciennea immaculata
Valenciennea parva
Valenciennea puellaris
Valenciennea sexguttata
Valenciennea strigata
Variola louti
Zanclus cornutus
Zebrasoma scopas
Zebrasoma veliferum
Zoramia fragilis
Zoramia leptacantha
Zoramia viridiventer

Supplementary Materials: Chapter 5

Table S1. Growth of Tonga's Special Management Area (SMA) program from the first SMA and Fish Habitat Reserve (FHR) in 2006 to May 2019.

Name	Year Established	Island group	Area (km2)	Reef area (km2)
Oua FHR	2006	Ha'apai	2.16	0.343
Oua SMA	2006	Ha'apai	41.68	12.1731
Ha'afeva FHR 1	2007	Ha'apai	0.44	0.0981
Ha'afeva FHR 2	2007	Ha'apai	0.95	0.3674
Ha'afeva SMA	2007	Ha'apai	14.3	3.3236
Atata FHR	2008	Tongatapu	1.54	0.7293
Atata SMA	2008	Tongatapu	8.4	2.7796
Euiki FHR 1 Tongatapu	2008	Tongatapu	0.5	0.4589
Euiki FHR 2 Tongatapu	2008	Tongatapu	0.37	0.2408
Euiki SMA	2008	Tongatapu	3.75	1.0188
Felemea FHR 1	2008	Ha'apai	0.44	0.0801
Felemea FHR 2	2008	Ha'apai	0.74	0.1205
Felemea SMA	2008	Ha'apai	17.1	3.8867
Ovaka FHR	2008	Vava'u	2.6	1.2198
Ovaka SMA	2008	Vava'u	9.21	1.3825
Nomuka FHR	2011	Ha'apai	0.53	0.3456
Nomuka SMA	2011	Ha'apai	68.2	12.7057
Taunga FHR	2013	Vava'u	1.21	0.2156
Taunga SMA	2013	Vava'u	7.74	0.439
Fafa FHR	2014	Tongatapu	1.59	1.2717
Kolonga FHR 1	2015	Tongatapu	0.15	0.1065
Kolonga FHR 2	2015	Tongatapu	0.7	0.4455
Kolonga SMA	2015	Tongatapu	1.64	0.5466
Kotu FHR 1	2015	Ha'apai	3.02	1.6214
Kotu FHR 2	2015	Ha'apai	0.19	0.0613
Kotu SMA	2015	Ha'apai	16.86	5.34
Lapaha FHR	2016	Tongatapu	0.19	0
Lapaha SMA	2016	Tongatapu	1.1	0
Nukuleka FHR	2016	Tongatapu	0.51	0
Nukuleka SMA	2016	Tongatapu	2.63	0
Eueiki FHR Vavau	2017	Vava'u	1.19	0.278
Falevai FHR	2017	Vava'u	0.36	0.1506
Falevai SMA	2017	Vava'u	3.98	0.1283
Fonoi FHR	2017	Ha'apai	1.91	0.4467
Fonoi SMA	2017	Ha'apai	22.33	4.027
Ha'atafu FHR 1	2017	Tongatapu	0.17	0.1557
Ha'atafu FHR 2	2017	Tongatapu	0.24	0.1194
Ha'atafu SMA	2017	Tongatapu	5.35	2.7495
Holonga FHR	2017	Tongatapu	0.3	0
Holonga SMA	2017	Tongatapu	0.93	0
Hunga FHR 1	2017	Vava'u	1.46	0.8812
Hunga FHR 2	2017	Vava'u	1.32	0.0117
Hunga SMA	2017	Vava'u	20.73	5.0914
Koloa FHR 1	2017	Vava'u	0.06	0
Koloa FHR 2	2017	Vava'u	0.2	0
Koloa SMA	2017	Vava'u	4.52	0.9991
Lape FHR	2017	Vava'u	0.58	0.1115
Lape SMA	2017	Vava'u	1.98	0.0915
Mango FHR	2017	Ha'apai	2.78	0.9053
Mango SMA	2017	Ha'apai	39.75	7.1316
Matuku FHR	2017	Ha'apai	0.55	0.1333
Matuku SMA	2017	Ha'apai	16.89	1.9162
Ofu FHR 1	2017	Vava'u	0.29	0.1363
Ofu FHR 2	2017	Vava'u	0.38	0
Ofu SMA	2017	Vava'u	4.93	0.5324
Pangaimotu FHR	2017	Tongatapu	1.4	0.607
Talihau FHR	2017	Vava'u	0.36	0.0025
Talihau SMA	2017	Vava'u	2.52	0.1206
Utulei FHR	2017	Vava'u	0.21	0.0533

Utulei SMA	2017	Vava’u	4.16	0.2585
Utungake FHR	2017	Vava’u	1.08	0.0506
Utungake SMA	2017	Vava’u	2.34	0.0849
Fakakakai FHR	2018	Ha’apai	0.94	0.2859
Fakakakai SMA	2018	Ha’apai	10.74	1.8502
Faleloa FHR 1	2018	Ha’apai	0.45	0.0252
Faleloa FHR 2	2018	Ha’apai	0.25	0.0903
Faleloa SMA	2018	Ha’apai	15.83	2.7593
Ha'ano FHR	2018	Ha’apai	0.87	0.3132
Ha'ano SMA	2018	Ha’apai	11.96	1.1966
Kelefesia FHR	2018	Ha’apai	1.31	0.9353
Kelefesia SMA	2018	Ha’apai	32.72	8.8818
Lofanga FHR 1	2018	Ha’apai	0.36	0.2479
Lofanga FHR 2	2018	Ha’apai	0.45	0.2655
Lofanga SMA	2018	Ha’apai	14.83	1.8547
Muitoa FHR	2018	Ha’apai	0.72	0.1655
Muitoa SMA	2018	Ha’apai	10.81	1.3336
Ofolanga FHR 1	2018	Ha’apai	1.8	0.3562
Ofolanga FHR 2	2018	Ha’apai	1.2	0.2274
Ofolanga SMA	2018	Ha’apai	40.7	5.3858
Pukotala FHR	2018	Ha’apai	0.23	0.0808
Pukotala SMA	2018	Ha’apai	5.68	0.7384
Tufuva FHR	2018	Eua	0.33	0.2473
Uiha FHR 1	2018	Ha’apai	0.37	0.0798
Uiha FHR 2	2018	Ha’apai	0.46	0.0382
Uiha SMA	2018	Ha’apai	17.09	3.083
Holoeva FHR	2019	Vava’u	0.25	0.0561
Holoeva SMA	2019	Vava’u	1.5	0.49
Houma FHR 1	2019	Eua	0.58	0.3201
Houma FHR 2	2019	Eua	0.23	0.1243
Houma SMA	2019	Eua	17.48	1.3355
Kapa FHR	2019	Vava’u	0.58	0.0736
Kapa SMA	2019	Vava’u	2.33	0.1713
Tufuva SMA	2019	Eua	7.24	0.6678

Table S2. List of proposed Tonga SMAs and their current status from October 2019.

Name	Status	Island group
Ahau SMA 1	Confirmed	Tongatapu
Ahau SMA 2	Confirmed	Tongatapu
Alaki SMA	Proposed	Tongatapu
Falaleu SMA	Confirmed	Vava'u
Fatai SMA	Proposed	Tongatapu
Folaha SMA	Proposed	Tongatapu
Ha'akio, Ta'anea and Tu'anikivale SMA	Proposed	Vava'u
Ha'ateiho SMA	Proposed	Tongatapu
Hoi SMA	Confirmation required	Tongatapu
Holopeka SMA	Confirmed	Ha'apai
Kanokupolu FHR 1	Submitted to cabinet	Tongatapu
Kanokupolu FHR 2	Submitted to cabinet	Tongatapu
Kanokupolu SMA 1	Submitted to cabinet	Tongatapu
Kanokupolu SMA 2	Submitted to cabinet	Tongatapu
Kolomotua FHR	Letter of interest	Tongatapu
Kolomotua/Sopu SMA	Letter of interest	Tongatapu
Kolovai/Foui SMA 1	Proposed	Tongatapu
Kolovai/Foui SMA 2	Proposed	Tongatapu
Koulo SMA	Confirmed	Ha'apai
Lifuka/Hihifo SMA	Confirmed	Ha'apai
Longomapu SMA	Confirmed	Vava'u
Longoteme SMA	Proposed	Tongatapu
Makaunge SMA	Confirmed	Tongatapu
Makave FHR 1	Submitted to cabinet	Vava’u
Makave FHR 2	Submitted to cabinet	Vava’u
Makave SMA	Submitted to cabinet	Vava’u
Malapo SMA	Proposed	Tongatapu
Manuka FHR	Submitted to cabinet	Tongatapu
Manuka SMA	Submitted to cabinet	Tongatapu
Masilamea SMA	Proposed	Tongatapu
Matamaka FHR 1	Confirmed	Vava’u
Matamaka FHR 2	Confirmed	Vava’u
Matamaka SMA	Confirmed	Vava’u
Navutoka SMA	Confirmation required	Tongatapu
Nuapapu FHR 1	Submitted to cabinet	Vava’u
Nuapapu FHR 2	Submitted to cabinet	Vava’u
Nuapapu SMA	Submitted to cabinet	Vava’u
Nukumotu FHR	Cabinet decision required	Tongatapu
Nukumotu SMA	Cabinet decision required	Tongatapu
Nukunuku FHR	Submitted to cabinet	Tongatapu
Nukunuku SMA	Submitted to cabinet	Tongatapu
Oko'a, Houma, Mangia and Utui SMA	Letter of interest	Vava'u
Olo'ua SMA	Proposed	Vava'u
Otea SMA	Proposed	Vava'u
Pangaimotu SMA	Confirmed	Vava'u
Pea SMA	Proposed	Tongatapu
Puke SMA	Confirmation required	Tongatapu
Sopu FHR	Letter of interest	Tongatapu
Takamotonga SMA	Proposed	Tongatapu
Talafoou SMA	Confirmed	Tongatapu
Te'ekiu SMA	Letter of interest	Tongatapu
Tefisi SMA	Proposed	Vava'u
Tofoa SMA	Confirmed	Tongatapu
Toula SMA	Confirmed	Vava'u
Vaini SMA	Proposed	Tongatapu
Vaipua SMA	Confirmed	Vava'u
Veitongo SMA	Proposed	Tongatapu

Supplementary Materials: Chapter 6

Table S1. Fish Habitat Reserve (FHR) covariate balance pre- and post- matching. The first column lists the covariates used to match control and FHR transects, and for each covariate match statistics are provided before and after matching, indicated in the 'unmatched' and 'matched' rows, to show how well the matching model performed. The third and fourth column present mean covariate values for FHR and control transects. The fifth column shows the mean difference between FHR and control means. The sixth and seventh column respectively show mean and maximum differences in each covariate Quantile – Quantile (QQ plot), with lower values indicating a better match. The lower table shows the total, matched and unmatched number of control and FHR transects respectively following the matching procedure.

Fish Habitat Reserve

Variable		**Treatment mean**	**Control mean**	**std. mean diff.**	**mean eQQ diff.**	**max eQQ diff.**
Depth	*unmatched*	4.82	6.37	-1.55	1.55	4.00
	matched	4.86	4.91	-0.04	0.92	4.00
Distance to land	*unmatched*	0.70	0.51	0.19	0.35	3.41
	matched	0.68	0.70	-0.03	0.31	3.41
Distance to village	*unmatched*	1.74	4.67	-2.92	3.31	56.25
	matched	1.78	2.50	-0.73	4.55	52.24
Fishing pressure	*unmatched*	24.95	21.77	3.17	6.41	33.81
	matched	26.12	31.14	-5.02	6.52	28.78
Habitat - Fringing	*unmatched*	0.55	0.34	0.21	0.21	1.00
	matched	0.55	0.55	0.00	0.26	1.00
Habitat - Semi exposed	*unmatched*	0.13	0.30	-0.18	0.17	1.00
	matched	0.09	0.09	0.00	0.02	1.00
Habitat Exposed	*unmatched*	0.32	0.36	-0.03	0.04	1.00
	matched	0.36	0.36	0.00	0.23	1.00
Island Ha'apai	*unmatched*	0.54	0.28	0.26	0.26	1.00
	matched	0.49	0.49	0.00	0.20	1.00
Island Tongatapu	*unmatched*	0.28	0.14	0.14	0.14	1.00
	matched	0.31	0.31	0.00	0.14	1.00
Island Vava'u	*unmatched*	0.18	0.59	-0.40	0.41	1.00
	matched	0.20	0.20	0.00	0.33	1.00
Live coral cover	*unmatched*	19.44	17.47	1.98	2.88	21.31
	matched	20.15	24.32	-4.17	3.09	1.00
Rugosity	*unmatched*	2.76	3.03	-0.27	0.27	1.00
	matched	2.85	2.90	-0.05	0.13	1.00
Slope	*unmatched*	2.72	2.84	-0.12	0.18	1.00
	matched	2.74	2.93	-0.19	0.19	1.00
Surveyor 1	*unmatched*	0.92	0.63	0.29	0.29	1.00
	matched	0.95	0.95	0.00	0.17	1.00
Surveyor 2	*unmatched*	0.08	0.18	-0.09	0.09	1.00
	matched	0.05	0.05	0.00	0.17	1.00
Surveyor 3	*unmatched*	0.00	0.08	-0.08	0.08	1.00
	matched	0.00	0.00	0.00	0.00	0.00
Surveyor 4	*unmatched*	0.00	0.11	-0.11	0.11	1.00
	matched	0.00	0.00	0.00	0.00	0.00
Wave energy	*unmatched*	307.31	333.96	-26.64	108.50	2179.56
	matched	310.87	385.86	-74.99	228.82	2179.57

	Control	Treated
All transects	1285	143
Matched transects	247	129
Unmatched transects	1038	14

Table S2. Special Management Area (SMA) covariate balance pre- and post- matching. The first column lists the covariates used to match control and SMA transects, and for each covariate match statistics are provided before and after matching, indicated in the 'unmatched' and 'matched' rows, to show how well the matching model performed. The third and fourth column present mean covariate values for SMA and control transects. The fifth column shows the mean difference between SMA and control means. The sixth and seventh column respectively show mean and maximum differences in each covariate Quantile – Quantile (QQ plot), with lower values indicating a better match. The lower table shows the total, matched and unmatched number of control and SMA transects respectively following the matching procedure.

		Special Management Area				
Variable		**Treatment mean**	**Control mean**	**std. mean diff.**	**mean eQQ diff.**	**max eQQ diff.**
Depth	*unmatched*	5.82	6.37	-0.54	0.62	2.80
	matched	5.41	5.54	-0.13	0.39	2.50
Distance to land	*unmatched*	0.58	0.51	0.07	0.17	3.03
	matched	0.64	0.99	-0.35	0.23	3.02
Distance to village	*unmatched*	1.37	4.67	-3.30	3.43	54.65
	matched	1.40	1.70	-0.29	0.43	3.61
Fishing pressure	*unmatched*	15.66	21.77	-6.11	6.84	37.41
	matched	15.98	18.28	-2.30	6.60	27.34
Habitat - Fringing	*unmatched*	0.18	0.34	-0.16	0.16	1.00
	matched	0.17	0.17	0.00	0.06	1.00
Habitat - Semi exposed	*unmatched*	0.45	0.30	0.15	0.15	1.00
	matched	0.49	0.49	0.00	0.03	1.00
Habitat Exposed	*unmatched*	0.36	0.35	0.01	0.01	1.00
	matched	0.33	0.33	0.00	0.09	1.00
Island Ha'apai	*unmatched*	0.72	0.28	0.44	0.44	1.00
	matched	0.69	0.69	0.00	0.31	1.00
Island Tongatapu	*unmatched*	0.12	0.14	-0.02	0.02	1.00
	matched	0.12	0.12	0.00	0.09	1.00
Island Vava'u	*unmatched*	0.16	0.58	-0.04	0.43	1.00
	matched	0.18	0.18	0.00	0.40	1.00
Live coral cover	*unmatched*	20.60	17.54	3.06	3.59	29.60
	matched	19.59	21.54	-1.95	2.95	7.74
Rugosity	*unmatched*	2.76	3.04	-0.27	0.32	1.00
	matched	2.78	2.72	0.05	0.22	1.00
Slope	*unmatched*	2.62	2.84	-0.22	0.22	1.00
	matched	2.64	2.65	-0.01	0.11	1.00
Surveyor 1	*unmatched*	0.76	0.63	0.13	0.13	1.00
	matched	0.84	0.84	0.00	0.01	1.00
Surveyor 2	*unmatched*	0.15	0.18	-0.03	0.03	1.00
	matched	0.06	0.06	0.00	0.06	1.00
Surveyor 3	*unmatched*	0.00	0.08	-0.08	0.08	1.00
	matched	0.00	0.00	0.00	0.00	0.00
Surveyor 4	*unmatched*	0.09	0.11	-0.02	0.02	1.00
	matched	0.09	0.09	0.00	0.02	1.00
Wave energy	*unmatched*	346.75	334.15	12.59	81.96	1450.55
	matched	311.00	222.23	88.77	130.74	770.00

	Control	Treated
All transects	1285	200
Matched transects	397	159
Unmatched transects	888	41

Table S3. Covariate balance pre and post matching for Fish Habitat Reserves (FHR) and Special Management Areas (SMA) respectively. Values over 25 suggest imbalanced covariates. Chi-squared tests using the Xbalance package were performed to determine whether at least one covariate was unbalanced pre and post matching.

	Fish Habitat Reserve		Special Management Area	
Variable	Prematching	Post matching	Prematching	Post matching
Depth	71.95	18.03	22.14	8.74
Distance to land	31.11	16.82	10.09	7.03
Distance to village	40.69	21.55	46.37	4.65
Fishing pressure	17.22	15.00	41.23	15.70
Habitat - Exposed	7.17	9.03	1.95	2.02
Habitat - Fringing	43.30	19.74	37.41	1.23
Habitat - Semi exposed	44.07	10.71	31.74	2.83
Island - Ha'apai	54.88	7.53	98.46	8.74
Island - Tongatapu	35.62	2.25	5.06	14.56
Island - Vava'u	91.03	6.63	97.81	20.22
Live coral cover	13.58	16.02	22.09	22.35
Rugosity	25.74	9.48	29.73	1.70
Slope	12.75	14.25	22.58	9.62
Surveyor 1	72.85	4.80	28.74	12.87
Surveyor 2	27.77	4.80	7.21	7.84
Surveyor 3	42.82	NA	42.82	NA
Surveyor 4	49.83	NA	6.82	9.43
Wave energy	5.55	9.48	2.41	23.28

		chi-squared	df	p value
FHR	Prematching	212	15	< 0.05
	Post matching	17.1	13	0.19
SMA	Prematching	287	15	< 0.05
	Post matching	38.8	14	0.067

Table S4. Progress of Tonga's SMA program and relevance of this study towards a) Tonga Fisheries Sector Plan objectives, b) National Biodiversity Strategy and Action Plan, and c) Aichi targets for marine biodiversity conservation and sustainability.

A) Tonga's Fisheries Sector Plan 2015

Overall objective: *"To increase the sustainable shared benefits for Tonga from optimal use of its living marine resources."*

Relevant Section	Subcomponent	Activity/Actions/Target	Relevance of the current study within the SMA program
4.1 Sustainable community fisheries	4.1.1 Enhancement of the SMA program 4.1.2 Expansion of the SMA network 4.1.3 Management and development of coastal fisheries	1. Periodic surveys of coastal fisheries 2. Monitoring and review of SMA performance 3. Target to exceed 100 SMAs throughout Tonga	First study to demonstrate ecological impacts of SMA program. Improved biomass, density and size of key target reef fish families and improvements to species richness with no-take FHRs, but not SMAs.
6.3 Monitoring and Evaluation	158. Key performance indicators	Lists key performance indicators suggested to be useful for monitoring and evaluation. Caveat is that many metrics focus on targets that are not linked to impact, such as: • Number threatened fish species • % MPAs in territorial waters • % marine area covered by management	The 19 quantified metrics of impact included in this study may be more useful for determining the long term efficacy of the program than area based targets.
8.1 Result framework	C1. Sustainable community fisheries	1. Availability of food fish maintained 2. Independent review of SMA performance 3. Suggestions for best practices in SMAs 4. SMA monitoring report 5. Increased productivity of major fisheries (decline in CPUE reversed)	The present study provides support for points 1-4. However this study was unable to demonstrate point 5, which requires further studies using catch data.
9.2 Thirteen goals for the Fisheries Division Strategic Action Plan		Goal 4: Improve fisheries information and catch data on resource status through strengthening of existing data management framework and reporting process. Goal 6: Increase production by 15 percent through opening of closed fisheries, research new fisheries, and scientific monitoring of existing fisheries in order to better target fishing effort.	Fisheries information has been improved through ecological surveys. Scientific monitoring has demonstrated the positive impacts of FHRs, but not of SMAs.

		Goal 8: Continue to support Community Based Management capacity building, enforcement capability, and expansion of Special Management Areas for sustainable food supply.	Results suggest that food supply within FHRs is improving, although these areas are closed to fishing. Further management may be required within SMA areas.

B) National Strategic Biodiversity Action Plan (2006) and Kingdom of Tonga's fifth national report to the Convention on Biological Diversity (Chapter 2) (2014)

Relevant Section	Subcomponent	Activity/Actions/Target	Relevance of the current study within the SMA program
Theme Area 2: Marine Ecosystems	Objective 2.2. Marine Conservation Areas	To expand the existing network of protected areas to effectively conserve major coastal and marine habitats of biological and socio-economic value.	The SMA program is expanding rapidly. However the current study raises two points: 1. Only the FHRs appear to be effectively conserving resources, and the reef area within FHRs remains small (FHRs 3% total reef area, SMAs 20% total reef area). 2. Area based targets can generate further problems for conservation.
	Objectives 2.3. Sustainable management of marine biodiversity	To promote the use of environmentally sound practices in the management of marine resources.	This study demonstrates that the FHRs support this objective, although it remains unclear for SMA areas.
	Objective 2.4. Promote scientific research and monitoring	To promote scientific research and regular monitoring of critical marine ecosystems, and the proper management of scientific data to support the conservation and sustainable management of marine ecosystems.	These results provide the first large-scale analysis of the efficacy of the SMA program that demonstrate clear positive impacts of management.
Theme Area 3: Species Conservation	Objective 3.1. Protection of priority species	To ensure the protection of viable populations of all priority conservation species of Tonga.	Reef fish species are being protected with FHRs and there is clear evidence that their numbers are increasing, along with the overall species

			richness within the no-take areas.
	Objective 3.2, Sustainable use and management of species	To ensure the sustainable use and management of species of economic and cultural significance.	Populations of target species are increasing in overall biomass, density and average size within no-take FHRs.
	Objective 3.4. Research and Monitoring	To encourage basic scientific research monitoring surveys to identify, document and monitor progress in the conservation of priority species and to support on-going planning and conservation efforts.	These surveys represent the first national surveys completed across the three main island groups of Tonga.

*In addition these results aid with barrier 3 in the CBD report 2014 – That "The monitoring programme for protected areas and biodiversity conservation activities in Tonga are very weak due to lack of financial support from government".

C) Aichi 2020 biodiversity targets and Kingdom of Tonga's fifth national report to the Convention on Biological Diversity (Chapter 3) (2014)

Target	Listed relevant national targets	Relevance of the current study within the SMA program
Target 6: *"By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits."*	2.3 Marine resources are managed sustainably. 2.4 Knowledge of health of critical marine ecosystems is current. 2.5 A general public that is well informed of marine conservation issues and supportive of marine conservation objectives.	No-take FHRs have successfully minimized overharvest within their boundaries and improvements to both target species and species richness are being observed. However this area remains small and further management is necessary beyond FHR boundaries, including within SMAs.
Target 10: *"By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning."*	7.1 Concepts of conservation and sustainable use of biodiversity are integrated into sectoral policies, programs and plans. 7.2 Agencies and organizations of varied interests and areas of specialization collaborate on conservation work 7.3 Environmental impact assessments are an acceptable planning requirement for all development activities.	Within FHR areas, pressures have been mitigated to the extent that recovery has been observed. Pressures may be reduced within SMA areas, although further research is necessary to assess whether management within SMA areas is helping maintain ecosystem function.
Target 11: *"By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes."*	2.2 A 50% increase in the total area of marine ecosystem under conservation management in 10 years. 4.1 Priority species are well protected and their population increasing. 7.1 Concepts of conservation and sustainable use of biodiversity are integrated into sectoral policies, programs and plans. 7.4 Biodiversity valuation results are acceptable and incorporated into cost benefit analyses of development proposals	FHRs are expanding throughout the country, although as of October 2019 they still only cover 3% of total reef habitat in Tonga. While SMAs cover 20%, there are no observable positive impacts within these areas and therefore they should not be considered as effectively conserved.

Appendices

Appendix 1. Smallhorn-West P, Govan H (2018) Towards reducing misrepresentation of national achievements in marine protected area targets. *Marine policy* 97, 127-129

Appendix 2. Smallhorn-West P, Garvin J, Slayback D, DeCarlo T, Gordon S, Fitzgerald S, Halafihi T, Jones G, Bridge T (2019) Coral reef annihilation, persistence and recovery at Earth's youngest volcanic island. *Coral reefs*

Appendix 3. Smallhorn-West P (2020) Progress towards conserving Tonga's coral reefs. *SPC Fisheries newsletter #160*

Appendix 4. Smallhorn-West P., Sheehan J., Rodrigues-Troncoso A., Malimali S., Halafihi T., Mailau S., Le'ota A., Ceccarelli D., Stone K., Hokafonu T., Matoto L., Pongi K., Pressey B., Jones G. (2020) *Kingdom of Tonga Special Management Area Report 2020*

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Towards reducing misrepresentation of national achievements in marine protected area targets

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ABSTRACT

Parties to the Convention on Biological Diversity (CBD) adopted 20 targets, known as the Aichi Targets, to benchmark progress towards protecting biodiversity. These targets include Target 11 relating to Marine Protected Area coverage and the World Database on Protected Areas (WDPA) is the accepted international database for tracking national commitments to this target. However, measuring national progress towards conservation targets relies on sound data. This paper highlights the large-scale misrepresentation, by up to two orders of magnitude, of national marine protected area coverage from two Pacific Island nations in multiple online databases and subsequent reports, including conclusions regarding achievements of Aichi 11 commitments. It recommends that for the target driven approach to have value, users of the WDPA data should carefully consider its caveats before using their raw data and that countries should strive for a greater degree of accountability. Lastly it also concludes that protected area coverage may not be the best approach to environmental sustainability and that the remaining 19 targets should be considered to a greater extent.

1. Introduction

With global declines in marine biodiversity there is a strong need for reliable international commitments for the sustainable management of the oceans. The highest level of international commitment is embodied by the Strategic Plan for Biodiversity 2011–2020 and its 20 Aichi Targets, which were adopted by the parties to the Convention on Biological Diversity (CBD) [1]. The expansion of protected area coverage is Aichi target 11, which by 2020 calls for 10 per cent of coastal and marine areas to be effectively and equitably conserved through protected areas and other effective area-based conservation measures. In this article we define MPAs using the CBD definition of "any defined area within or adjacent to the marine environment, together with its overlying waters and associated flora, fauna and historical and cultural features, which has been reserved by legislation or other effective means, including custom, with the effect that its marine and/or coastal biodiversity enjoys a higher level of protection that is surroundings" [2].

The Aichi 11 target of 10% protection for marine and coastal areas by 2020 has received much global attention. While the relation between coverage and conservation impact is disputed [3], it should be a relatively easy to measure indicator by which to assess the commitment of countries to marine conservation. National contributions to this target are often measured using the World Database on Protected Areas (WDPA) (www.protectedplanet.net) as well as other similar online datasets (e.g. MPA global - www.mpaglobal.org, PIPAP – www.pipap.sprep.org and Reefbase - www.reefbase.org). Combined, these databases form the backbone of reporting on progress towards meeting Aichi 11 and other global marine conservation targets. The criteria used in the WDPA for calculating protected area coverage are based on the IUCN and CBD definitions of protected areas. Changes to protected area coverage can be submitted to the WDPA only by government agencies; updates by external bodies (e.g. NGOs) are recommended to be submitted jointly with the management authority of each country [4].

National progress towards global conservation targets can only be assessed when the data employed are sound. Given the widespread use of the online datasets in measuring MPA coverage [5–8] it is important to ensure that they accurately measure the progress of nations towards stated targets. With large datasets some error is inevitable and misrepresentations of MPA coverage can arise from the existence of paper parks [9], protected area downgrading, downsizing and degazettment [10] or mistakes in the database [11]. However, while minor errors in online datasets could be judged as inevitable, gross and persistent misrepresentations of national achievements can undermine the overall

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process of marine conservation and should be addressed as a priority.

Misrepresentations in protected area coverage can result from both over- and under-representation of actual targets. While Visconti et al. [11] discussed the ramifications of under-representation due to limitations in the WDPA for identifying small-scale and community-based protected areas, large-scale overestimations of protected area coverage have received less attention. The consequences of overestimates in protected area coverage can be particularly negative as they may also lead to the subsequent cessation of conservation action if targets appear to have been met. Flow on effects, whereby management authorities aim to replicate reported success stories and researchers aim to understand factors leading to success are also more likely to arise from over, rather than under-representation of protected area coverage.

Attentive observers of the impressive and blossoming MPA coverage in the Pacific (e.g. Marae Moana – Cook Islands; Le Parc Naturel de la Mer de Corail – New Caledonia) may have noticed strange oscillations in the total figure and achievement of national targets [12]. Investigating misrepresentation of protected area coverage in this region is particularly pertinent given recent high-profile commitments and increasing claims of moral high ground in ocean stewardship. This paper uses two case studies from Tonga and Kiribati to draw attention to the large-scale misrepresentation, by up to two orders of magnitude, of national MPA coverage in this region, arising from both the long-term persistence of a massive paper park and data/quality control errors in the WDPA reporting process. Both large-scale misrepresentations occur in multiple online databases and subsequent reports, including conclusions regarding achievements of Aichi 11 commitments. It concludes that countries should actively strive for a greater degree of external accountability and that users of the WDPA should carefully consider its caveats before using their raw data for the target driven approach to have value.

2. Tonga

Tonga's marine protected area coverage on the WDPA (as of 31/01/2018) is reported as 10,133.87 km^2, from 17 MPAs. Of these by far the largest is the Ha'apai Conservation Area (HCA), endorsed in 1995 by the Ha'apai Development Committee and Cabinet as a 10,000 km^2 multi-use conservation area in central island group of Tonga, at a cost of $179,000 USD [13]. However, there is no evidence that the HCA, which is listed in the WDPA, PIPAP, MPA global and Reefbase, has enjoyed a higher level of protection than its surroundings in the last fifteen years. This conclusion is directly corroborated by several sources. Two reports on the current status of MPAs in the South Pacific [14,15] both deemed the HCA as inactive and recommended its removal from the national list of MPAs and therefore the WDPA. As early as 2002 the terminal evaluation of the South Pacific Biodiversity Conservation Program (SPBCP) also suggested that the HCA was unworkable due to a disconnect between the community, who saw the HCA as a government project, and the coordinating committee of government employees who did not feel ownership for the area. Two consultants, a government official and community members were also asked about the current status of the HCA and stated either that it is not currently active or that it was not achieving any more management than adjacent areas. One consultant expanded on this to confirm that there is no information on the legal status or management of the HCA, and no management plan has been found during several local reviews by Ministry officials.

The WDPA additionally lists 16 other MPAs in Tonga, nine of which were established in the 1970's and collectively cover 34.65 km^2 and six of which were established as part of the Special Area Management (SMA) program starting in 2006 and cover 99.22 km^2. The SMA program is currently under a rapid expansion and in 2017 implemented or was in the process of implementing an additional 18 areas and a further 25 on a waiting list. The best estimate of the current of the coverage of SMAs in Tonga is ~250 km^2. While local communities manage the SMA reserves, the nine pre-SMA areas were centrally managed. However, discussion with consultants and community members support the conclusion that lack of local acceptance and limited resources for enforcement resulted in management for the non-SMA areas becoming non-existent shortly after their inception.

In conclusion it is apparent that both by area and by number Tonga's MPA coverage has been over-reported. The reported figures are 10,133.87 km^2 from 17 marine parks, whereas the realistic current figure of those listed on the WDPA is 99.22 km^2 from seven active marine parks. This represents a two order of magnitude discrepancy between the WDPA database and the actual figures. In Govan [14] the coverage of the HCA was sufficient to account for approximately 40% of total MPA coverage in the South Pacific (as of 2009).

3. Kiribati

The Phoenix Island Protected Area (PIPA) in Kiribati was one of the largest marine protected areas in the world when established in 2006, and according to the WDPA (31/01/2018) its representative marine area is 397,447 km^2. In 2010 PIPA was also inscribed as a World Heritage area and on inscription was added to the WDPA database, inadvertently leading to a double entry of its size. Govan [15] warned that at least as late as 2013 the WDPA dataset listed the total coverage of PIPA as 818,750 km^2 owing to its double counting. This misrepresentation has now been partly corrected: the online WDPA portal lists the corrected figure, however the MPA is still listed twice in the raw database, with the representative marine area column being correct but the doubling still present in the total area protected column. Therefore, if care is not taken to incorporate the correct column in the dataset, potential still exists for users of the raw database figures to double count one of the largest marine protected areas on earth.

4. Reporting on progress towards conservation targets

The incorrect WDPA data for both Tonga and Kiribati have propagated through regional and international reports quantifying the achievement of MPA targets. For example, both the regional report "State of Conservation in Oceania" [12] and a presentation by representatives of the Convention on Biological Diversity (CBD)[1] report Tonga's total MPA coverage as 9.4% (8457 km^2) of their territorial waters, or 1.51% of their total marine area and highlight it as a success case for achieving the Aichi CBD Target 11. In reality Tonga's current total MPA coverage is closer to 0.1% of its territorial waters and 0.014% of its total marine area. The same CBD presentation reports Kiribati as having reached 22% coverage of their total marine area. As the actual figure is 11% this overestimate is likely due to the issues highlighted with the WDPA dataset in Section 3. In addition, there are various international reports citing the WDPA data for global targets of marine conservation and there is a high risk that these reports could be citing incorrect figures for the South Pacific [5–8].

5. Conclusions

This article documents how large misrepresentations of the MPA coverage of two Pacific island countries have occurred in the WDPA and thus their contributions to achieving the Aichi 11 CBD target. This has led to overestimates of Tonga's contribution to marine protected area coverage targets by two orders of magnitude and Kiribati's by double. Large-scale misrepresentations significantly interfere with measuring progress towards marine conservation and while errors within the WDPA framework were highlighted five years ago in Visconti et al. [11], problems remain. While Visconti et al. [11] pointed to possible underestimates in protected area coverage due to the omission of community managed areas, this article points to gross overestimates.

[1] *Gidda S. (2010) Target 11 & CBD PoWPA.*

These are of greater concern, given both that conservation efforts may cease if targets appear to be met and that governments and management may aim to replicate reported successes.

The misrepresentations discussed in this article from the South Pacific are likely symptomatic of larger-scale issues regarding both balancing responsibility between government and the WDPA as well as the priority countries accord, or are able to accord, in achieving accuracy and accountability in this matter. It is clear that both the HCA and PIPA misrepresentations have been known for some time [13–15], but it is also evident that the responsibility for rectification has likely fallen between the perceived roles of government and the WDPA. Misrepresentations may also be driven by failures in procedure, including lack of clarity for how to make changes, lack of specified roles and lack of funding or technical capacity. Given the potential for embarrassment on the international stage, malignant issues such as either intentional misrepresentation or low initiative to highlight known problems could also exist. There is therefore a clear need to develop both greater clarity and accountability in the process of updating protected area coverage.

While systems for the updating and removal of protected areas from the WDPA do exist, any system in place from which the HCA and PIPA entries areas are not subject to removal clearly needs refining. The first priority should be to clarify the roles and responsibilities of both governments and the WDPA in the reporting and updating process. Governments should also be subject to a greater degree of external accountability from the WDPA or other external bodies. Quality control and data checking within the WDPA could also focus on large protected areas first, as these carry the greatest relative analytical weight. For regional analyses it should also not be prohibitively time consuming for the data user to check for double entries.

The WDPA data forms the backbone of reporting on global marine conservation targets and therefore it is important for countries, no matter how small, to ensure accurate and regular assessment of the status of national marine protected areas and at least annual reporting to the WDPA. This high degree of accountability will be vital for a region that is increasingly claiming moral high ground in ocean stewardship (e.g. Pohnpei Ocean Statement: A Course To Sustainability. Declaration of the Pacific Island Forum Leaders, Pohnpei, FMS). Countries and other data users are encouraged to ensure correct figures are regularly updated in the national and WDPA databases and to carefully read the WDPA guidelines [4] before using their raw data.

Finally, this assessment gives rise to some concerns. Aichi target 11 would at first glance be one of the more easily measurable of the 20 targets. However, given the complications revealed in monitoring even the geographical extent of MPAs, serious concerns may arise relating to whether it will be possible to record their effectiveness and equitability. Potential solutions to assessing effectiveness are being developed, but given the concerns that protected area coverage may not be the best indicator or approach to achieving environmental sustainability [3] more serious consideration is needed of the remaining 19 targets.

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Conflict of interest

None.

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NOTE

Coral reef annihilation, persistence and recovery at Earth's youngest volcanic island

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Abstract The structure and function of coral reef ecosystems is increasingly compromised by multiple stressors, even in the most remote locations. Severe, acute disturbances such as volcanic eruptions represent extreme events that can annihilate entire reef ecosystems, but also provide unique opportunities to examine ecosystem resilience and recovery. Here, we examine the destruction, persistence and initial recovery of reefs associated with the hydromagmatic eruption that created Earth's newest landmass, the Hunga Tonga–Hunga Ha'apai volcanic island. Despite extreme conditions associated with the eruption, impacts on nearby reefs were spatially variable. Importantly, even heavily affected reefs showed signs of rapid recovery driven by high recruitment, likely from local refuges. The remote location and corresponding lack of additional stressors likely contribute to the resilience of Hunga's reefs, suggesting that in the absence of chronic anthropogenic stressors, coral reefs can be resilient to one of the largest physical disturbances on Earth.

Keywords Refuge population · Ecosystem resilience · Coral reef · Volcanic eruption · Reef recovery

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Introduction

Massive acute disturbances such as explosive (hydromagmatic) volcanic eruptions represent extreme events that can annihilate entire ecosystems, but also provide unique opportunities to examine ecosystem resilience and recovery (Fridriksson and Magnússon 1992). For example, the eruptions of Ksudach (1907), Surtsey (1963) and Mt St Helens (1980) were critical to developing the theory of plant succession (Fridriksson and Magnússon 1992; del Moral and Wood 1993) and for demonstrating the importance of refuge populations for ecosystem recovery (Grishin et al. 1996). However, equivalent studies of marine ecosystems encompassing both the impact and early stages of recovery are lacking. Consequently, the extent to which key factors promoting recovery in terrestrial systems (e.g., refuge populations) also apply to the marine realm remains unknown.

Many coral reefs occur in regions of high volcanic activity; indeed, submarine volcanoes are the foundation for coral atolls (Terry and Goff 2012). Volcanic eruptions can cause severe short-term negative impacts on nearby reefs (Vroom and Zgliczynski 2011), but over longer timescales can provide new substratum for coral settlement (Tomascik et al. 1996; Pinault et al. 2013, 2014) and facilitate larval dispersal (Bryan et al. 2012). Similar to terrestrial ecosystems, local recruitment from refuge populations is thought to have initiated the rapid recovery of corals following the eruption of Krakatau in 1883 (Starger et al. 2010). However, due to the low frequency and unpredictability of substantial volcanic eruptions in close proximity to coral reefs, empirical evidence documenting their immediate and direct impacts is lacking. Likewise, the role of refuge populations in their potential recovery and patterns of ecological succession are poorly understood. Here, we utilize multiple lines of evidence to reconstruct the eruption of the Hunga Tonga-Hunga Ha'apai (HTHH) volcano, which in 2015 spawned Earth's newest landmass, and document the short- to medium-term effects of the eruption on adjacent reef communities.

From December 2014 to January 2015 a *Surtseyan* (Garvin et al. 2018) eruption in the Tonga-Kermadec volcanic arc constructed a subaerial tuff cone edifice, creating a new 185 hectare landmass between two preexisting islands, provisionally named Hunga Tonga–Hunga Ha'apai (HTHH) (Fig. 1). The eruption released approximately 0.1 km^3 of new deposits into the atmosphere and created an eruption plume 10–12 km high and visible from > 50 km away. Typical of a Surtseyan eruption, the collapse of water-rich tephra (ash) eruption clouds into ground-

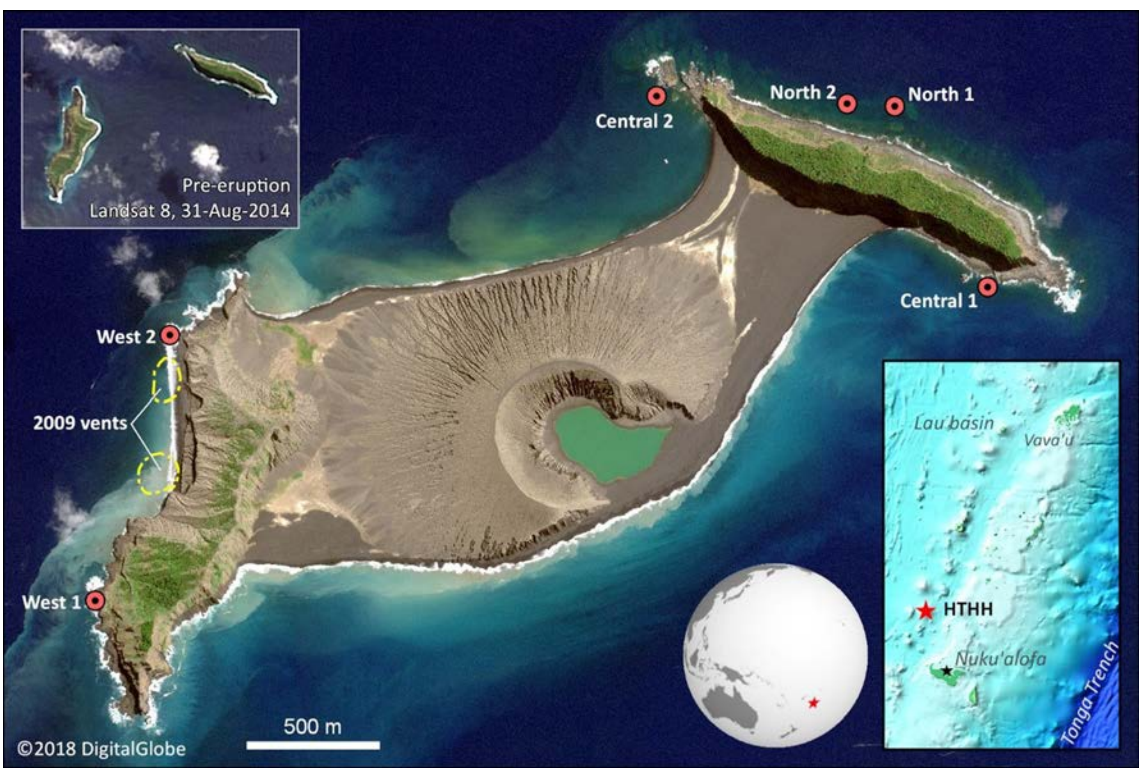

Fig. 1 Development of the Hunga Tonga–Hunga Ha'apai system in Tonga. Image is from DigitalGlobe's Worldview-2 satellite, 19-Aug-2018. Coral survey sites are labeled red points. Yellow dashed lines are the minor vents from the 2009 eruption, which during that time were above sea level (Fig. S2)

hugging pyroclastic debris flow scoured the ground and released massive sediment plumes into the surrounding ocean (Wohletz et al. 1989). High sediment loads were confirmed by xCT and SEM images of brecciated carbonate samples with distinct palagonitization indicative of low temperature mineralization, as well as ballistic and density current sedimentation during the constructive phase of the eruption (Supplementary materials) (Fig. S1). Large sediment plumes drifting to the southwest were also visible in satellite imagery (Fig. S2). Landsat 8 thermal imagery indicated sea surface temperature anomalies of ~ 5 °C near the eruption epicenter during an active eruptive phase on 21 December 2014 (Fig. S2).

Methods

Ecological surveys

Ecological surveys of benthic and reef fish communities were conducted at six sites around the two preexisting islands surrounding HTHH in August and October 2018. The tombolo itself was composed of unconsolidated black sand and was therefore not surveyed. Sites were divided into three sections: (1) the northern flank of Hunga Tonga, which was largely sheltered from the blast, (2) the southern edge of Hunga Tonga, which was near the eruption epicenter, and (3) the western edge of Hunga Ha'apai, where large landslides and a previous eruption in 2009 created new habitat for reef formation.

Four 30 m belt transects were conducted at 4–6 m depth parallel to the depth contour at each site. The size and abundance of all large mobile fish species was recorded in a 5 m wide belt along the transect, while all small site-attached species were recorded along a 2 m belt. Fish sizes were converted to biomass using length–weight relationships on Fishbase (www.fishbase.org). Benthic community composition was quantified using 1 m^2 photoquadrats captured every two meters along each transect: each colony > 5 cm diameter was identified to genus, and it's size recorded using Image-J in order to compare both coral cover and size structure.

Generalized linear models with Tukey's post hoc comparisons were used to examine differences in coral genera richness, coral cover, reef fish species richness and reef fish biomass between western, central and northern groups (Table S1). Differences in the size structure of three common coral genera and two common reef fish species were also compared between western, central and northern groups using generalized linear models (Figure S3; Table S2) in R (R Core team 2017).

Core samples

To determine the immediate effects of the eruption on reef communities and the proximate drivers of coral mortality, core samples were collected from 13 surviving *Porites* colonies (Table S3). Nine cores were collected from the two central sites, and four from the northern sites. No large *Porites* colonies were present at western sites. All cores were processed through a Skyscan 1176 computed tomography (CT) scanner following the methods of DeCarlo et al. (2019) to identify growth anomalies or stress bands coinciding with eruptions. CT scans of the cores were analyzed using Osirix software (V.10.0.5), which enables the digital cutting of 3 mm virtual slices oriented along the axis of maximum growth. The images were then assessed for presence of discontinuities/partial mortality scars and anomalous high-density stress bands, and the timing of any features was determined from annual high/low density bands.

Water quality

Water quality parameters at each site were examined to examine whether spatial variation in environmental conditions among sites could influence community composition. A multi-parameter water quality sonde (YSI 6920-V2-2: Xylem, Hemmant, Australia) was used to collect in situ measurements of water temperature, salinity, turbidity and chlorophyll content. Sondes were deployed at each site at a depth of two meters for 60 min (30 s sampling interval) at a similar time of day and tidal period. Linear models with Tukey's post hoc comparisons were used to determine difference between sites. With the exception of one site exhibiting higher turbidity, sites were not clearly differentiated by any present day water quality indicators examined (Fig. S4; Table S4).

Results and discussion

Effects on coral communities

Despite the magnitude of the eruption and the small size of the island (3.5 km in its longest dimension), the effects on nearby coral reef habitat were spatially variable and ranged from virtual annihilation to no observable impacts depending on the extent to which topography provided protection from the blast (Fig. 2a–c). Reefs on the northern side of HTHH were protected by the preexisting island of Hunga Tonga and prevailing weather conditions, and reefs supported high coral cover and generic richness (Fig. 3a, b; Table S1). The size structure of the most abundant coral genera (*Acropora*, *Pocillopora* and *Porites*) showed that

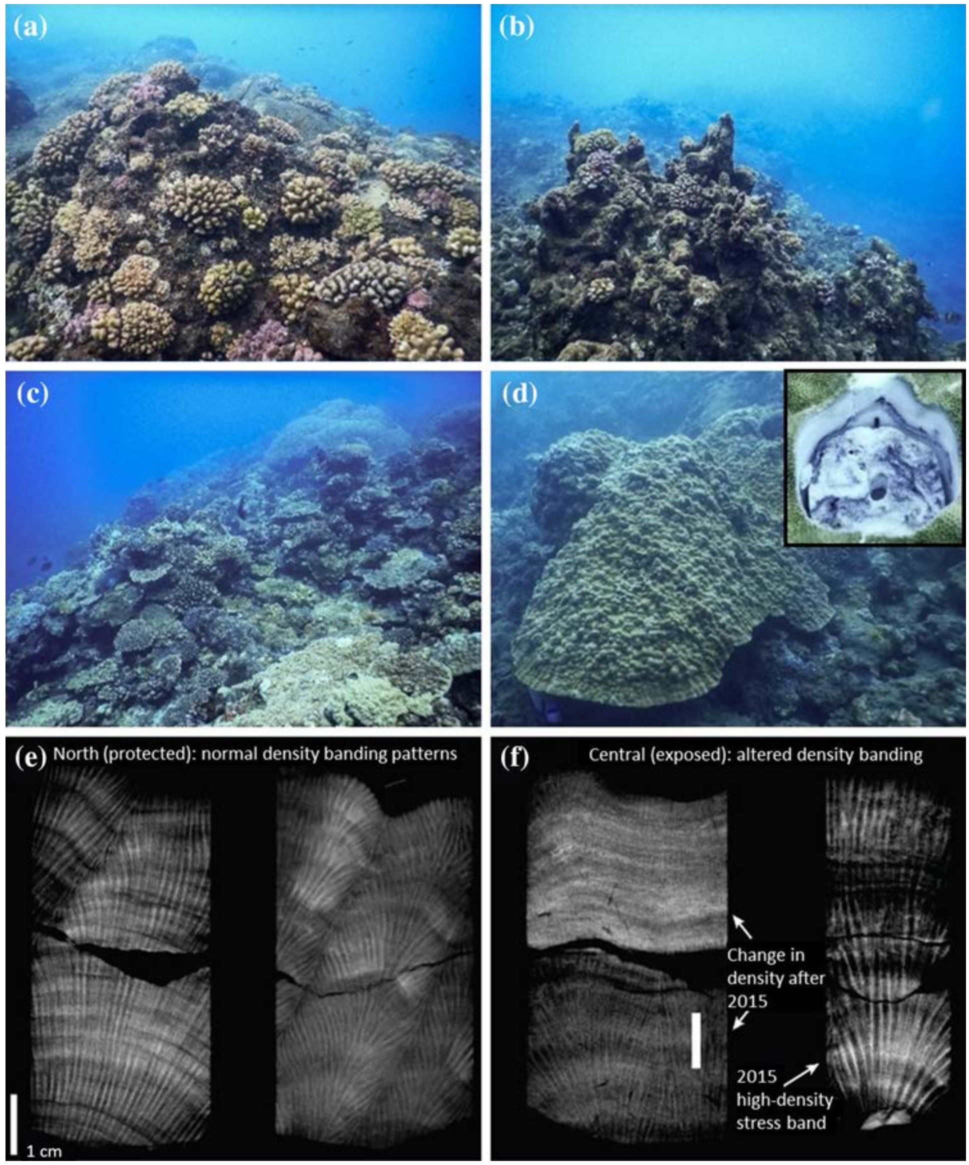

Fig. 2 Reef ecosystem around Hunga Tonga–Hunga Ha'apai. **a** Western flank with high recruitment on new substrate for coral growth; **b** central site adjacent to vent (< 1 km) with complex, yet dead, reef structure and high recruitment; **c** northern flank with high complexity and flourishing reef community; **d** surviving *Porites* colony near vent with distinct black band in core sample; **e** Ct scans of *Porites* colonies from the northern flank displaying uniform banding; **f** cores from central sites with high-density stress bands and modified banding patterns post-eruption

northern sites supported substantially more large colonies than central and western sites (Fig. 4a–c; Table S2). Given established growth rates for all three genera, some colonies at the northern sites clearly predate the eruptions. *Porites* cores from the northern sites also showed no evidence of stress banding or growth anomalies (Fig. 2e). In contrast, reefs along the southwest-facing coast of Hunga Tonga directly exposed to the central eruption vent were effectively annihilated. Coral cover was four times lower than in the northern sites almost 4 yr after the eruption, although in situ dead coral skeletons indicate that dense coral growth existed previously. Despite the extreme environmental conditions during the eruption (high temperatures and smothering by volcanic ash), a few large colonies of *Porites* survived (Fig. 2d), often with substantial partial mortality. Core samples confirmed that extreme conditions occurred at the time of the eruption, evidenced by high-density stress bands, growth anomalies (Fig. 2f), and thick black sediment bands. The *Porites* cores suggest that coral mortality could be attributed to both sedimentation from mantling ash deposits and elevated water temperatures associated with the eruption.

The eruption also created new habitat for reef formation along the western flank of Hunga Ha'apai. The magnitude of the explosive blasts were sufficient to partly denude vegetation on Hunga Ha'apai island (W and SW of the new edifice), resulting in landslides and deposition that created new substrate for coral growth and burying any preexisting communities. Coral cover and generic diversity at western sites were comparable to central sites, and no colonies (alive or dead) old enough to have survived the previous eruption were observed.

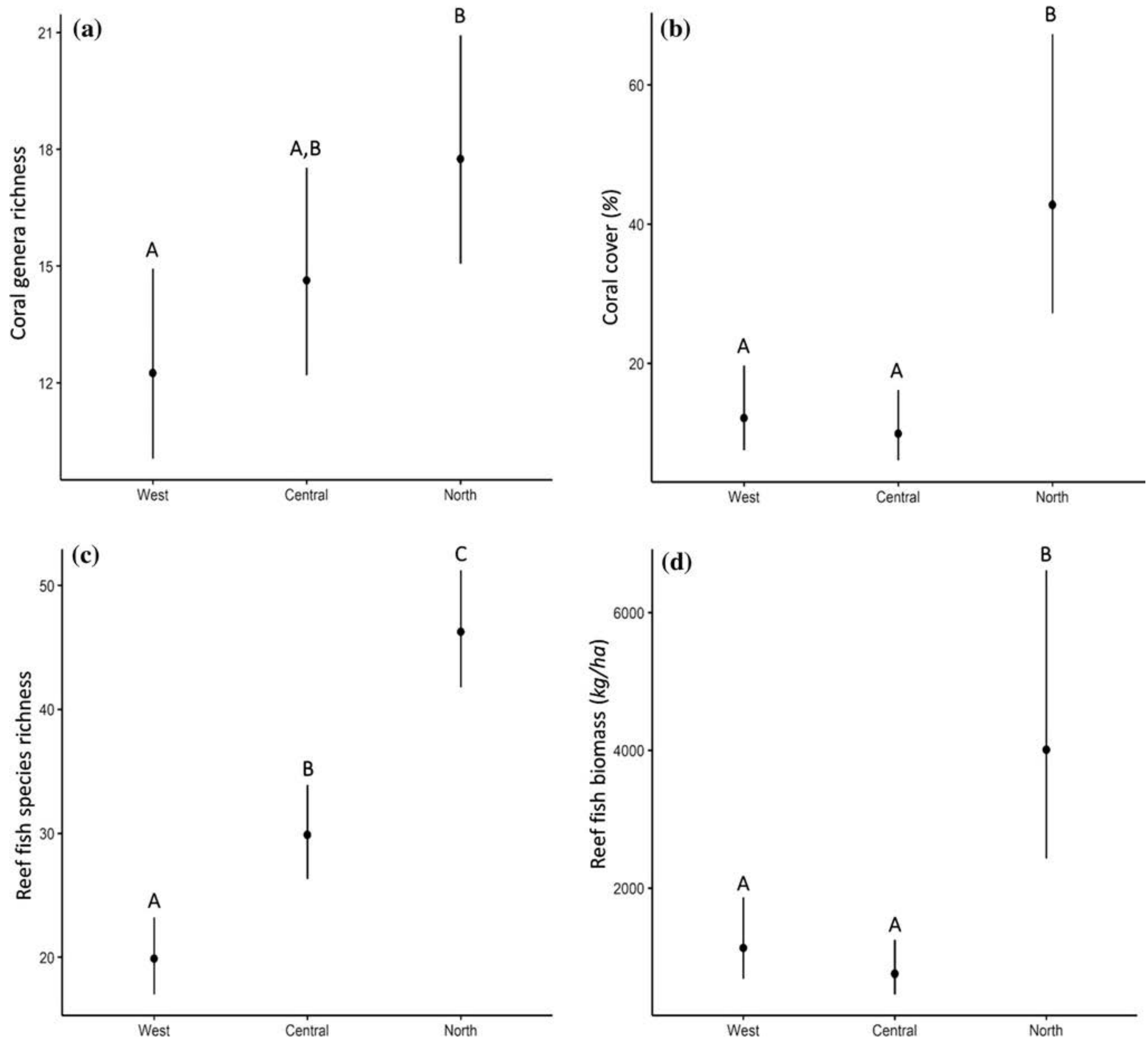

Fig. 3 Benthic and reef fish community structure around Hunga Tonga–Hunga Ha'apai (n = 24). **a** Coral generic richness; **b** coral cover; **c** reef fish species richness; **d** reef fish biomass. Letters indicate significant groupings ($p < 0.5$) based on generalized linear models (mean ± 95% CI) with Tukey's post hoc comparisons

Effects on fish communities

Reef fish community structure showed similar patterns to corals, with the protected northern sites having greater diversity and biomass than either central or western sites (Fig. 3c, d) (Table S1). Mean species richness at the northern sites was 34% and 56% greater than central and western sites, respectively, while mean fish biomass was also > 70% greater at the northern sites. Individuals of two common reef fish species (*Ctenochaetus striatus* and *Halichoeres hortulanus*) were also significantly smaller at the central and western sites than the northern sites. Populations of both *C. striatus* and *H. hortulanus* at the northern sites contained individuals > 4 yr old, while no adult individuals were observed at other sites indicating that these populations had re-establised post-eruption (Fig. 4d, e; Table S2) (Choat and Axe 1996; Hubble 2003). Adult individuals of other large, long-lived species (e.g., *Cheilinus undulatus, Chlorurus bleekeri, Plectorhinchus lineatus*) occurred exclusively at the northern sites.

Recovery and resilience

Almost all sites at HTHH are exhibiting strong signs of recovery 4 yr after the eruption due to high recruitment of juvenile corals. Coral recruitment was dominated by

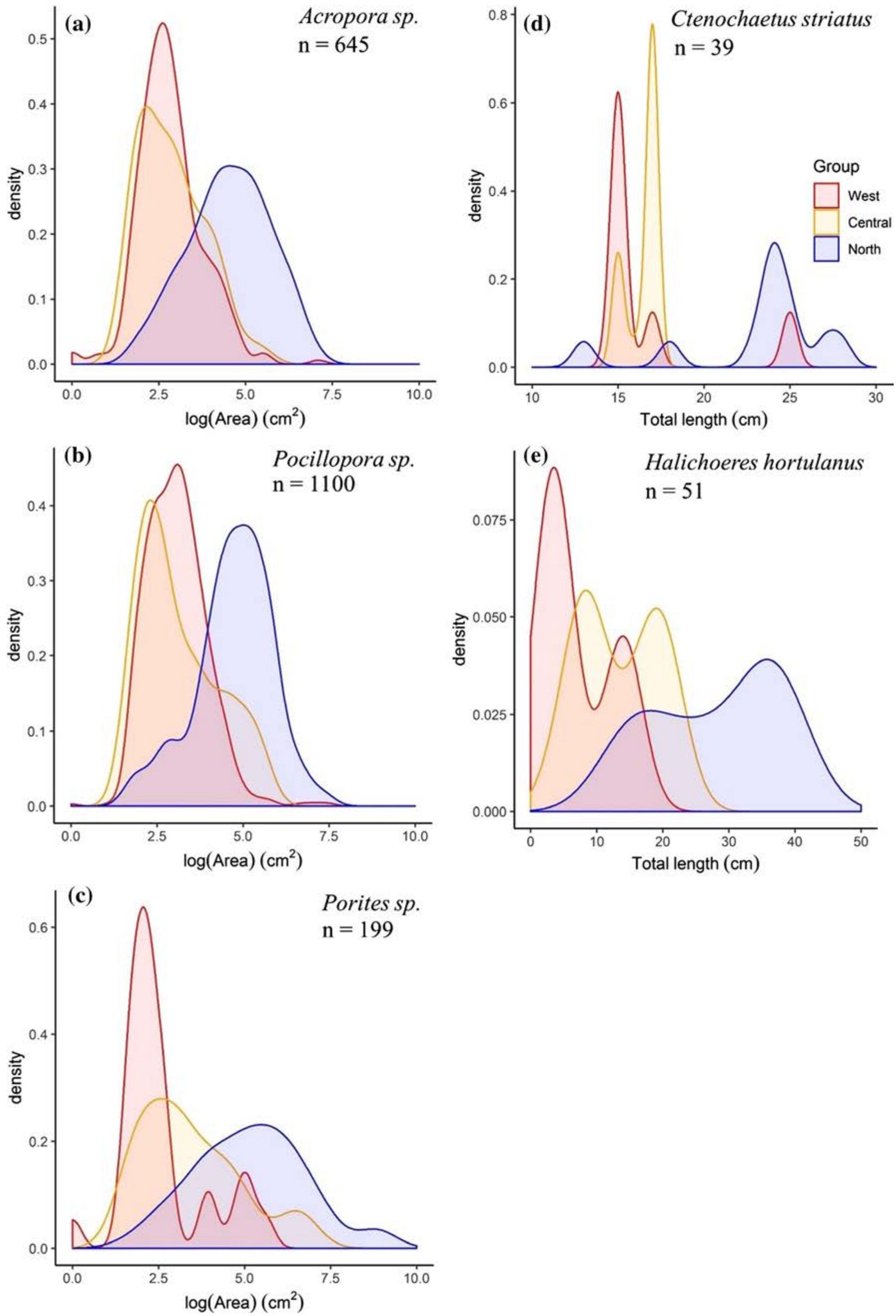

Fig. 4 Size structure of three common coral genera and two common reef fish species around Hunga Tonga–Hunga Ha'apai

numerous species of fast-growing *Acropora* and *Pocillopora* (Fig. 2a), resulting in diverse coral assemblages in the newly-created habitat at the western sites and recovery in the central site after near complete destruction. Generic diversity at the central and western sites was only marginally lower than in the north, although cover remains lower due to the lack of large colonies. In the absence of further disturbances, coral reefs around HTHH are likely to show rapid recovery.

There are several likely causes for the resilience of coral reefs at HTHH. Given its small size, geographic isolation (> 60 km west of the main Tongan archipelago) and the high dependence on self-recruitment in many coral reef systems (Jones et al. 1999; Figueiredo et al. 2013), recovery of reefs around HTHH is likely due primarily to recruitment from local refuges. The importance of local recruitment is supported by the fact that the composition of recruits at the impacted sites reflected the composition of undisturbed communities. Consequently, our results support the hypothesis that small refuge populations can be crucial for recovery from severe acute disturbances in marine as well as terrestrial ecosystems. The remote location of HTHH and corresponding lack of additional stressors is also likely to contribute to reef resilience.

Our results are consistent with previous findings that corals can recover from severe acute disturbances (Richards et al. 2008), in this case due to the refuge populations. However, the capacity of refuge populations to facilitate recovery is likely to decline as the frequency and severity of disturbances continues to increase. Our *Porites* cores suggest that the reefs of HTHH have not experienced severe thermal bleaching, at least since 2010, an increasing rarity among coral reefs globally (Hughes et al. 2018). Few corals outside of the northern sites have reached sexual maturity; therefore, full recovery of HTHH's reefs is dependent upon the continued supply of larvae from adult colonies on the northern flank of HTHH and the survival of juveniles to sexual maturity. The apparent resilience of HTHH's reefs should therefore be viewed through the framework of the increasing prevalence of disturbances in the Anthropocene, where natural disasters that were once the largest catastrophes on Earth may soon rate second to the cumulative impacts of anthropogenic activities.

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Compliance with ethical standards

Conflict of interest The authors declare no conflict of interest.

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