Bournemouth University

Coral conservation programmes in Bali, Indonesia: restoration of degraded reefs and localised socioeconomic benefits

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Award for which degree is submitted: Doctor of Philosophy

Faculty of Science and Technology

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Submission: January 2024

Data availability statement

Some of the data are available as supplementary material for each chapter, and others will become available when additional chapters from this thesis are published. The data that support the findings of this study are available from Zach Boakes upon reasonable request.

Abstract

Coral reefs, the "ocean's rainforests", are valued at over US $1 Trillion globally through the provision of ecosystem services such as including food production, biogeochemical cycling and tourism. Over the past several decades, these ecosystem services have been threatened by anthropogenic threats, which continue to cause a worldwide decline in coral reef biodiversity, functioning and habitat structure. Even with global warming limited to 1.5^{o}C (the IPCC's most optimistic scenario), the projected future for coral reefs is bleak, and this is expected to cause devastating impacts on the human communities who depend on coral reef health for their livelihoods.

In locations where coral reef degradation has already occurred, some local communities have established restoration programmes which utilise tools such as artificial reef deployment, coral propagation and marine protected area establishment, which aim to restore some of the ecosystem benefits associated with healthy natural reefs. Indonesia has experienced widespread coral reef degradation in recent decades and coral reef restoration programmes have been established across the archipelago as a response to this, with the nation now being recognised to have more of these programmes than any other country in the world. A substantial proportion of this reef restoration work is concentrated in the island of Bali. Despite the large number of these programmes on the island, very few monitor the health of their reefs, and even less publish research papers which assess if their work has resulted in tangible ecological and social benefits.

My PhD aimed to evaluate the extent to which coral reef restoration programmes in Bali have restored reef ecology, ecosystem functioning and generated localised social benefits. The primary research location was in Tianyar, north east Bali, on a coral reef restoration programme which had deployed over 20,000 artificial reef units at the end of 2023. Additionally, several other secondary coral restoration sites in Bali were also assessed as part of this research, which allowed comparisons between each of them. To assess biological community structure, this research employed various ecological surveying techniques (such as remote under water video to quantify fish communities and photoquadrats to quantify benthic communities). To assess ecosystem functioning, water samples were taken to test for inorganic nutrients, and sediment (surface and sediment trap) samples were taken to test for particulate organic carbon. To assess the generation of social benefits from coral reef conservation programmes, semi structured interviews and focus group discussions were held with stakeholders in three different coral reef conservation regions in Bali.

My results showed that fish communities on artificial reefs became similar (ecologically equivalent) to those on a nearby CR over a 3 year period, whilst benthic populations remained quite different. I

found that artificial reefs showed some resemblance to CRs in terms of functioning, but were not functioning fully as natural reefs. My findings also showed coral reef restoration can lead to localised social benefits through additional marine tourism, and can also promote the generation of pro-environmental behaviours within the community, through education and empowerment of local leaders which create 'ocean empathy' amongst their people. Overall, the findings of my PhD found that coral reef conservation programmes in Bali have been effective in capturing some of the benefits of natural CRs, although they were not yet shown to act as a direct replacement for the natural reefs which they may aim to replace. More work is needed to assess the benefits of coral reef conservation programmes over longer time scales.

Acknowledgments

I was first introduced to the idea of researching coral reef conservation as an undergraduate student at Bournemouth University. My supervisor (both at the time, and now as a PhD candidate), Professor Rick Stafford, had been hugely supportive of my previous work as the co-founder of 'North Bali Reef Conservation' and suggested I base my undergraduate dissertation around it. I keenly followed Rick's advice, and shortly after this was complete, he suggested I consider continuing this work as a PhD project. The rest was history. I owe a huge amount of debt to Rick, as well as my second supervisor, Dr Alice Hall, for their patience, knowledge and assistance throughout the duration of my PhD. Rick and Alice have provided especially important contributions in terms of conceiving ideas for the research, regularly reviewing my work and supporting with statistical analysis.

I also wish to acknowledge the generous support provided by my Indonesian research team, notably Professor Luh Putu Mahyuni and Dr I Gusti Ngurah Agung Suryaputra (as well as all other Indonesian researchers listed as co-authors). From their hard work assisting with obtaining research permits, to the long days collecting data in the field, the assistance of my local research team has been invaluable and it has been a true pleasure working with them. My deepest gratitude also extends out Ketut De Sujana Mahartana, as well as the rest of the Yowana Bhakti Segara and North Bali Reef Conservation team. The coral reef conservation work which this community has undertaken since 2017 is truly inspiring and I am humbled to have been permitted to conduct most of my PhD research in their village.

My gratitude also extends out to Earthwatch for their direct funding of this research, as well as the in-the-field assistance which they (and their dedicated volunteers) have generously provided. I especially wish to thank Sarah Wishart and Dr Stan Rullman, both from Earthwatch, for their kind support throughout. Furthermore, I am grateful for the Bournemouth University based researchers (notably Dr Marin Cvitanovic, Dr Daniel Franklin and Dr Kathy Hodder) who shared important knowledge and provided key support, as well as all other BU staff who offered help throughout various stages of my PhD.

I am grateful my partner, Janis Khansa Putri Argeswara for her endless encouragement, problem solving and patient listening, and can now confirm I understand the statement “your partner completes your PhD with you”, which I was told when I first started my doctoral studies. Finally, I owe a great debt of gratitude to family who have also been on this journey with me (even though I have spent most of it on the other side of the world to them!), especially to my parents who always encouraged me to be myself and trust in the things I love.

Author's Declaration

I hereby declare that this work is original in its entirety. All work was completed by myself, and/or supported by my supervisors or research partners which are listed as co-authors on my papers. The work of others included here has been referenced appropriately.

I grant the Bournemouth University repository permission with respect to online access of this work.

A research permit was obtained from Indonesia’s Ministry of Research (BRIN). Research permit number: 34/TU.B5.4/SIP/VII/2023.

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Chapter 1: Introduction

Coral reefs, large underwater habitats of calcium carbonate skeletons produced over time by coral polyps, are critically important to tropical coastlines (Hoegh-Guldberg et al. 2017). Often referred to as ‘rainforests of the sea’, coral reefs occupy less than 0.1% of the ocean floor, yet host 25% of the world’s marine species (Fisher et al. 2015). They provide ecosystem services estimated at a value of over US $1 Trillion globally (Costanza et al. 2014) through food provision, shoreline protection, biogeochemical cycling and tourism (Moberg and Folke 1999, Principe et al. 2012). Coral reefs and associated ecological communities face a bleak future from climate change (now established as the cause of mass bleaching (Hughes et al. 2017, Sully et al. 2019), as well as poor fishing practices which may damage the reefs and remove important fish and invertebrate species. Local communities often rely on the ecosystem services (i.e. food and tourism) that coral reefs provide (Principe et al. 2012, Pratchett et al. 2014), and one of the most important steps to protecting these services should be an immediate and large-scale reduction in greenhouse gas emissions (Ben-Romdhane et al. 2020). Despite not addressing the global issue of coral decline, small-scale conservation and restoration [*], multiple tools may be utilised to capture some of the ecosystem benefits of healthy coral reefs. These tools have been shown to be useful, despite the threat of climate change, as they can support communities who depend on their local reefs for their livelihoods (Hein et al. 2021, Hughes et al. 2023). In situations where coral reefs become degraded, artificial structures may be able to provide similar ecosystem services at a local-level (Komyakova et al. 2019, Boakes et al. 2022).

Indonesia makes up 16 % of the world’s total coral reef area (Burke et al. 2011) and has more coral reefs than any other country on Earth (Lamont et al. 2022). It sits within the Coral Triangle, an area recognised as the global centre of marine biodiversity (Allen and Erdman 2008) which is of global conservation importance (Briggs 2005). Despite this, it has been shown that over 95% of the coral reefs of Indonesian are threatened (Burke et al. 2011), primarily as a result of overfishing, pollution, mass bleaching and destructive fishing practices (Hadi et al. 2020, Boakes et al. 2022, Razak et al. 2022). Bali is a province of Indonesia, and has the second highest documented reef fish species

[*] The term ‘ecological restoration' is defined as "the action of aiding the recovery of a degraded, damaged, or destroyed ecosystem" (SER 2004), whereas ‘ecological conservation' encompasses a wider process that involves both preservation and protection (Parsons et al. 2017). We used both terms throughout, with ‘restoration’ being used in the context of ‘pro-active’ reef recovery techniques (e.g. artificial reefs deployment) and ‘conservation’ referring to both pro-active and ‘reactive’ reef recovery (e.g. marine protected area establishment).

richness in the Asia-Pacific (Mustika and Ratha 2013), however most recent regional data shows that 20% of Bali's coral reefs declining in health and 30% are already in poor condition (Marine and Fisheries Office 2017 data, as cited by Wicaksana (2020)).

The marine environment in Bali provides a unique research opportunity, because despite the documented decline in coral reef health, the island's communities, Non-Government Organisations (NGOs) and governments have greatly invested in artificial reefs as a reef restoration tool. Coral reef conservation projects are now widespread across Bali, from the coastal seas of Bali's largest tourism hubs, such as Nusa Dua, to some of its poorest regions like Buleleng and Karangasem (Puspasari et al. 2020). A minimal amount of research has been undertaken (and few published studies) in Bali which assesses the extent to which artificial reefs can mimic the natural communities, ecosystem processes and ecosystem services that occur on coral reefs. Additionally, I am the co-founder of the NGO 'North Bali Reef Conservation', which has built one of Indonesia's largest artificial reefs (18,000 units). This gives me the ability and flexibility to run research projects on the artificial reefs my NGO has deployed. I also work closely with multiple other coral reef conservation organisations which deploy artificial reefs across the island, allowing me to study these projects on a regional level too (especially useful for the social studies).

All countries in south east Asia are considered to have a critical lack of coral reef monitoring (Tun et al. 2005). Indonesia is no exception, and it has been highlighted that only around 16% of coral conservation projects in Indonesia involve a post-installation monitoring framework (Razak et al. 2022). This highlights the increasing importance of research projects that assess the effectiveness of coral conservation programmes in restoring previously degraded reef communities and generating localised social benefits. Furthermore, several of the chapters in this PhD thesis will assess the ecological and social benefits of artificial reefs, which are the primary method used to restore reefs in Bali (*personal communications with Dr Rahmadi Prasetyo, local coral reef ecologist).* Whilst there is a great deal of international research on the ecological benefits of artificial reefs, there are very few conducted within Indonesia, especially medium to long term studies. It is important to assess the effectiveness of artificial reefs in Indonesia, given their popular use within the country, especially as their results may differ from those in other nations due to ecological and societal differences. Looking at artificial reefs beyond the lens of just ecological benefits, there are very few studies (both globally and within Indonesia), that have assessed artificial reefs' impact on ecosystem functioning (i.e. nutrient dynamics and carbon storage) and society (i.e. socioeconomic benefits). This research is aiming to address these gaps in the knowledge through the following research objectives and questions, and its linkages are highlighted in figure 1.1.

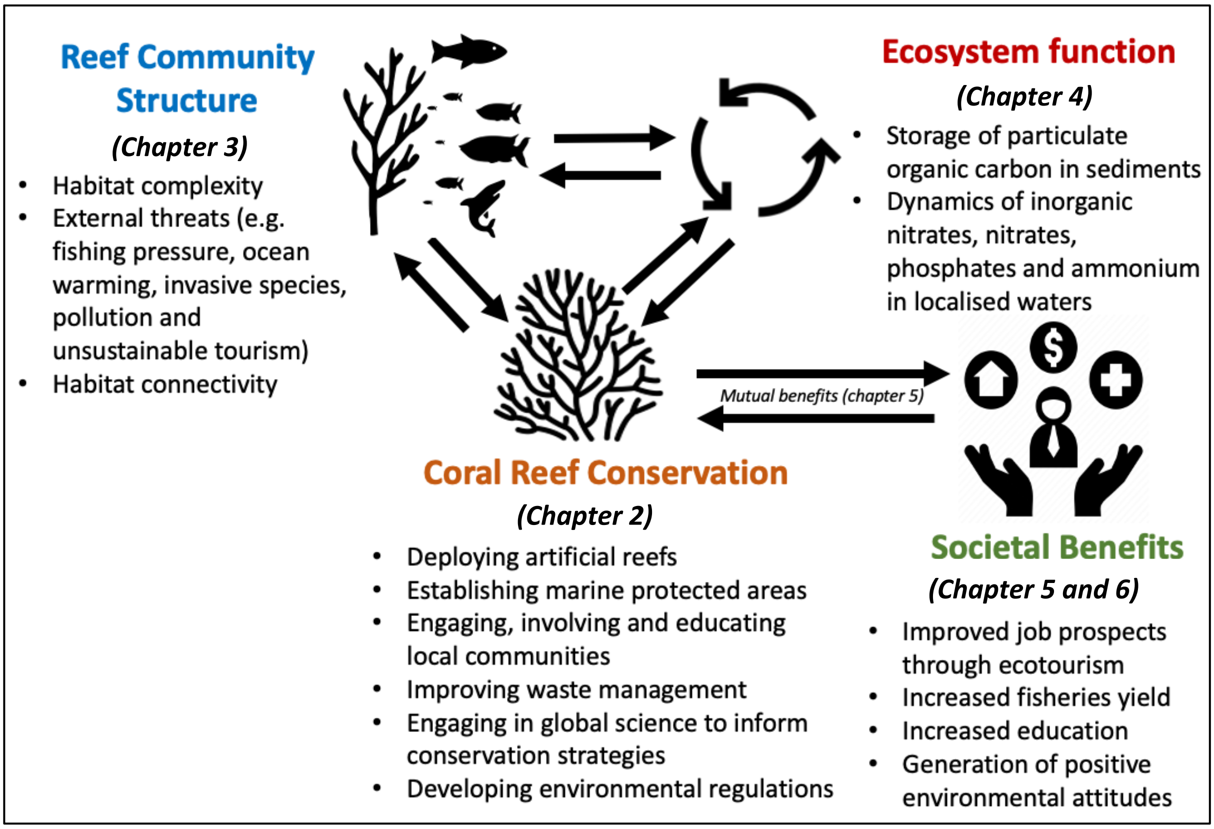

Figure 1. 1: Conceptual diagram highlighting the linkages between the major themes of my PhD thesis.

This work is innovative because it combines coral/artificial reef community monitoring with research focusing on the ecosystem functions and ecosystem services that coral reefs provide, and artificial reefs may mimic. It also involves collecting qualitative data (for example community and tourism surveys), which utilises the important opinions of local people, which are often overlooked in studies which research coral reef conservation (Aswani et al. 2018). Furthermore, this research strikes a balance between showing the global degradation of coral reefs through climate change and habitat destruction, and providing optimism, that some of these functions and services may be able to be provided and restored, at least at a local scale in the future.

This thesis will aim to answer the following research question: To what extent have coral conservation programmes in Bali, Indonesia, restored previously degraded reef communities and generated localised socioeconomic benefits?

PhD Aims, Objectives and Chapters

The overall aim of this PhD thesis has been to investigate if coral reef conservation projects (specifically those that deploy artificial reefs) in Indonesia have restored ecological communities, ecosystem processes, and generated societal benefits (such as ecotourism and pro environmental behaviours). Figure 1.1 highlighted how the concepts of coral reef conservation, reef community structure and nutrient cycling are all interlinked, and (directly and indirectly) lead to societal benefits.

The scientific objectives of my PhD were:
1) To review the current research (both published and grey literature) on the topic of 'coral reef conservation in Bali' and make suggestions on how reefs can be better conserved in Bali, based on international 'best practice' (*addressed in chapter 2*).
2) To assess the development of ecological communities on artificial reefs (deployed by coral conservation projects) and to compare this to other natural habitat types *(addressed in chapters 3 and 4)*.
3) To compare ecosystem functioning, including nutrient storage and dynamics (in sediments, pore water and bottom water) of artificial reefs (deployed by coral conservation projects) to natural habitat types *(addressed in chapter 4)*
4) To assess how coral reef conservation programmes can support ecosystem services and generate societal benefits. *(addressed in chapters 5 and 6).*

This thesis follows the integrated thesis format outlined within Bournemouth University's Research Degree Code of Practice and the details of the integrated papers and their publication status are listed in table 1.1. All publications have co-authors, which includes researchers from various UK and Indonesian universities. I confirm that I am the lead authors of all publications and contributed at least 75% of the substantive content. Aside from minor formatting tweaks, each published paper included within this integrated thesis has not been edited since publication.

Table 1. 1: PhD chapters and their publication status.

Chapter number	Reference	Publication status	Link to PhD objectives
Chapter 2	Boakes, Z., Hall, A.E., Ampou, E.E., Jones, G.C., Suryaputra, I.G.N.A., Mahyuni, L.P., Prasetijo, R. and Stafford, R., 2022. Coral reef conservation in Bali in light of international best practice, a literature review. *Journal for Nature Conservation*, *67*, p.126190.	*This chapter is already published. DOI:10.1016/j.jnc.2022.126190*	Mapping onto objective 1.
Chapter 3	***Initial published paper:*** Boakes, Z., Hall, A., Jones, G., Prasetyo, R., Stafford, R. and Yahya, Y., 2022. Artificial coral reefs as a localised approach to increase fish biodiversity and abundance along the North Bali coastline. *AIMS Geosciences*, *8*(2), pp.303-325. ***PhD chapter:*** Boakes, Z., Hall, A.E., Suryaputra, I.G.N.A. and Stafford, R., 2024. Convergence of tropical fish communities between artificial and natural reefs over time.	*I have already published one paper which utilised the preliminary data from this chapter. DOI: 10.3934/geosci.2022018. This chapter will be expanded over the next year to create a new paper using 5 years of monitoring data. I intend to submit this PhD chapter for publication in 2024.*	Mapping onto objective 2
Chapter 4	Boakes, Z., Suryaputra, I.G.N.A., Hall, A.E., Franklin, D.J. and Stafford, R., 2023. Nutrient dynamics, carbon storage and community composition on artificial and natural reefs in Bali, Indonesia. *Marine Biology*, *170*(10), p.130.	*This chapter is already published. DOI: 10.1007/s00227-023-04283-4*	Mapping onto objective 3.
Chapter 5	Boakes, Z., Mahyuni, L.P., Hall, A.E., Cvitanovic, M. and Stafford, R., 2024. Is tourism helpful or harmful for coral reef health? Stakeholder assessment, actions and management measures in three reef-based tourism areas in Bali Indonesia.	This chapter is not yet published or submitted for publication, as it was the last one I worked on for my PhD. I plan to submit it for publication in 2024.	Mapping onto objective 4.
Chapter 6	Boakes, Z., Mahyuni, L.P., Hall, A.E., Cvitanovic, M. and Stafford, R., 2023. Can Coral Reef Restoration Programmes Facilitate Changes in Environmental Attitudes? A Case Study on a Rural Fisher Community in North Bali, Indonesia. *Human Ecology*, pp.1-15.	*This chapter is already published. DOI: 10.1007/s10745-023-00452-7*	Mapping onto objective 4.

References

Allen, G. R. and Erdman, M. V, 2008. Reef fish of Nusa Penida, Indonesia. *Final Report to Conservation International*, 22.

Aswani, S., Lemahieu, A. and Sauer, W. H. H., 2018. Global trends of local ecological knowledge and future implications. *PloS one*, 13 (4), e0195440.

Ben-Romdhane, H., Jabado, R. W., Grandcourt, E. M., Perry, R. J. O., Al Blooshi, A. Y., Marpu, P. R., Ouarda, T. B. M. J. and Ghedira, H., 2020. Coral reefs of Abu Dhabi, United Arab Emirates: Analysis of management approaches in light of international best practices and a changing climate. *Frontiers in Marine Science*, 7, 541.

Boakes, Z., Hall, A. E., Ampou, E. E., Jones, G. C. A., Suryaputra, I. G. N. A., Mahyuni, L. P., Prasetyo, R. and Stafford, R., 2022. Coral reef conservation in Bali in light of international best practice, a literature review. *Journal for Nature Conservation*, 126190.

Boakes, Z., Hall, A., Jones, G., Prasetyo, R., Stafford, R. and Yahya, Y., 2022. Artificial coral reefs as a localised approach to increase fish biodiversity and abundance along the North Bali coastline. *AIMS Geosciences*, 8 (2), 303–325.

Briggs, J. C., 2005. The marine East Indies: diversity and speciation. *Journal of Biogeography*, 32 (9), 1517–1522.

Burke, L., Reytar, K., Spalding, M. and Perry, A., 2011. *Reefs at risk revisited*. Washington, DC: World Resources Institute.

Costanza, R., De Groot, R., Sutton, P., Van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S. and Turner, R. K., 2014. Changes in the global value of ecosystem services. *Global environmental change*, 26, 152–158.

Fisher, R., O'Leary, R. A., Low-Choy, S., Mengersen, K., Knowlton, N., Brainard, R. E. and Caley, M. J., 2015. Species richness on coral reefs and the pursuit of convergent global estimates. *Current Biology*, 25 (4), 500–505.

Hadi, T. A., Abrar, M., Giyanto, B. P., Johan, O., Budiyanto, A., Dzumalek, A. R., Alifatri, L. O. and Sulha, S., 2020. Suharsono, The status of Indonesian coral reefs 2019. *Research Center for Oceanography-Indonesian Institute of Sciences, Jakarta*, 1–71.

Hein, M. Y., Vardi, T., Shaver, E. C., Pioch, S., Boström-Einarsson, L., Ahmed, M., Grimsditch, G. and McLeod, I. M., 2021. Perspectives on the use of coral reef restoration as a strategy to support and improve reef ecosystem services. *Frontiers in Marine Science*, 299.

Hoegh-Guldberg, O., Poloczanska, E. S., Skirving, W. and Dove, S., 2017. Coral reef ecosystems under climate change and ocean acidification. *Frontiers in Marine Science*, 4, 158.

Hughes, T. P., Baird, A. H., Morrison, T. H. and Torda, G., 2023. Principles for coral reef restoration in the anthropocene. *One Earth*.

Hughes, T. P., Kerry, J. T., Álvarez-Noriega, M., Álvarez-Romero, J. G., Anderson, K. D., Baird, A. H., Babcock, R. C., Beger, M., Bellwood, D. R. and Berkelmans, R., 2017. Global warming and recurrent mass bleaching of corals. *Nature*, 543 (7645), 373–377.

Komyakova, V., Chamberlain, D., Jones, G. P. and Swearer, S. E., 2019. Assessing the performance of artificial reefs as substitute habitat for temperate reef fishes: Implications for reef design and placement. *Science of the total environment*, 668, 139–152.

Lamont, T. A. C., Razak, T. B., Djohani, R., Janetski, N., Rapi, S., Mars, F. and Smith, D. J., 2022. Multi-dimensional approaches to scaling up coral reef restoration. *Marine Policy*, 143, 105199.

Moberg, F. and Folke, C., 1999. Ecological goods and services of coral reef ecosystems. *Ecological economics*, 29 (2), 215–233.

Mustika, P. L. and Ratha, I., 2013. Towards the Bali MPA Network. *In*: *Bali Marine Rapid Assessment Program 2011*. BioOne, 131.

Parsons, E. C. M., MacPherson, R. and Villagomez, A., 2017. Marine "conservation": you keep using that word but I don't think it means what you think it means. *Frontiers in Marine Science*, 4, 299.

Pratchett, M. S., Hoey, A. S. and Wilson, S. K., 2014. Reef degradation and the loss of critical ecosystem goods and services provided by coral reef fishes. *Current Opinion in Environmental Sustainability*, 7, 37–43.
Principe, P., Bradley, P., Yee, S., Fisher, W., Johnson, E., Allen, P. and Campbell, D., 2012. Quantifying coral reef ecosystem services. US Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC.
Razak, T. B., Boström-Einarsson, L., Alisa, C. A. G., Vida, R. T. and Lamont, T. A. C., 2022. Coral reef restoration in Indonesia: A review of policies and projects. *Marine Policy*, 137, 104940.
SER, 2004. Society for ecological restoration international science & policy working group.
Sully, S., Burkepile, D. E., Donovan, M. K., Hodgson, G. and Van Woesik, R., 2019. A global analysis of coral bleaching over the past two decades. *Nature communications*, 10 (1), 1–5.
Tun, K., Chou, L. M., Cabanban, A., Tuan, V. S., Suharsono, T. Y., Sour, K. and Lane, D., 2005. Status of coral reefs, coral reef monitoring and management in Southeast Asia 2004. *In*: .
Wicaksana, I. B. A., 2020. 2020 Bali's Coral Conservation: from pandemic challenge to Government coral garden project. *Bali Tourism Journal*, 4 (2), 35–39.

Chapter 2: 'Coral reef conservation in Bali in light of international best practice, a literature review'

Objectives: To identify good practice and areas for improvement for marine conservation in Bali and wider Indonesia, compared to typical and best practice internationally, and to make recommendations for improvements based on the findings.

Contribution to new knowledge: No studies have assessed coral reef conservation 'best practice' in Indonesia, and very few globally. Many of the findings from this review come in the form of small-scale local studies, including those written in Indonesian language, and therefore not published in an international journal. It is important to have include these small-scale studies within this literature review, as many provide the most thorough insight into the current situation in Bali and wider Indonesia. This literature review raises awareness of these important findings, which may not be not otherwise be available to the international scientific community.

How this fits in the PhD: This chapter provides an overall context for coral reef conservation within Bali and allowed me to assess what is and isn't already known in this field (mapping onto objective 1).

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Abstract:

Bali, Indonesia sits within the coral triangle and is internationally recognised for its high coral reef diversity. The health of Bali's marine ecosystems has declined in recent decades, and this is thought to be due to threats from climate change, destructive fishing practices, pollution, outbreaks coral eating invertebrates, coral disease and unsustainable tourism. As a response, multiple conservation strategies have been introduced by the island's communities, non-government organisations and governments, with the aim of preventing further decline, as well as restoring already degraded coral reefs. This literature review provides an in-depth analysis of the tools used to conserve Bali's coral reefs, and compares them to those used in other countries. In light of international 'best practice' in coral reef conservation, this review makes suggestions on how Bali could better conserve its coral reef ecosystems. These include (1) increasing its designation of official Marine Protected Areas (MPAS) and strengthening management of existing ones, (2) creating an MPA network, (3) substantially reducing marine plastic pollution, (4) continuing artificial reef construction in degraded habitats, (5) continuing to develop Bali as an ecotourism destination, (6) increasing engagement in global science to inform marine conservation decision-making, and (7) developing more marine monitoring programmes.

Key words: Coral Reef Restoration, Marine Conservation, Indonesia, Artificial Reefs, Ecotourism

Introduction

Coral reefs: A global perspective

Coral reefs, large underwater habitats of calcium carbonate skeletons produced over time by coral polyps, are critically important to tropical coastlines (Hoegh-Guldberg et al. 2017). Often referred to as 'rainforests of the sea', coral reefs occupy less than 0.1% of the ocean floor, yet host 25% of the world's marine species (Fisher et al. 2015). They provide ecosystem services estimated at a value of over US $1 Trillion globally (Costanza et al. 2014) through food provision, shoreline protection, biogeochemical cycling and tourism (Moberg and Folke 1999, Principe et al. 2012). The provision of these ecosystem services is under threat (Bell et al. 2006) as anthropogenic activities have caused a worldwide long-term decline in coral reef biodiversity, abundance and habitat structure (Pandolfi et al. 2011, Hughes et al. 2018). The cumulative effect of this damage has resulted in declines of associated nearshore tropical biodiversity (Pratchett et al. 2014), altering ecosystem functioning and processes (Richardson et al. 2018).

The first United Nations Educational, Scientific and Cultural Organization (UNESCO) assessment of global coral reef decline predicts that all 29 coral-containing World Heritage sites will no longer be functioning coral reef ecosystems by 2100 under a business-as-usual emissions scenario, due to coral bleaching mostly associated with ocean warming and acidification (Heron et al. 2017). The same study indicates that climate-related losses of reef ecosystem services will total approximately US $500 billion by 2100, with the greatest of these impacts experienced by people who rely upon reef services for day-to-day subsistence Under these scenarios, it is predicted that reefs previously dominated by hard and soft corals, will experience regime shifts, changing the ecosystem to one that is instead dominated by algae (Vercelloni et al. 2020). Alongside aggressive and immediate global-scale interventions to reduce greenhouse gas emissions and their impact on coral reefs (as highlighted by Pörtner et al. (2014) by the International Panel for Climate Change (IPCC) 'Ocean Systems' report), various other local scale options may be considered to offset the decline of coral reef biodiversity, abundance and habitat structure.

Introduction to Indonesia's / Bali's coral reefs

Indonesia makes up 12.5 % of the world's total coral reef area (Susiloningtyas et al. 2018). It sits within the Coral Triangle, an area recognised as the global centre of marine biodiversity (Allen 2008) which is of global conservation importance (Briggs 2005). Bali is a province of Indonesia (figure 2.1), and has the second highest documented reef fish species richness in the Asia-pacific (Mustika and Ratha 2013), with at least 805 documented fish species (Allen and Erdmann 2013).

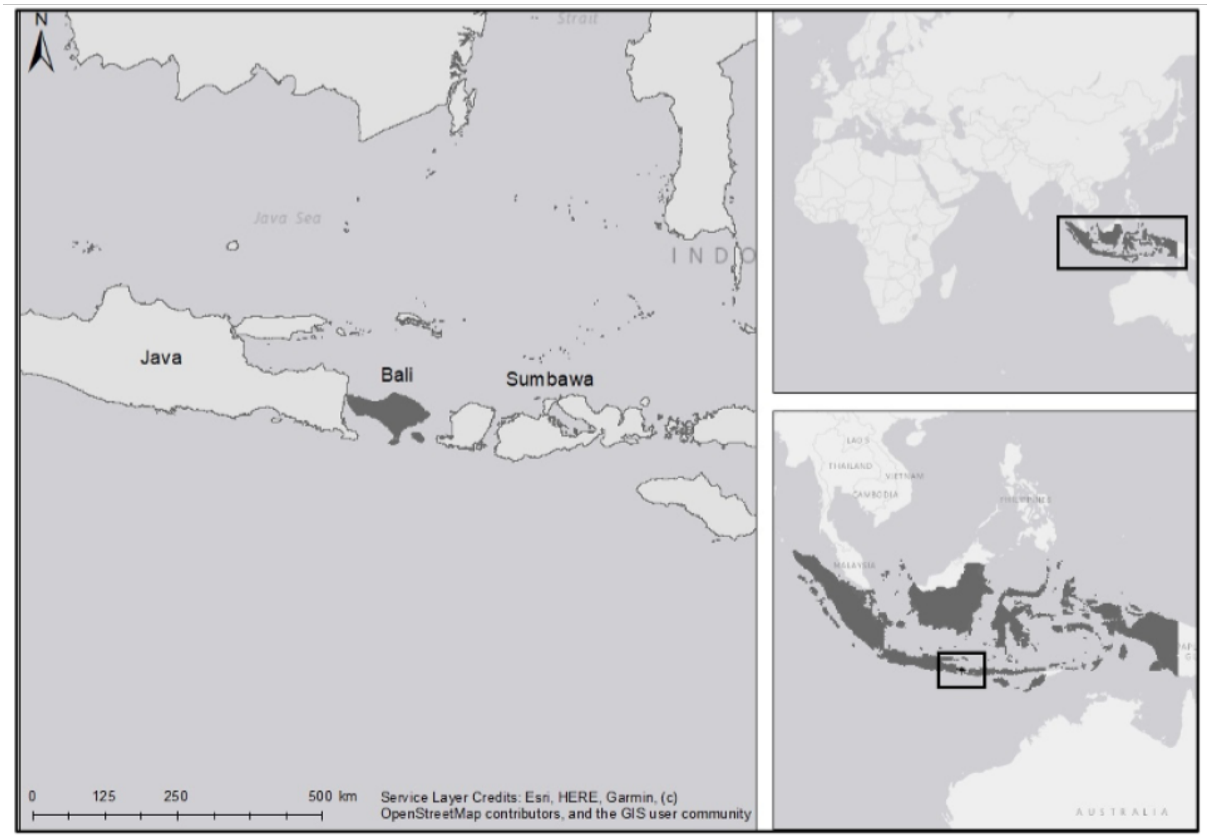

Figure 2. 1: Location of Bali within Indonesia *(Created using ArcGIS OpenStreetMap powered by Esri).*

Research suggests that 86% of Indonesia's coral reefs face medium or high levels of threat (Burke et al. 2012). Studies on Bali's reef in 2011, collected from the 27 reefs across the island, showed that its corals were generally in good condition (Lazuardi et al. 2013). More recent data from 2017 has highlighted similar results, suggesting that 50% of its corals are in good health, whilst 20% are declining and 30% are poor (Marine and Fisheries Office 2017 data, as cited by Wicaksana (2020). Reefs in Bali are exposed to multiple threats, that combined together, make the ecosystems less resilient rising sea temperatures (Salm 2005). Resilience can be defined as "the capacity of a system to resist and recover from disturbance and undergo change while still retaining essentially the same function, structure and integrity” (GBRMPA 2018). A reduction in coral reef resilience through combined threats can result in coral mortality and regime shifts, where the reef will become dominated by algae instead of coral, as reported in reefs across Bali (Tito and Ampou 2020). This alternative algal state is generally viewed as less desirable in terms of the provision of ecosystem services and it is unlikely, especially with rising temperatures, that a coral reef will recover to its original state after a regime shift (Selgrath et al. 2017).

As discussed by Ridley (2012), a literature review is “in itself a research study, using the literature as data to be coded, analyzed and synthesized to reach overall conclusions”. This literature is aiming to identify good practice and areas for improvement for marine conservation in Bali and wider Indonesia, compared to typical and best practice internationally. It will start by identifying the main threats to the coral reefs of Bali, then main tools for conservation in Bali will be discussed and analysed. This review will end by making suggestions for marine conservation in Bali, informed by internationally recognised ‘best practice’.

Methods

The purpose of this study was to provide a comprehensive review of marine conservation issues in Bali, acknowledging that much of the relevant literature would not be present in peer reviewed papers or published in English. We therefore used Google Scholar as the main search engine. Key themes related to the research title were selected and within each theme key discussion points chosen. For example, the within the theme ‘Coral Reef Threats’, the key points included: ‘Bleaching’, ‘Nutrient Enrichment’, ‘Damaging SCUBA practices’, ‘Coral Disease’, ‘Crown of Thorns Starfish’, ‘Plastic Pollution’ and ‘Destructive Fishing Practices’ (see also subheadings below for full list). Following the methods of Lison et al. (2020), each theme and key point was then systematically searched for in relation to Bali, for example ‘Destructive Fishing Practices Bali’. The use of Google Scholar was key here as the term Destructive Fishing Practices AND Bali returned only two references in Web of Science, whereas over 16,000 results were returned from Google Scholar. Relevant papers were selected, normally from the first three pages of results sorted by relevance, based on the examination of the paper title and abstract. Additional grey literature was obtained through local knowledge of many of the authors, as well as contacts with local government departments and NGOs.

Results/ Discussion

The most substantial threats to the reefs of Bali are listed and discussed below:

Coral bleaching

Anthropogenic greenhouse gas emissions have led to an increase in atmospheric carbon and average global temperatures (Clark et al. 2020). As a consequence, ocean surface temperatures are thought to have increased by approximately 0.4 - 0.5°C since the 1980s (Pörtner et al. 2014). The ocean has also absorbed approximately half of the anthropogenic atmospheric CO2 emissions in the past 200 years (Raven et al. 2005) and this has reduced the ocean surface pH by more than 0.1 (Pörtner et al. 2014). The increase in ocean temperatures, alongside the reduction in ocean pH has been attributed with worldwide coral bleaching (Heron et al. 2017).

Coral bleaching occurs as a stress response to changes in temperature, and results in a loss of the coral's endosymbiotic dinoflagellates (Lesser 2011), which leads to a decrease in growth rate and fertility and can result in mortalities (Sully et al. 2019; Ampou et al. 2020). Corals have a limited temperature threshold which they are able to tolerate, and localised increases of 1-2 °C can result in severe bleaching events (Ampou 2020). These events are predicted to increase in frequency in the future, and consequently, it is thought that 90% of global coral reefs may be at risk of long-term degradation (Grottoli et al. 2014). After a bleaching event it is possible for a coral reef to recover, but in these situations they are under greater stress and are more subject to mortality from other threats (Normile 2016). If a reef is resilient, it may be able return to its original state after a disturbance (Salm 2005, Ampou et al. 2020), although resilience varies between coral species, with some being more vulnerable to threats than others (Roche et al. 2018). This can be further explained by the research of Foden et al. (2013) which highlighted that 15-32% of coral species have high sensitivity and low adaptive capacity to climate change, and are therefore most vulnerable to climate change. As a result of this, studies have highlighted the loss of the most vulnerable coral species on a reef, resulting a loss heterogeneity and ecosystem function (Strychar et al. 2005).

Coral bleaching has been documented on multiple coral reefs in Bali, including southern reefs in Sanur, Nusa Dua and Serangan (Wicaksana 2020), northern reefs in Buleleng (Suparno et al. 2019, Tito et al. 2019) and the reefs of Nusa Penida and Nusa Lembongan (Prasetia et al. 2017). It is one of the most substantial threats to the reefs of Bali and has been attributed with a loss of live coral cover of 44.4% in North West Bali (Suparno et al. 2019). Most recent data has highlighted the occurrence of a bleaching event in May 2020, caused by a significant increase in sea surface temperature, widespread across all of Bali's coasts (Ampou 2020).

Destructive fishing practices

Destructive fishing practices (DFPs) are any method used by fishers that causes direct damage to the surrounding habitat (Bacalso and Wolff 2014). DFPs known to be used in Indonesia include blast (dynamite), cyanide fishing and inshore trawling (Erdmann et al. 2000). Cesar et al. (1996) discussed that DFPs not only result in exploitation of a local fishery, but also cause substantial physical damage to the surrounding habitat structure (usually hard substrata like corals) on which commercial species depend. Despite being illegal, the use of DFPs is thought to be widespread across Indonesia (Pet-Soede and Erdmann 1998). Estimates suggest that up to 80% of the country's coral reefs have been targeted by DFPs, which are used more frequently in poorer regions, often by communities that are experiencing poverty and/or insufficient fish catches from standard, less destructive techniques (Erdmann et al. 2000). There is limited available information on current use of DFPs in Bali, although it is thought that dynamite fishing, which uses an explosive blast that instantly kills the fish (as well as destroying the surrounding habitat), was still in use in some regions in 2013 (Doherty et al. 2013). Additionally, cyanide fishing, which increases mortality of target and nearby non target species (Madeira et al. 2020), was made illegal under Indonesian law in 1985 (Fisheries Regulation Act, 1985; Halim (2002)). It was previously a widespread method used by ornamental fishers across Bali to catch live fish sold for the aquarium trade (Frey and Berkes 2014), and was still thought to be in use in 2013 (Doherty et al. 2013). Since experiencing a decline in the health of their coral reefs as a result of cyanide and dynamite fishing, some communities in Bali have replaced their use with more sustainable harvesting methods (Frey 2013).

Plastic pollution

Worldwide plastic production is thought to have increased from approximately 1.5 million tonnes in 1950 to 322 million tonnes in 2015 (Villarrubia-Gómez et al. 2018). This exponential rise in the global production of plastics, as well as a mismanagement of its disposal, is estimated to have led to between 4.8 and 12.7 million tonnes of plastic entering the oceans per year (Jambeck et al. 2015). The occurrence of plastic debris has now been documented across coastlines worldwide (Barnes et al. 2009). Plastic entering the sea is of global concern due to its persistence in marine environment and its impact on wildlife and potentially humans (Barnes et al. 2009). Marine plastic pollution has also been shown to attract persistent organic pollutants (POPs), and has been linked with the ingestion of these pollutants by marine megafauna (Clukey et al. 2018). Despite these threats, the extent to which plastic pollution is harmful to marine environment is debatable, especially when compared to other threats from climate change and overfishing (Stafford and Jones 2019).

Increased demand for single use plastics (Sur et al. 2018), alongside a lack of expenditure in its waste management (Glaser et al. 2010), has led to Indonesia becoming the world's second largest plastic polluter (Shuker and Cadman 2018). Unsurprisingly, plastic pollution is therefore a substantial issue in Bali (Turak and Devantier 2013, Giesler 2018, Brooijmans et al. 2019). Much of the islands plastic is disposed of by being dumped in rivers or the sea (posing serious direct marine pollution threats (Lestari and Trihadiningrum 2019)) or by being burnt (releasing organic aerosols thought to pose serious risks to human health and the environment (Velis and Cook 2020)). Most recent data on plastic pollution in Benoa Bay, South Bali indicates that microplastics are abundant in Bali's marine environment, being detected in the surface waters of all four research stations (Suteja et al. 2021). There is currently limited literature which assesses the impacts of plastic pollution on coral reefs in Bali, and it can be assumed that the issues highlighted above can be applied to the situation in Bali. Germanov et al. (2019) used boat trawls in Bali and its neighbouring island of Java, and concluded that plastic abundance in these marine environments ranged from 20,000 – 449,000 pieces km–2, with higher estimates in the wet season due to increased land run off. The same study suggested that reef manta rays (*Mobula alfredi;* Krefft 1868), which are listed as vulnerable on the International Union for Conservation of Nature (IUCN) red list (Marshall et al. 2019), may ingest between 110 and 980g of plastic for every kg of plankton. The bio accumulative ingestion of plastics by manta rays and other mega faunal filter feeders has been shown to cause endocrine disruption, as well as altering reproductive fitness and potentially offloading toxicity from a mother to her offspring (Germanov et al. 2018). Despite the limited literature available, it is clear that that marine plastic pollution is a substantial issue to the marine ecosystems of Bali. More research is required to quantify the extent of this threat.

Crown of Thorns Starfish

Outbreaks of the Crown of thorns starfish (CoTS; *Acanthaster planci;* Linnaeus, 1758), are a substantial threat to coral reef ecosystems (Deaker et al. 2020). CoTS are coral eating invertebrates native to the Indo-pacific. They are not considered a substantial threat in 'normal' reef populations, however their numbers can increase dramatically due to an increase nutrient supply (Brodie et al. 2005), amongst others. This is thought to occur because nutrient loading increases phytoplankton abundance, which provides a reliable food source for CoTS larvae (Fabricius et al. 2010). Brodie et al.

(2005) showed that when phytoplankton concentrations double, CoTS' chance of survival to adulthood can increase almost ten-fold.

CoTS outbreaks have resulted in a 50% loss of coral cover on some reefs in Indonesia (Plass-Johnson et al. 2015). Multiple studies have highlighted the threat of CoTS within North West Bali. Suparno et al. (2019) documented CoTS populations within Bali Barat National Park (BBNP) in North West Bali in 2016 and suggested that their presence may be due to effluent from a local shrimp farm. Doherty et al. (2013) discussed how outbreaks of CoTS, as well as the coral eating drupella snail (D*rupella cornus;* Röding 1798) which are known to cause similar impacts (Al-Horani et al. 2011), have resulted in mass deaths of corals around Menjangan Island, in North West Bali. A more recent study of Menjangan Island, has shown that CoTS outbreaks are predicted to occur during Bali's wet season due to increased nutrient loading (Pradisty et al. 2020). There are multiple community projects and Non-Government Organisations (NGOs) that work to remove CoTS in Bali. Some of this is documented within literature, such as the 1997 programme in BBNP, which removed more than 700,000 CoTS individuals (Boekschoten et al. 2000).

Coral disease

Outbreaks of coral diseases have caused devastating mortalities to reefs around the globe (Walton et al. 2018) and are increasing with frequency and severity (Maynard et al. 2015). 'White Syndromes' (WS) are described as the most destructive and widespread group of worldwide coral diseases (Bruno et al. 2007, Hobbs and Frisch 2010), and have been associated with mortalities as high as 96% of *Acropora* (Oken, 1815) plate corals on some coral reefs (Hobbs and Frisch 2010). Coral diseases are associated with mortalities of corals in reefs on the Great Barrier Reef, Red Sea, Caribbean, Philippines (Williams and Miller 2005, Aronson and Precht 2006, Hobbs and Frisch 2010).
Coral diseases also appear to also be widespread across reefs in Indonesia's national marine parks and on its most diverse reefs (Johan et al. 2015, Ampou et al. 2020). They are documented in Bali, on reefs in Buleleng (Karim 2019, Suparno et al. 2019) and Nusa Penida (Ampou 2018), although research on this is relatively limited. Most literature on coral disease in Bali/ Indonesia appears to have been undertaken in the past 2 decades. This is likely because coral disease is thought to be a relatively recent issue, perhaps because it is linked with rising sea temperatures (Aeby et al. 2020, Ampou et al. 2020), which has also gained more research attention in recent decades (Pörtner et al. 2014). The link between coral diseases and thermal stress has been further studied by Bruno et al. (2007), who found a highly significant relationship between rising sea temperature and increased emergence of coral disease outbreaks. Current literature suggests that coral diseases, alongside

bleaching, are one of the greatest threats to some of Indonesia's coral reefs (Subhan et al. 2020). More research is required to identify the causes of coral disease outbreaks, as well as quantifying their overall threats and outlining potential management methods.

Damaging SCUBA practice

The tourism sector makes up approximately 68% (in 2014) of Bali's GDP (Antara and Sumarniasih 2017). According to Gerungan and Chia (2020), Bali has some of the best SCUBA diving sites in South East Asia, which are an important source of income for coastal communities (Tapsuwan and Rongrongmuang 2015). For example, one of Bali's most famous dive locations, Tulamben in North East Bali, has 14 dive centres that generate income for a village (in the poorest region of Bali) that was previously almost entirely reliant on subsistence fishing (De Brauwer et al. 2017).

Dive tourism, however, has been associated with negative environmental consequences when there is a lack of management (Haddock-Fraser and Hampton 2012). One of the most substantial ecological impacts of dive tourism is the physical damage to corals caused by divers who lack experience or respect environmentally conscious dive practices (Davenport and Davenport 2006). Divers swimming too close to the sea floor can stir up benthic sediments, which smothers the corals (Abidin and Mohamed 2014). Coral skeletons may also be broken if divers step on or accidentally collide with them (Mastny 2001). In both cases, this is thought to negatively affect the coral's biological processes, including growth and sexual reproduction (Davenport and Davenport 2006). Another environmental impact of dive tourism can be overfishing/exploitation of a local fishery, as the demand for fish increases in tourist restaurants (Tompkins 2003).

There is existing literature that assesses the impact of dive tourism on the coral reefs of Bali. Suparno et al. (2019) discussed how SCUBA diving activities across the island have been correlated with structural damage of coral reefs. A substantial increase in broken and upturned corals between 2002 – 2011 was observed in Bali Barat National Park dive sites by Doherty et al. (2013), who attributed this to diving boats having inadequate access to mooring buoys, and instead using anchors which destroy the corals. Gerugan and Chia (2020) highlighted how scuba dive tourism at one of Bali's most famous dive sites, 'Manta Bay', in Nusa Penida is poorly managed by dive centres, and consequently frequently reported stepping on or colliding with corals. The same study interviewed local people, who agreed that dive tourism has contributed towards the degradation of the 'Manta Bay' coral reef.

Despite tourism being associated with the degradation of corals, it is thought that it may be helping to protect the charismatic species that gives 'Manta Bay' its name. Manta rays (*Mobula alfredi;* Krefft 1868) are heavily fished in some parts of Indonesia (Lewis et al. 2015). There is also a large international demand for non-consumptive manta ray dive tourism, which is calculated to have an industry value of USD $140 million per year worldwide (O'Malley et al. 2013). It is thought that manta rays may be worth up to $1 million each when they are alive (through the tourism income they generate), compared to $500 when they're fished (Hani et al. 2019). In sites like 'Manta Bay', dive tourism (and the income generated from it) provides a compelling reason to protect *Mobula alfredi* (Krefft 1868), and is the main driver of strict regulations which prohibit all extractive activities. Hani et al. (2019) commented that sustainable manta ray dive tourism at 'Manta Bay' and other sites in Indonesia requires strict governance, adequate regulations/enforcement and collaborative management.

A study by Piskurek (2001) assessed the sustainability of dive tourism in Pemutaran, North West Bali. The study concluded that divers cause very little damage to the Pemutaran reef, especially when compared to the threats caused by pollution and overfishing. The contrasting results of this study may be due to the stricter diving regulations limiting the number of divers permitted on the reef, as well as mooring buoys to stop boats anchoring and rest stations for tired divers and snorkellers (to reduce stepping on corals). This case study provides a promising example amongst many negative ones, that the ecological consequences of dive tourism can be reduced with adequate management.

Nutrient enrichment

A decline in coral reef health is frequently linked to nutrient enrichment (Szmant 2002, D'Angelo and Wiedenmann 2014). Although the in-situ effects are mostly non-lethal and modest (Koop et al. 2001), research has shown that the increase of nutrients level can lead to coral diseases (Bruno et al. 2003, Voss and Richardson 2006, Vega Thurber et al. 2014, Lapointe et al. 2019), coral bleaching (D'Angelo and Wiedenmann 2014, Vega Thurber et al. 2014, Lapointe et al. 2019), outbreaks of CoTS (Fabricius et al. 2010), a decrease of coral growth (Ferrier-Pagès et al. 2000, Loya et al. 2004, Lapointe et al. 2019) and phase shifts to algae dominated reefs (Baum et al. 2016, Adam et al. 2021). Nutrient enrichment has been highlighted as an issue in Bali, due to high concentrations of nitrates and phosphates from river discharge; however there have only been a few studies in Indonesia which explore the link between nutrients enrichment and coral decline (e.g. Baum et al. 2015, 2016;

Faizal et al. 2020), and no such studies are known in Bali. More research in Bali is required to quantify water quality, and study its link to coral health.

Threats to Bali's reefs: an island perspective

It must be noted that the threats to Bali's reefs vary spatially across the island, and also vary in terms of their severity/associated consequences. For example, CoTS outbreaks have only been documented in the reefs in North West Bali (Doherty et al. 2013) and there is no other literature to suggest that they threaten reefs in other regions of the island. Similarly, SCUBA diving activity is limited to a few dive sites. It is not known to occur on a large proportion of the islands reefs and is therefore of limited threat to the islands total coral reef biodiversity.

In 2020, half of Bali's corals are thought to remain unbleached (Wickasana 2020), and it appears that the severity of bleaching may be far worse in other countries. Coral bleaching is now generally accepted as the primary threat to coral reefs globally (Boström-Einarsson et al. 2020) and has been linked with severe coral mortalities on world heritage listed reefs around the world (Heron et al. 2017). The great barrier reef (GBR) in Australia may be most affected by bleaching in terms of total coral losses (Lewis and Mallela 2018). It is of importance to mention this, and to highlight how the total bleaching of Bali's reefs appears to be relatively low compared to other parts of the world.

Marine restoration and conservation

'Ecological restoration' is defined as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER 2004). In comparison, 'ecological conservation' is a slightly broader term, which incorporates preservation and protection, as well as restoration (Parsons et al. 2017). So far, this literature review has discussed some of the main threats to Bali's coral reefs. The remainder of this review will now focus on the restoration and conservation of coral reefs, whilst specifically looking at what has been undertaken, both past and present, globally and in Bali. We discuss methods used, and evaluate their overall successes and failures.

Climate change mitigation

As previously highlighted, coral bleaching is generally regarded as the greatest global threat to coral reefs (Boström-Einarsson et al. 2020). Therefore, effective coral reef conservation should include a global mitigation of climate change through aggressive and large reductions in greenhouse gas emissions (Ben-Romdhane et al. 2020). The 2015 Paris Agreement is the most recent international treaty to address climate change, and includes an agreement, signed by 196 countries, which aims to limit global warming to well below 2 degrees compared to pre-industrial levels (Schurer et al. 2018). As highlighted by the IPCC, climate change mitigation will require global efforts to reduce net emissions from energy supply, transport, buildings, industry, agriculture and land/ natural resource use (Edenhofer et al. 2014).

The 2020 European Commission report on global greenhouse gas (GHG) emissions ranks Indonesia as the 9^{th} highest greenhouse gas emitter compared to the rest of the world (Crippa et al. 2020). The same report highlighted how Indonesia's total GHG emissions are still 58x lower than China's and 18x lower than the US'. Indonesia is one of the world's largest producers and exporters of coal (Dwiki 2018) and it is deforesting its rainforests faster than any other nation (Tacconi et al. 2019). Whilst it is out of the scope of this review to provide an in-depth assessment of Indonesia's climate change contributions and mitigations, it must be noted that Indonesia has joined the Paris Agreement, and alongside other newly developed countries, it aims to reduce emissions by 29% by 2030, compared to a business-as-usual scenario (Wijaya et al. 2017). Indonesia has recently declared it aims to reach net-zero emissions by 2070 (van Soest et al. 2021), a target which has been criticised as unambitious by activities and other governments.

Coral reef conservation initiatives

Global initiatives are not just limited to climate change, they also exist to directly protect the world's coral reefs. Previous examples include the 1992 convention on Biological Diversity (Bell 1992) and the 2000 International Coral Reef Network (ICRAN) (Ben-Romdhane et al. 2020). A more recent initiative is the 'Aichi Targets', a strategic plan to conserve international coral reef biodiversity developed at the 2011 – 2020 Convention of Biological Diversity (Leadley et al. 2014).

As well as global greenhouse gas emission reductions and international coral conservation initiatives, multiple small/medium scale tools may be considered to conserve, restore and increase the resilience of coral reefs. These methods may include coral transplantation (Endo et al. 2008, Onaka et al. 2013, Barton et al. 2017, Baria-Rodriguez et al. 2019), building coral nurseries with coral species that are resistant to bleaching (Camp et al. 2017, Morikawa and Palumbi 2019), community development and education (Sigit et al. 2019), marine protected area establishment (Edgar et al. 2014, Zhao et al. 2020), genetically modifying reef building corals (Cleves et al. 2018), constructing artificial reefs (ARs) (Bohnsack and Sutherland 1985, Baine 2001, Keller et al. 2017) and coral microbiome manipulation (Rosado et al. 2019). Whilst these methods may be unable to conserve large-scale ecosystem function and processes (Pörtner et al. 2014), they have been shown to provide some degree of protection at a localised level and in some cases, restore ecosystem services in areas which have lost reefs. Each method varies in terms of its overall effectiveness, implementation feasibility and how well researched it is. The remainder of this review will discuss what has already been done to restore and conserve Bali's reefs, and how this relates to what is being undertaken on a global scale.

Marine Protected Areas

Marine protected areas (MPAs) impose regulations on marine areas as a natural conservation and social management tool to enhance the ecological resilience of a marine area (Costello 2014). The designation of MPAs is increasing worldwide (Edgar et al. 2014) and the IUCN has recently called for the 'full protection' of 30% of the worlds oceans by 2030 (Zhao et al. 2020), due to their global importance in protecting marine ecosystems from the effects of human exploitation and activities (Perez et al., 2017; Marcos et al. 2021). 'Full protection' MPAs are areas which are completely closed off to all extractive activities, and as highlighted by Perez et al. (2017) provide three main benefits: (1) preserving of biological diversity at a regional level, (2) allowing the natural variability of the system to be differentiated from the effects of regulation and to be integrated in to sampling schemes as controls, and (3) maintaining the natural size and age structure of natural populations and therefore maximizing potential fecundity. However, MPAs may fail to reach their targets, with Marcos et al. (2021) discussing how many existing MPAs are mere "paper parks", where legislation is not enforced, necessary enforcement does not exist and management planning is lacking.

Indonesia's Ministry of Marine Affairs and Fisheries has established marine reserves which aim to protect marine biodiversity, whilst supporting sustainable fisheries and tourism (Ruchimat et al. 2013). However, it is thought that less than 15% of the country's MPAs are functionally meeting their management objectives (Burke et al. 2012). Marine reserves in Bali have undergone multiple successes and failures since their inception in the 1970s (Polunin et al. 1983). Some of the first reserves such as BBNP in West Bali (Polunin et al. 1983), as well as Ambon reserve at Pombo Island in North East Indonesia (Sumadhiharga 1977) were unsuccessful in achieving their aims. In situations like these, DFPs and other ecologically-harmful activities continue unregulated within the reserve, as seen with multiple examples in Indonesia (Robinson et al. 1981).

Table 2. 1: Summary of Bali's three officially designated marine protected areas.

Location	Size	Details	Ecological successes / failures	Community perception
Bali Barat National Park, North Bali	34 km^2 (Mahmud et al. 2016)	This was Bali's first MPA, which was established in the early 1970s (Polunin et al. 1983). The MPA uses a zoning system, which includes core zones, marine protected zones, utilisation zones and traditional zones (Mahmud et al. 2016).	Suparno et al. (2019) showed that between 2011 - 2016 fish biomass had doubled on average across the MPA. However, in some areas of the reserve, regulations were poorly enforced and the use of DFPs continued to occur (Doherty et al. 2013). Suparno et al. (2019) discussed that the unclear boundaries in the MPA has resulted in non-compliance with regulations. Suparno et al. (2019) also revealed that between the same dates, the reserve had lost 44.4% of its living coral cover. This coral mortality was primarily associated with the 2016 global bleaching event.	Most local fishers believe the health of the reef ecosystem within the MPA had worsened since 2010 (Pedju 2018), likely due to coral bleaching (Suparno et al. 2019) and stakeholder noncompliance (Doherty et al. 2013).
Pemutaran, North Bali	*Unknown*	The Pemutaran village notake-zone was established by a community conservation organisation in 2003, and was given official MPA status in 2014 (Pedju 2018). The MPA's regulations were established and are enforced by the community (Bottema and Bush 2012).	The MPA is part of an integrated local conservation initiative, which includes other projects, including a turtle hatchling conservation centre (Suparno et al. 2019) and the Biorock TM artificial reef programme (Hilbertz and Goreau 1996). This community work has received multiple UN coastal management awards Trialfhianty 2017) and has resulted in substantial increases in Pemutaran's marine biodiversity (Jamison, 2009).	The majority of local fishers believe the health of reef ecosystem within the MPA had improved since 2010 (Pedju 2018).
Nusa Penida	20 km^2 (Yunitawati	Established in 2010 (Daulat et al. 2019), the	Ruchimat el al. (2013) criticised the lack of clear zone boundaries within the MPA, and commented	The majority of local fishers

Island, East Bali	and Clifton 2019)	marine reserve protects15 km^2 of coral reefs and hosts charismatic marine megafauna which attracts over 200,000 tourists per year (Ruchimat et al. 2013). The Nusa Penida marine reserve uses a zoning system, which includes zones for tourism, fishing, seaweed farming and religious activities (Ruchimat et al. 2013).	that this has led to certain stakeholders not complying with the regulations. Despite this, initial surveys of the MPA recorded a doubling of fish biomass between 2010 – 2012 (unpublished data, discussed by Yunita and Clifton (2019)). Since then, there has been a lack of follow up surveys, so it is difficult to draw further conclusions. Yunita and Clifton (2019) also discussed how coral cover within the reserve has remained stable at around 70% between 2011 – 2016, which would be considered 'excellent' condition by Indonesia's standards (Zamani and Madduppa 2011). Weeks et al. (2014) commented that other progress within the reserve includes the development of a long term management plan, strict enforcement through regular patrols and the establishment of a multi stakeholder task force. The reserve also has a designated learning site which offers training on MPA principles, zone planning, financing and general management.	believe the health of reef ecosystem within the MPA had improved since 2010 (Pedju 2018).

Table 2.1 outlines the successes and failures of Bali's three official MPAs. Despite challenges associated with lack of clear zonation and user non-compliance, it is evident that the three MPAs within Bali have contributed towards the conservation of the marine environment, although some have been more successful than others.

Artificial reefs

ARs are structures built of natural or man-made materials which are designed to protect, enhance, or restore components of marine ecosystems (Baine 2001). Once placed on the sea floor, ARs can restore a previously degraded and/or unproductive ecosystem by providing previously unavailable resources for both juvenile and adult species (Becker et al. 2017, Israel et al. 2017). It is of general agreement that ARs are effective at attracting fish and thus, can be important within fisheries management (Bohnsack and Sutherland 1985). However the 'attraction versus production' debate (Pickering and Whitmarsh, 1997) remains topical within current AR literature (Roa-Ureta et al. 2019).

Baine et al. (2001), discussed that as well as restoration and conservation, ARs have multiple other purposes in coastal management. Some of these include increasing fisheries yield (Bohnsack and Sutherland 1985, Keller et al. 2017), boosting dive tourism (Kirkbride-Smith et al. 2016, Bideci and Cater 2019), coastal protection (Harris 2009) and preventing bottom trawling (Fabi and Spagnolo 2011). Literature has demonstrated the potential of ARs to mitigate habitat loss (Baine 2001), increase larval and juvenile recruitment, survival, and growth (Bohnsack and Sutherland 1985) and maintain biodiversity in marine systems (Becker et al. 2017).

Artificial reefs in Bali

The first officially reported instalment of ARs in Indonesia was the 1989 deployment of 60,000 AR units in Jakarta Bay (Azis 2010, cited by Puspasari et al. (2020)). AR deployment in Bali's marine environment is now widespread, from the coastal seas of Bali's largest tourism hubs, such as Nusa Dua, to some of Bali's poorest regions like Buleleng and Karangasem (Puspasari et al. 2020). The materials used to build Bali's AR structures vary greatly, with some of the most common including concrete substrate blocks, reef balls and Biorock (Global Coral Reef Alliance, Cambridge, MA). Multiple organisations are responsible for the deployment of ARs in Bali, including international NGOs, community groups, village governments and the central government. The use of ARs as a habitat enhancement tool is becoming widespread within Indonesia. A restoration programme in Buleleng, North Bali, extending across 6 villages, built and deployed 13,000 AR structures in 2020 (LINI 2021). The current Indonesian government is committed to protecting its coral reefs, and will invest 1.5 trillion IDR (Approximately 105 million USD) in labour intensive coral reef restoration activities like artificial reef deployment and coral monitoring (Karunia 2021). From this fund, 111.2 billion IDR (Approximately 8 million USD) will be spent on coral reef restoration in Bali, in areas including Sanur, Serangan, Pandawa and Buleleng (Wicaksana 2020).

The ecological success of ARs can be categorised by benthic species and mobile species, and factors which can impact this success are highlighted in table 2.2. With regard to benthic species, studies in Bali and wider Indonesia showed mixed results in terms of ARs potential to increase abundance and diversity. For example, in Seribu Islands, close to Jakarta, Azis (2010), as cited by Puspasari et al.

(2020), showed that coral cover only increased 6% from the initial condition after 10 years of AR instalment. In contrast, ARs in Jemeluk Bay, Karangasem, Bali, increased coral cover by 59% over 15 years (Hartati 2017). With regard to mobile species, ARs in Bali have demonstrated potential to significantly increase abundance and diversity. The ARs of Jemeluk Bay, North East Bali, displayed a 3.2x increase in number of fish species and a 25.6x increase in fish abundance 10 years after deployment (Puspasari et al. 2020). Similarly, Syam et al. (2017) showed that the ARs of Lebah, North East Bali, attracted 267 fish species over a 10 year period.

Table 2. 2: Factors which impact diversity and abundance of benthic and mobile species on artificial reefs, based on both Indonesian and international literature.

Factor	Impacting diversity and abundance of:	Explanation	'Best practice' suggestion for creating the artificial reef
Angle of the substrata	Benthic species	Perkol-Finkel et al. (2006) discussed how coral recruitment on AR is usually higher on vertical or inclined surfaces. This is because horizontal ARs are thought to have higher sedimentation levels (as sand can more easily settle on a flat surface), making it more difficult for coral larvae to attach themselves (Clark and Edwards 1999).	Create ARs with vertical or inclined surfaces (Perkol-Finkel et al. 2006).
Structural complexity	Benthic species Mobile species	Coral larvae more successfully settle on complex AR surfaces that are easier to grip and become attached to (Carleton and Sammarco 1987). ARs are more likely to attract mobile species if they are designed with structural features that mimic those of natural reefs (Komyakova et al. 2019). These structural features commonly include hiding spaces, more than one exit, shadow against light, high surface area and hollow interior spaces (Baine 2001). ARs that are created with these features will increase colonisation of juvenile fish (that require protective space), as well as attracting spawning adults (that require a textured surface to lay eggs) (Perkol-Finkel et al. 2006, Herbert et al. 2017).	Create ARs with high structural complexity (Perkol-Finkel et al. 2006, Herbert et al. 2017).

Composition of AR	Benthic species	The composition of surface substrata, in terms of chemistry and toxicity is thought to affect coral settlement (Baine 2001). For example, the use of rubber tyres as ARs has been associated with the leaching of heavy metals, which are toxic to benthic invertebrates (Collins et al. 2002).	Create ARs with nontoxic materials. Concrete is generally the most favoured building material (Baine 2001).
Age of AR	Benthic species Mobile species	Coral cover on ARs will increase as the corals grow over time (Wenker and Stevens 2020). ARs may take up to one century to mimic natural reefs in terms of coral cover (Perkol-Finkel et al. 2005). Unpublished data from The Indonesian Nature Foundation (as discussed by Puspasari et al. (2020)), showed that 15 month old ARs had four times higher coral recruitment than 7 month old ARs. These ARs were deployed in Buleleng, North Bali. Colonisation rate of mobile species onto artificial reefs is generally greatest within the first few months after deployment, and decreases with time (BaileyBrock 1989, Pickering and Whitmarsh 1997, Arney et al. 2017).	Allow time for colonisation of the AR. Regularly monitor the programmes ecological success and change methods/ objectives if necessary (Boström and Einarsson et al. 2020).
Location of AR	Benthic and mobile species	Shortly after development, it is expected that species from other reefs will colonise the AR (Koeck et al. 2011). Komyakova et al. (2019) suggests if the ARs are spatially isolated from other reefs, then they may be undetected by species looking for new habitats, and this will limit colonisation. This applies to benthic and mobile species at larval and adult life stages. Location is also an important factor, as the environmental conditions (e.g. wave action, temperature, depth and water quality) of a particular area may influence the ecological success of the AR (Baine 2001).	Create ARs that are close to natural reefs and/or built with corridors to allow species to move between reefs (Relini et al. 1994). Monitor the AR to ensure that deployment area is suitable (Baine 2001).

Fishing pressure	Mobile species	ARs are likely to reach their full potential (in terms of increasing diversity and abundance) when they are not subject to fishing pressure (Addis et al. 2016). Syam et al. (2017) described how most ARs within Bali are regularly fished, and target commercial species are therefore frequently missing. This can be problematic in situations where functional species are missing, such as marine mesopredators like black tip reef sharks (*Carcharhinus limbatus;* Muller and Hënle, 1839), which are thought to be overfished across Indonesia (Sembiring et al. 2015). The loss of sharks on a coral reef has been shown to alter the food chain below it, leading to potential declines in populations of herbivorous fish (Ruppert et al. 2013). Herbivorous fish are fundamental to the dynamics of reef communities as they reduce algal cover and provide corals more space to colonise benthic habitats (Estes et al. 2011), and the loss of these species on an AR may result in changes to ecosystem function and processes.	If possible, deploy the ARs within an MPA (Addis et al. 2016).

Artificial reefs and tourism related socio-economic benefits

ARs provide experiences to non-consumptive recreational marine users, such as divers, anglers and snorkellers (Stolk et al. 2005). AR marine tourism is thought to have multiple benefits, as discussed by Stolk et al. (2007):

1. Redistribution of tourists away from natural reefs. This can be help to reduce the threats associated with dive on coral reefs as discussed in section 1.5.
2. Highly valued experiences to tourists, which can be easier to access than natural reefs. ARs such as shipwrecks can provide exciting and unusual dive experiences
3. Generation of revenues which can be used to employ communities and further develop ecologically beneficial programmes.

Bali has multiple AR dive tourism sites. Pemutaran, in North West Bali, hosts an AR which is one of Bali's most popular dive sites (Trialfhianty 2017). This is a Biorock^{TM} AR programme, which has led to Pemutaran becoming a highly popular dive tourism site and its communities have successfully used tourism income to develop multiple coral reef restoration programmes (Trialfhianty 2017). The effectiveness of this AR in attracting dive tourism was highlighted Budisetyorini and Cahyani (2016) who showed that approximately 70% of tourists primarily visit Pemutaran to see the AR structures.

Decentralisation policy and NGOs

The Indonesian Decentralisation Policy (Act No.33), which established in 2004, gave greater authority, political power, and financial resources directly to local regencies and municipalities (Soejoto et al. 2015) and promoted the role of NGOs in pursuing conservation objectives (Atmodjo et al. 2020). The Decentralisation Policy has enabled and encouraged support from NGOs following the identification of an emerging threat to marine diversity within Indonesia. For example, Raja Ampat, the global epicentre for coral reef biodiversity, underwent vast developments in fisheries and oil/gas extraction in the early 2000s, posing substantial threats to internationally protected marine species including sea turtles and cetaceans (Mangubhai et al. 2012). This, alongside the newly implemented Decentralisation Policy, prompted conservation efforts from international NGOs like 'Conservation International' and 'The Nature Conservancy'. A large proportion of NGO effort in Bali is focused on the plastic pollution problem. Examples of this includes 'EcoBali', which offers plastic collection services and 'BYEBYEPLASTIC' which organises beach cleans and works with communities and the government to reduce the production and consumption of plastic products (Brooijmans et al. 2019).

Factors that influence community engagement

Tightly knit fisher communities are a common feature of coastal villages in Bali because fisher groups frequently gather for the planning and implementation of regular community events and religious ceremonies (Ginaya 2018). The involvement of these community groups is important for the success of a sustainable coral reef management programme in Bali (Suadi 2009). There are multiple factors that contribute towards community participation in a marine conservation programme. These are described in table 2.3.

Table 2. 3: Factors which can influence community engagement in a conservation programme.

Factor	Explanation	Bali/ Indonesia example
Perceived personal benefit	The level of personal benefit is thought to be a substantial factor determining communities willingness to participate in a conservation programme (Berkes 2010). *Note:* It can be problematic if support for marine conservation is driven purely by financial gain, because motivation to continue supporting the programme's objectives may reduce with a decrease in financial gain (Stem et al. 2003).	Berkes (2010) conducted interviews with fishers that participate in the 'Yayasan Alam Indonesia Lestari' (LINI) coral reef conservation NGO, based in Les Village, North Bali. From the interviews, it was clear that fishers efforts were not merely for conservation, but largely for an improvement of their livelihoods, such as increased fishing yields.
Education	Educational programmes that increase local people's knowledge of sustainable resource management have been shown to increase community participation in marine conservation projects. This is discussed by Hines et al. (1987), who highlighted that outreach programmes like these can lead to an increased individual sense of responsibility in taking care of their resources.	Leisher et al. (2012) demonstrated that educating local communities about the ecological and socio-economic benefits of marine protected areas (MPAs) within Raja Ampat, Indonesia, led to a substantial increase in community compliance and active participation. The study suggests that investments in MPA education and outreach is an effective tool to engage communities in conservation objectives.
Meeting community needs	Inclusion of the community in marine conservation decision-making determines how motivated they are to participate in the programme and/or contribute towards achieving its objectives (Lundquist and Granek 2005). When local fishers feel their livelihoods are not considered within the establishment of an MPA, they are likely to ignore regulations and fish illegally (Lundquist and Granek 2005),	Elliott et al. (2001) discussed this concept in terms of the Wakatobi Marine Park in Eastern Indonesia. In 1996, the marine park established zoning regulations and fishing restrictions which were criticised for not considering the livelihoods requirements of the local Sama-Bajo fishers. Glaser et al. (2010) discussed how MPAs in Indonesia have greater potential when they are developed and enforced by local people, with regulations that protect nature whilst considering the needs of the community.

	which will undermine the project's success (Campbell et al. 2012).	
Influence from local leaders	The success of marine resource management programmes often rely on support from influential local leaders(McLeod and Palmer 2015), which may 'bridge the gap' between local people and marine conservation objectives (Trialfhianty 2017).	Frey and Berkes (2014) concluded that local leaders, associated with the 'LINI' NGO in Les Village, North Bali, made great contributions towards encouraging local fishers to stop using cyanide. The study discussed how the widespread use of cyanide fishing bought the Les Village reef to the brink of collapse in 2006, but through the gradual phasing out of this technique, the reef was restored to relative health. It is now understood, that with the help of local leaders, fishers in Les Village have developed a sense of ownership over protecting their reef, and trust one another to not use cyanide. This case study provides a striking example of how community action, particularly with the help of local leaders, can generate positive environmental change on the reefs of North Bali.

Marine Ecotourism

Bali was visited by 3.5 million international tourists and 7.3 million domestic tourists in 2018 (Wardana et al. 2018). Despite the vast economic benefits tourism has bought to the Bali, it has been criticised for destroying the islands rich culture and high biodiversity (Tomomi 2010, Byczek 2011). For example, certain villages within South Bali were previously recognised for religious ceremonies and traditional music, however since the influx of tourism, these locations have been criticised for losing their cultural heritage and are now associated with westernised drinking and drug problems (Tomomi 2010). Tourism in these areas have been predominantly facilitated by large hotels and mega-resorts, which are owned by ex-patriots and provide limited benefits to local people. Mass tourism within Bali has also been associated with water pollution due to insufficient waste management, as well as water scarcity and loss of ecologically diverse and agriculturally productive land (Chong 2020).

Ecotourism can be defined as responsible travel to natural areas that conserves the environment and sustains the well-being of local people (Wall 1997). Global literature has highlighted that the influx of ecotourism can provide economic opportunities to areas with high unemployment (Garrod et al. 2003, Shani et al. 2012). For example, the coastal village of Kaikoura in New Zealand, was transformed from an economically depressed area to one with a successful ecotourism industry focused primarily around marine mammal tours (Orams 2002).

Volunteer tourism is also considered a form of ecotourism which involves individuals undertaking an organised holiday that includes some form work to help the destination's local community or restore its environment (Wearing 2001). Many international volunteer organisations exist that aim to facilitate this type of work, and an example includes the 'Marine Conservation Cambodia' project, which hosts international volunteers who assist in activities aiming to conserve endangered sea horses in the area of Koh Rong in Cambodia (Kitney et al. 2018).

Table 2. 4: Examples of ecotourism projects contributing to marine conservation in Bali.

Ecotourism Project Title	Conservation Issue	Conservation Activity	Overall Success:
'Turtle Conservation and Education Centre' (TCEC) in Serangan Island, South Bali.	-30,000 turtles poached per year around Serangan Island (Tomomi 2010). Poaching offers high incomes to local people as turtles are highly desired for Balinese Ceremonies (McLeod and Palmer 2015). -Development of tourist resorts lead to the loss of suitable turtle nesting sites (Tomomi 2010).	-Offering educational sessions encouraging the public not to consume turtle products (Tomomi 2010). -Providing live turtles for religious ceremonies without killing them (Tomomi 2010). -Donations collected from tourists which used for activities (and employment of local people) that protect sea turtles.	-Tomomi (2010) concluded that ecotourism is expected to be the most effective way to protect turtles around Serangan Island. However (at the time of the study) incomes raised from ecotourism were insufficient in providing alternative livelihoods for turtle poachers, and ecotourism was far from preventing the illegal turtle trade.

'North Bali Reef Conservation' (NBRC) in Karangasem, North Bali.	-Widespread coral reef destruction, mostly due to the anchoring of fishing boats (NBRC 2019). - Substantial marine plastic pollution issue due to overconsumption of single use plastics and lack of waste management infrastructure (NBRC 2019).	- Developing a volunteer-tourism programme, where international volunteers worked with local fishers to: - Build and deploy over 3000 artificial reef units. - Start a community plastic recycling centre -Run ongoing plastic awareness educational school programmes (NBRC 2019).	-After approximately 1 year, the artificial reefs were shown to have 5-6x higher fish biodiversity compared to a nearby control site (NBRC 2019). -Successful establishment of North Bali's first community run plastic recycling centre (NBRC 2019). There is currently no data on whether this has been effective in reducing marine plastic pollution.
Pemutaran Reef Restoration.	-Between 1980 – 2000, Pemutaran was thought to be one of the poorest villages in Bali. Its severe poverty contributed towards fishers use DFPs, which resulted in a widespread destruction of local reefs (Trialfhianty 2017).	-Using ecotourism to provide alternative jobs to (DFP) fishers, who were consequently able to earn much higher incomes. -Establishing ecotourism allowed the development of marine conservation projects, including a turtle hatchling conservation (and adult rehabilitation) organisation (Suparno et al. 2019) and the Biorock ™ artificial reef programme (Hilbertz and Goreau 1996).	-Marine conservation projects shown to be an important driver of ecotourism in Pemutaran (Trialfhianty 2017). These projects are also dependant on ecotourism, as it often provides their main source of funding. -Pemutaran is an example highlighting how marine conservation and ecotourism can work together synergistically to improve ecosystems and livelihoods of in Balinese coastal communities.

Ecotourism has demonstrated potential as a marine conservation tool, as highlighted with examples in Bali in table 2.4. Ecotourism in Bali is thought to have started in the 1980s and has become increasingly favoured by tourists in recent years (Tomomi 2010). Astarini et al. (2019) demonstrates that the tourism market in Bali is moving towards sustainable ecotourism and suggests that Karangasem, one of Bali's poorest regions, is well suited to develop this industry. Some small scale eco-tourism projects have already been established in this region, such as the Tenganan ecotourism village, which puts particular emphasis on protecting its local highly biodiverse flora and fauna (Karmini 2020), or Jasri village, which received Indonesia's 2013 Village Tourism award for its work developing tourism in a manner that preserves local Hindu culture (Amerta 2017).

Assuming it is appropriately managed so that local ecology and culture are well preserved, ecotourism has great potential in bringing some of Bali's poorest regions out of poverty (Byczek 2011) whilst providing ecological benefits such as supporting conservation and restoration (Tomomi 2010, NBRC 2019). Research has shown that in terms of attracting environmentally minded consumers, it is important for Bali's ecotourism businesses to demonstrate genuinely sustainable practices, rather than just using the word 'eco' as a meaningless marketing tool (Mahyuni et al. 2020). In 2016, the Indonesian Ministry for Tourism established Regulation No. 14/2016, which demands that all ecotourism projects advertised as 'sustainable', should accommodate local community empowerment, cultural preservation, and environmental conservation (Sugiri and Mahyuni 2019). After a recent temporary suspension of tourism in Bali due to the covid-19 pandemic, it is hoped that upon reopening, Bali transitions into more, genuinely sustainable forms of tourism (Stafford and Choe 2020).

What else could be done to protect Bali's reefs?

So far, this review has highlighted some of the main tools used to restore and conserve Bali's coral reefs. Ben-Romdhane et al. (2020) discussed international best practices in terms of the effective management of coral reefs, using examples like Australia (GBR), Belize, Bermuda and the Cayman Islands. When comparing marine conservation between these 'best practice' example nations and Bali, it is clear that Bali has demonstrated some level of effective coral reef management. Examples of this may include the development of effective ecotourism projects, the construction of large scale habitat enhancement projects like artificial reefs and the establishment of co-management schemes that involve communities in marine conservation decision making processes. As discussed by Goreau and Hayes (2021), urgent action (through pro-active restoration and threat reduction) is needed to increase the resilience and uphold the world's coral reef ecosystems. In light of international coral conservation best practices (Ben-Romdhane et al. 2020), it is suggested that Bali reefs could be better protected by:

Increasing official designation of MPAs

The previously cited literature has highlighted that well enforced MPAs are an effective tool to restore and conserve coral reefs (Costello 2014, Edgar et al. 2014). The IUCN has recently called for the full protection of 30% of the world's oceans by 2030 as an international marine conservation target (Zhao et al. 2020). At the time of the last World Database of Protected Areas report (2018), seven of the world's countries had already designated 30% of their waters as MPAs (Germany, US, France and Australia, Belgium, Jordan and New Zealand, although these MPAs were still not fully protected (IUCN 2018). The same report highlighted that in 2018, Indonesia's total MPAs made up only 3% of their waters, and although there is no data on this in Bali, it is thought to also be a relatively low percentage. The establishment and enforcement of MPAs is associated with multiple economic and societal challenges, especially for developing nations (Sowman and Sunde 2018). It is however, encouraging that Indonesia has recently reached its target to declare 200,000 km^2 of its territorial waters as MPAs by 2020 (Suparno et al. 2019). It is hoped that the designation of more MPAs in Indonesia, will lead to declines in illegal activities, such as the use of DFPs and the fishing of internationally protected species like reef manta rays *(Mobula alfredi;* Krefft 1868*)*.

Additionally, as previously highlighted, unclear MPA boundaries in Bali has led to user non-compliance (Suparno at al. 2019). It is suggested that precise zoning and clear boundaries are used to mark out the Bali's MPAs. Within these boundaries should be sites of key importance, such as nursery grounds, fish aggregation sites and resilient habitats (e.g. reefs that survive bleaching), and a zonation method that is agreed and approved by the local community should be in place. This will increase user compliance and lead to a more effective MPA network.

Creating an MPA network

MPAs have been shown to be more effective if they are a connected network of protected areas (Daly et al. 2018). For example, a network of MPAs in Hawaii was shown to provide greater ecological and economic outcomes than the sum of outcomes of individual MPAs of the same size (Grorud-Colvert et al. 2014). One possible reason why an MPA network is more effective is because it may protect all core habitats of migratory marine species, thus more effectively conserving their populations (Daly et al. 2018). Another reason is due to the benefits arising from congruent transnational or transregional management, resulting in an overall more effective management of the MPA (Daly et al. 2018).

There are many unofficial community managed marine reserves within Bali's Buleleng and Karangasem regencies that are not recognised as government designated MPAs (Mustika and Ratha 2013). MPAs within Bali, would experience increased socio-economic and ecological benefits if they form a collective officially designated MPA network. This concept was first introduced in Bali by Mustika and Ratha (2013), who discussed that a network would lead to better ecologically connected MPAs, with more effective management. The authors also commented that the decline in populations of migratory megafauna (including turtles, sharks and marine mammals) in Bali is an urgent conservation issue, and that migratory routes and critical habitats of these species would be better protected if a large, highly connected MPA network is established. The proposed Bali MPA network is also expected to synchronise marine management decisions through enabling the exchange of knowledge and experience between regions (Berdej and Armitage 2018). It is thought that administrative separations between regencies have resulted in different marine management decisions and policies. The island of Bali is relatively small, so ecological marine systems are particularly connected, thus a synchronisation of marine management practices between regencies would foster more effective management (Berdej and Armitage 2018).

Substantially reducing plastic pollution

The previously cited literature has highlighted that Indonesia is the world's second largest plastic polluter (Shuker and Cadman 2018) and this greatly threatens its marine ecosystems (Turak and Devantier 2013, Giesler 2018, Brooijmans et al. 2019). This issue is especially prevalent in Bali, which declared a state of 'garbage emergency' in 2017 (Garcia et al. 2019). Currently, Indonesia has weak legal and institutional frameworks in place to manage its plastic pollution problem (Garcia et al. 2019). Countries such as Canada, which are moving towards becoming plastic waste free (Walton et al. 2018), may be seen as a 'best practice' nation in terms of waste management. It is beyond the scope of this study to evaluate how Bali can resolve its plastic 'emergency', although some suggestions, as discussed by Garcia et al (2019) may include:

- Developing national legal frameworks and local level regulations which work with plastic producers and consumers.
- Continuing to work with religious groups, NGOs, schools and other educational bodies which encourage communities to adopt more environmentally conscious practices, such as reducing single use plastics.

- Strengthening local waste management and recycling infrastructure.

Continuing artificial reef instalment, ensuring that 'best practice' recommendations are followed

The previous literature has highlighted that ARs have been used in Bali as a habitat enhancement tool to successfully restore marine biodiversity and abundance (Syam et al. 2017, Puspasari et al. 2020). ARs are continuing to be built for marine restoration in Bali (LINI 2021), and funding for these projects appears to be increasing over the coming years (Karunia 2021). In terms of achieving restoration objectives, it is important that programmes follow guidelines in light of 'best practice' for building ARs. Table 2.2 highlights some of the main factors which contribute towards the success of AR programmes, and makes 'best practice' suggestions for creating an AR.

Developing Bali as an ecotourism destination

The reviewed literature within table 2.4 has highlighted that ecotourism has contributed towards successful marine conservation in Bali (McLeod and Palmer 2015, Trialfhianty 2017, NBRC 2019). Developing ecotourism within Bali is suggested as a tool which can be used to bring some of Bali's poorest regions out of poverty whilst simultaneously contributing to environmental conservation (Byczek 2011, Astarini et al. 2019). As Bali's businesses develop this industry, it is important that they demonstrate genuinely sustainable practices (Mahyuni et al. 2020) that accommodate local community empowerment, cultural preservation, and environmental conservation (Sugiri and Mahyuni 2019).

Increasing engagement in global science to inform marine conservation decision-making

Scientific research is important in biodiversity conservation as it informs practical decision making and provides organisations with information and tools to achieve their objectives (Mair et al. 2018). Recent literature has highlighted multiple innovative methods which may be considered for coral conservation. These can include (but are not limited to) building nurseries with coral species that are resistant to bleaching (Camp et al. 2017, Morikawa and Palumbi 2019), most effective techniques for coral transplantation (Endo et al. 2008, Onaka et al. 2013), genetically modifying reef building corals (Cleves et al. 2018) and coral microbiome manipulation (Rosado et al. 2019). Furthermore, Boström-Einarsson et al. (2020) highlighted that scientific research can support coral reef conservation programmes with aspects such as developing time and spatial scales, designing restoration methods and running adequate monitoring programmes.

However, there is a known gap between scientific research and practical restoration, which is an issue persisting globally in coral reef conservation (Habel et al. 2013, Mills et al. 2020). This gap, which is often caused by the lack of communication between the scientific community and conservation managers, has led to some coral reef restoration programmes being undertaken with little scientific input, ineffective management, and ultimately resulting in the organisation not achieving its objectives (Boström-Einarsson et al. 2020). Research has highlighted that some reef conservation activities in Bali may have been unsuccessful in achieving their objectives. For example, the BBNP MPA was ineffective in enforcing fishing regulations, and consequently it resulted in limited ecological (Doherty et al. 2013) and socio-ecological benefits (Pedju 2018). In this example, a greater understanding of scientific research in MPA regulations and enforcement may have helped conservation managers to come up with solutions to stop illegal fishing. As Bali progresses with coral reef conservation, it is important that global science is used to inform conservation decision making process. It is suggested, that this engagement can be achieved through focus group discussions (FGDs) and integrated studies between researchers and marine conservation managers. Collaborations like this could lead to successful ongoing monitoring programmes, as well as more effective decision making.

Developing more marine monitoring programmes

Marine monitoring programmes are generally recognised as important because they help scientists and conservation managers to characterise and understand coastal dynamics and vulnerabilities (Bastos et al. 2016). Marine monitoring enables environmental stressors to be identified, and in some cases, reduced or removed (Nõges et al. 2016) and is said to be “urgently required” for the protection of global marine biodiversity and ecosystem functioning (Danovaro et al. 2016). Bali’s coral reefs have experienced an increase in active management measures over the last few decades (for example AR deployment, ecotourism activities and MPA establishment). It is important that these programs are monitored (in terms of the improvement of ecological conditions), so that future management decisions are informed on what is and isn’t successful.

The previously highlighted literature has highlighted that water quality, especially nutrient pollution, can impact coral health (Szmant 2002, D’Angelo and Wiedenmann 2014). It appears that marine monitoring programmes are relatively limited, especially with regards to water quality and it been suggested that Indonesia should develop more water quality monitoring programmes (E.E. Ampou, Personal Communication). More specifically, this should be conducted during the transitional months between Indonesia's dry and wet seasons (March/ April and September/October) to best represent average water quality. More monitoring programmes like this will be useful to understand the link between water quality and coral reef degradation in Bali, which could lead to the development of water quality control measures.

Conclusion

Anthropogenic activities have caused a worldwide long-term decline in coral reef biodiversity, abundance and habitat structure (Pandolfi et al. 2011, Hughes et al. 2018). The greatest threat to the world's reefs is coral bleaching, due to ocean warming and acidification (Heron et al. 2017), which is a consequence of increased atmospheric greenhouse gas emissions (Pörtner et al. 2014). This review has assessed coral reef threats in Bali and wider Indonesia and has highlighted that 86% of Indonesia's coral reefs face medium or high levels of threat (Burke et al. 2012). Within Bali, Wickasana (2020) has shown 50% of its corals are in good health, whilst 20% are declining and 30% are poor to multiple threats as highlighted by part 1. Coral bleaching is present on the reefs around Bali (Ampou and Tito 2019; Karim 2019; Suparno et al. 2019), but the extent of the bleaching may be less severe than other reefs around the world, especially when compared to reefs like the GBR (Lewis and Mallela 2018).

Alongside a global mitigation of climate change through aggressive and large reductions in greenhouse gas emissions (Ben-Romdhane et al. 2020), multiple small/medium scale tools may be considered to conserve, restore and increase the resilience of coral reefs. Marine conservation appears to have first started in Bali in the 1970s (Polunin et al. 1983), likely as a result of a widespread decline of the island's coral reef health. Some primary conservation tools used across Bali so far have included MPA establishment, ecotourism development and artificial reef deployment. Engagement of the local community has been shown to be important for the success of marine conservation programme (Suadi 2009), and this is often influenced by other factors.

This review has compared marine conservation in Bali to international 'best practices'. Marine conservation projects in Bali will likely gain further momentum in coming years, especially with Bali's 111.2 billion IDR support fund for coral reef restoration in 2021 - 2022 (Karunia 2021). It has made suggestions on how marine conservation in Bali can improve by following international best practices.

These include:

1. Increasing its designation of official MPAs and strengthening management of existing ones
2. Creating an MPA network
3. Substantially reducing marine plastic pollution
4. Continuing artificial reef construction, ensuring that it follows best practices
5. Developing Bali as an ecotourism destination
6. Increasing engagement in global science to inform marine conservation decision-making
7. Developing more marine monitoring programmes

Most of the literature used to review this topic has been scientific papers. Some of these papers have been written by international scientists and are published in highly regarded journals. In contrast, some of the reviewed papers were small-scale local studies, many were written in Indonesian and were not published in a journal. It is important to have include these small-scale studies within this literature review, as many provide the most thorough insight into the current situation in Bali.

Acknowledgements

I would like to thank Rick Stafford, Alice Hall and Georgia Jones from Bournemouth University, who provided support with writing, proof reading and theoretical ideas. I would also like to thank my Indonesian research team, Eghbert Elvan Ampou, I. Gusti Ngurah Agung Suryaputra, Luh Putu Mahyuni and Rahmadi Prasetjo, who offered invaluable advice with regards to local research and general support.

Permits

A research permit was obtained from Indonesia’s Ministry of Research (BRIN).
Research permit number: 34/TU.B5.4/SIP/VII/2023.

Conflicts of Interest

The authors declare no conflicts of interest.

References

Abidin, S. Z. Z. and Mohamed, B., 2014. A review of SCUBA diving impacts and implication for coral reefs conservation and tourism management. *EDP Sciences*, 12, 1093.

Adam, T. C., Burkepile, D. E., Holbrook, S. J., Carpenter, R. C., Claudet, J., Loiseau, C., Thiault, L., Brooks, A. J., Washburn, L., and Schmitt, R. J., 2021. Landscape-scale patterns of nutrient enrichment in a coral reef ecosystem: implications for coral to algae phase shifts. *Ecological Applications*, 31 (1), e2227.

Addis, D. T., Patterson Iii, W. F., and Dance, M. A., 2016. The potential for unreported artificial reefs to serve as refuges from fishing mortality for reef fishes. *North American Journal of Fisheries Management*, 36 (1), 131–139.

Aeby, G. S., Howells, E., Work, T., Abrego, D., Williams, G. J., Wedding, L. M., Caldwell, J. M., Moritsch, M., and Burt, J. A., 2020. Localized outbreaks of coral disease on Arabian reefs are linked to extreme temperatures and environmental stressors. *Coral Reefs*, 39 (3), 829–846.

Al-Horani, F. A., Hamdi, M., and Al-Rousan, S. A., 2011. Prey Selection and Feeding Rates of Drupella cornus (Gastropoda: Muricidae) on Corals from the Jordanian Coast of the Gulf of Aqaba, Red Sea. Jordan Journal of Biological Sciences, 4 (4), 1-8

Allen, G. R., 2008. Conservation hotspots of biodiversity and endemism for Indo-Pacific coral reef fishes. Aquatic conservation marine and freshwater ecosystems, 18 (5), 541-556. Allen, G. R. and Erdmann, M. V, 2013. Reef Fishes of Bali, Indonesia. In: Bali Marine Rapid Assessment Program 2011. 1, 15.

Amerta, I. M. S., 2017. The role of tourism stakeholders at Jasri tourism village development, Karangasem regency. *International Journal of Social Sciences and Humanities (IJSSH)*, 1 (2), 20–28.

Ampou, E. E, Tito, C., 2019. Begini Kondisi Kesehatan Terumbu Karang di Perairan Nusa Penida.

Ampou, 2020. *PEMUTIHAN KARANG DI PERAIRAN BALI UTARA DAN SEKITARNYA (PERIODE APRIL - MEI 2020*), [online]. Denpasar: SINTA.

Ampou, E. E., Manessa, M. D. M., Hamzah, F., and Widagti, N., 2020. Study of Sea Surface Temperature (SST), Does It Affect Coral Reefs? Jurnal Ilmiah Perikanan dan Kelautan, 12 (2), 199–213.

Antara, M. and Sumarniasih, M. S., 2017. Role of tourism in economy of Bali and Indonesia. *Journal of Tourism and Hospitality Management*, 5 (2), 34–44.

Arney, R. N., Froehlich, C. Y. M., and Kline, R. J., 2017. Recruitment patterns of juvenile fish at an artificial reef area in the Gulf of Mexico. *Marine and Coastal Fisheries*, 9 (1), 79–92.

Aronson, R. B. and Precht, W. F., 2006. Conservation, precaution, and Caribbean reefs. *Coral Reefs*, 25 (3), 441–450.

Astarini, I. A., Heriyatmo, A. N., Mahardikayanti, N. K. Y., Susanti, M. E., Utari, I. A. M. D., Sukmadewi, N. W. R., Dewi, N. M. W. S., Endrayanti, N. W. N., Lestari, W. P., and Dewi, N. K. A., 2019. Promoting Ecotourism Destination at Jungutan Village, Karangasem, Bali. Journal of Tourism, Hospitality and Sports, 41, 2312–5187,

Atmodjo, E., Lamers, M., and Mol, A. P. J., 2020. Governing Dynamics in Marine Conservation Tourism in Raja Ampat, Indonesia. *Tourism Planning & Development*, 17 (6), 655–673.

Babcock, R. C., Shears, N. T., Alcala, A. C., Barrett, N. S., Edgar, G. J., Lafferty, K. D., McClanahan, T. R., and Russ, G. R., 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. *Proceedings of the National Academy of Sciences*, 107 (43), 18256–18261.

Bacalso, R. T. M. and Wolff, M., 2014. Trophic flow structure of the Danajon ecosystem (Central Philippines) and impacts of illegal and destructive fishing practices. *Journal of Marine Systems*, 139, 103–118.

Bailey-Brock, J. H., 1989. Fouling community development on an artificial reef in Hawaiian waters. *Bulletin of Marine Science*, 44 (2), 580–591.

Baine, M., 2001. Artificial reefs: a review of their design, application, management and performance. *Ocean & coastal management*, 44 (3–4), 241–259.

Baria-Rodriguez, M. V., dela Cruz, D. W., Dizon, R. M., Yap, H. T., and Villanueva, R. D., 2019. Performance and cost-effectiveness of sexually produced Acropora granulosa juveniles compared with asexually generated coral fragments in restoring degraded reef areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 29 (6), 891–900.

Barnes, D. K. A., Galgani, F., Thompson, R. C., and Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 364 (1526), 1985–1998.

Barton, J. A., Willis, B. L., and Hutson, K. S., 2017. Coral propagation: a review of techniques for ornamental trade and reef restoration. *Reviews in Aquaculture*, 9 (3), 238–256.

Bastos, L., Bio, A., and Iglesias, I., 2016. The importance of marine observatories and of RAIA in particular. *Frontiers in Marine Science*, 3, 140.

Baum, G., Januar, H. I., Ferse, S. C. A., and Kunzmann, A., 2015. Local and regional impacts of pollution on coral reefs along the Thousand Islands north of the megacity Jakarta, Indonesia. *PloS one*, 10 (9), e0138271.

Baum, G., Januar, I., Ferse, S. C. A., Wild, C., and Kunzmann, A., 2016. Abundance and physiology of dominant soft corals linked to water quality in Jakarta Bay, Indonesia. *PeerJ*, 4, e2625.

Becker, A., Taylor, M. D., and Lowry, M. B., 2017. Monitoring of reef associated and pelagic fish communities on Australia's first purpose built offshore artificial reef. *ICES Journal of Marine Science*, 74 (1), 277–285.

Bell, D. E., 1992. The 1992 convention on biological diversity: the continuing significance of US objections at the Earth Summit. *Geo. Wash. J. Int'l L. & Econ.*, 26, 479.

Ben-Romdhane, H., Jabado, R. W., Grandcourt, E. M., Perry, R. J. O., Al Blooshi, A. Y., Marpu, P. R., Ouarda, T. B. M. J., and Ghedira, H., 2020. Coral reefs of Abu Dhabi, United Arab Emirates: Analysis of management approaches in light of international best practices and a changing climate. *Frontiers in Marine Science*, 7, 541.

Berdej, S. and Armitage, D., 2018. Bridging for Better Conservation Fit in Indonesia's Coastal-Marine Systems. Front. *Fishing for Human Perceptions in Coastal and Island Marine Resource Use Systems. Lausanne: Frontiers Media. doi: 10.3389*, 4, 30.

Berkes, F., 2010. Linkages and multilevel systems for matching governance and ecology: lessons from roving bandits. *Bulletin of Marine Science*, 86 (2), 235–250.

Bideci, C. and Cater, C., 2019. In search of underwater atmosphere: A new diving world on artificial reefs. *Atmospheric turn in culture and tourism: Place, design and process impacts on customer behaviour, marketing and branding,* [online]. Bradford: Emerald Publishing Limited.

Boekschoten, G. J., Best, M. B., and Putra, K. S., 2000. Balinese reefs in historical context. *Citeseer,* 23–27.

Bohnsack, J. A., 1993. Marine reserves: they enhance fisheries, reduce conflicts, and protect resources. *Oceanus*, 36 (3), 63–72.

Bohnsack, J. A. and Sutherland, D. L., 1985. Artificial reef research: a review with recommendations for future priorities. *Bulletin of Marine Science*, 37 (1), 11–39.

Boström-Einarsson, L., Babcock, R. C., Bayraktarov, E., Ceccarelli, D., Cook, N., Ferse, S. C. A., Hancock, B., Harrison, P., Hein, M., and Shaver, E., 2020. Coral restoration–A systematic review of current methods, successes, failures and future directions. *PloS one*, 15 (1), e0226631.

Bottema, M. J. M. and Bush, S. R., 2012. The durability of private sector-led marine conservation: A case study of two entrepreneurial marine protected areas in Indonesia. *Ocean & coastal management*, 61, 38–48.

De Brauwer, M., Harvey, E. S., McIlwain, J. L., Hobbs, J.-P. A., Jompa, J., and Burton, M., 2017. The economic contribution of the muck dive industry to tourism in Southeast Asia. *Marine Policy*, 83, 92–99.

Brickhill, M. J., Lee, S. Y., and Connolly, R. M., 2005. Fishes associated with artificial reefs: attributing changes to attraction or production using novel approaches. *Journal of Fish Biology*, 67, 53–71.

Briggs, J. C., 2005. The marine East Indies: diversity and speciation. *Journal of Biogeography*, 32 (9), 1517– 1522.

Brodie, J., Fabricius, K., De'ath, G., and Okaji, K., 2005. Are increased nutrient inputs responsible for more outbreaks of crown-of-thorns starfish? An appraisal of the evidence. *Marine pollution bulletin*, 51 (1–4), 266–278.

Brooijmans, S., Franken, M., de Iongh, Z., de Jong, W., and van Marsbergen, A., 2019. *Plastic Discharge in Bali's Rivers,* [online]. Delft: Delft University of Technology.

Bruno, J. F., Petes, L. E., Drew Harvell, C., and Hettinger, A., 2003. Nutrient enrichment can increase the severity of coral diseases. *Ecology letters*, 6 (12), 1056–1061.

Bruno, J. F., Selig, E. R., Casey, K. S., Page, C. A., Willis, B. L., Harvell, C. D., Sweatman, H., and Melendy, A. M., 2007. Thermal stress and coral cover as drivers of coral disease outbreaks. *PLoS Biol*, 5 (6), e124.

Budisetyorini, B. and Endah Cahyani, S., 2016. *THE EFFECT OF BIOROCK CORAL REEF RESTORATION ON TOURIST TRAVEL DECISIONS AT PEMUTERAN BAY, BALI,* [online]. Badung: STPB Institutional Repository.

Burke, L., Reytar, K., Spalding, M., and Perry, A., 2012. *Reefs at risk revisited in the Coral Triangle,* [online]. Washington: World Resource Institute.

Byczek, C., 2011. Blessings for All? Community-Based Ecotourism in Bali Between Global, National, and Local Interests–A Case Study. *Austrian Journal of South-East Asian Studies*, 4 (1), 81–106.

Camp, E. F., Nitschke, M. R., Rodolfo-Metalpa, R., Houlbreque, F., Gardner, S. G., Smith, D. J., Zampighi, M., and Suggett, D. J., 2017. Reef-building corals thrive within hot-acidified and deoxygenated waters. *Scientific reports*, 7 (1), 1–9.

Campbell, S. J., Hoey, A. S., Maynard, J., Kartawijaya, T., Cinner, J., Graham, N. A. J., and Baird, A. H., 2012. Weak compliance undermines the success of no-take zones in a large government-controlled marine protected area. *PloS one*, 7 (11), e50074.

Carleton, J. H. and Sammarco, P. W., 1987. Effects of substratum irregularity on success of coral settlement: quantification by comparative geomorphological techniques. *Bulletin of Marine Science*, 40 (1), 85–98.

Cesar, H., 1996. *Economic analysis of Indonesian coral reefs.* Washington: World Bank. Report 20455.

Chong, K. L., 2020. The side effects of mass tourism: the voices of Bali islanders. *Asia Pacific Journal of Tourism Research*, 25 (2), 157–169.

Clark, M. A., Domingo, N. G. G., Colgan, K., Thakrar, S. K., Tilman, D., Lynch, J., Azevedo, I. L., and Hill, J. D., 2020. Global food system emissions could preclude achieving the 1.5° and 2° C climate change targets. *science*, 370 (6517), 705–708.

Clark, S. and Edwards, A. J., 1999. An evaluation of artificial reef structures as tools for marine habitat rehabilitation in the Maldives. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 9 (1), 5–21.

Cleves, P. A., Strader, M. E., Bay, L. K., Pringle, J. R., and Matz, M. V, 2018. CRISPR/Cas9-mediated genome editing in a reef-building coral. *Proceedings of the National Academy of Sciences*, 115 (20), 5235–5240.

Clukey, K. E., Lepczyk, C. A., Balazs, G. H., Work, T. M., Li, Q. X., Bachman, M. J., and Lynch, J. M., 2018. Persistent organic pollutants in fat of three species of Pacific pelagic longline caught sea turtles: Accumulation in relation to ingested plastic marine debris. *Science of the Total Environment*, 610, 402– 411.

Collins, K. J., Jensen, A. C., Mallinson, J. J., Roenelle, V., and Smith, I. P., 2002. Environmental impact assessment of a scrap tyre artificial reef. *ICES Journal of Marine Science*, 59 (suppl), S243–S249.

Costanza, R., De Groot, R., Sutton, P., Van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S., and Turner, R. K., 2014. Changes in the global value of ecosystem services. *Global environmental change*, 26, 152–158.

Costello, M. J., 2014. Long live Marine Reserves: A review of experiences and benefits. *Biological Conservation*, 176, 289–296.

Crippa, M., Guizzardi, D., Muntean, M., Schaaf, E., Solazzo, E., Monforti-Ferrario, F., Olivier, J. G. J., and Vignati, E., 2020. *Fossil CO 2 emissions of all world countries–2020 report*, [online]. Luxembourg: European Commission. Report 30358.

D'Angelo, C. and Wiedenmann, J., 2014. Impacts of nutrient enrichment on coral reefs: new perspectives and implications for coastal management and reef survival. *Current Opinion in Environmental Sustainability*, 7, 82–93.

Daly, R., Smale, M. J., Singh, S., Anders, D., Shivji, M., K. Daly, C. A., Lea, J. S. E., Sousa, L. L., Wetherbee, B. M., and Fitzpatrick, R., 2018. Refuges and risks: Evaluating the benefits of an expanded MPA network for mobile apex predators. *Diversity and Distributions*, 24 (9), 1217–1230.

Danovaro, R., Carugati, L., Berzano, M., Cahill, A. E., Carvalho, S., Chenuil, A., Corinaldesi, C., Cristina, S., David, R., and Dell'Anno, A., 2016. Implementing and innovating marine monitoring approaches for assessing marine environmental status. *Frontiers in Marine Science*, 3, 213.

Daulat, A., Pranowo, W. S., and Amri, S. N., 2019. MANGROVE FOREST CHANGE IN NUSA PENIDA MARINE PROTECTED AREA, BALI-INDONESIA USING LANDSAT SATELLITE IMAGERY. *International Journal of Remote Sensing and Earth Sciences (IJReSES)*, 15 (2), 141–156.

Davenport, J. and Davenport, J. L., 2006. The impact of tourism and personal leisure transport on coastal environments: a review. *Estuarine, coastal and shelf science*, 67 (1–2), 280–292.

Deaker, D. J., Agüera, A., Lin, H.-A., Lawson, C., Budden, C., Dworjanyn, S. A., Mos, B., and Byrne, M., 2020. The hidden army: corallivorous crown-of-thorns seastars can spend years as herbivorous juveniles. *Biology letters*, 16 (4), 20190849.

Doherty, O., Milner, C., Dustan, P., Campbell, S., Pardede, S., Kartawijaya, T., and Alling, A., 2013. Report on Menjangan Island's coral reef: A Bali Barat National Park marine protected area. *Atoll Research Bulletin*, 19 (599), 1–18.

Dwiki, S., 2018. Development of environmental policy in Indonesia regarding mining industry in comparison with the United States and Australia: The lesson that can be learned. *Evergreen*, 5 (2), 50–57.

Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Minx, J. C., Farahani, E., Susanne, K., Seyboth, K., Adler, A., Baum, I., and Brunner, S., 2014. *AR5 climate change 2014: Mitigation of climate change,* [online]. New York: IPCC.

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., Barrett, N. S., Becerro, M. A., Bernard, A. T. F., and Berkhout, J., 2014. Global conservation outcomes depend on marine protected areas with five key features. *Nature*, 506 (7487), 216–220.

Elliott, G., Mitchell, B., Wiltshire, B., Manan, I. A., and Wismer, S., 2001. Community participation in marine protected area management: Wakatobi National Park, Sulawesi, Indonesia. *Coastal Management*, 29 (4), 295–316.

Endo, S., Prasetyo, R., Nishihira, M., and Onaka, S., 2008. Experimental coral transplantation in the moat of Kuta beach, Bali Island, Indonesia. *PROCEEDINGS OF COASTAL ENGINEERING, JSCE*. Japan Society of Civil Engineers, 1121–1125.

Erdmann, M., Pet-Soede, C., and Cabanban, A., 2000. *Destructive fishing practice*, [online]. Bali: International Coral Reef Symposium.

Estes, J. A., Terborgh, J., Brashares, J. S., Power, M. E., Berger, J., Bond, W. J., Carpenter, S. R., Essington, T. E., Holt, R. D., and Jackson, J. B. C., 2011. Trophic downgrading of planet Earth. *science*, 333 (6040), 301–306.

Fabi, G. and Spagnolo, A., 2011. *Artificial reefs in the management of Mediterranean Sea fisheries,* [online]. Florida: CRC Press.

Fabricius, K. E., Okaji, K., and De'Ath, G., 2010. Three lines of evidence to link outbreaks of the crown-ofthorns seastar Acanthaster planci to the release of larval food limitation. *Coral Reefs*, 29 (3), 593–605.

Faizal, A., Amri, K., Rani, C., Nessa, M. N., and Jompa, J., 2020. Dynamic model; the effects of eutrophication and sedimentation on the degradation of Coral Reefs in Spermonde Archipelago, Indonesia. *Earth and Environmental Science*. 12084.

Ferrier-Pagès, C., Gattuso, J.-P., Dallot, S., and Jaubert, J., 2000. Effect of nutrient enrichment on growth and photosynthesis of the zooxanthellate coral Stylophora pistillata. *Coral Reefs*, 19 (2), 103–113.

Fisher, R., O'Leary, R. A., Low-Choy, S., Mengersen, K., Knowlton, N., Brainard, R. E., and Caley, M. J., 2015. Species richness on coral reefs and the pursuit of convergent global estimates. *Current Biology*, 25 (4), 500–505.

Foden, W. B., Butchart, S. H. M., Stuart, S. N., Vié, J.-C., Akçakaya, H. R., Angulo, A., DeVantier, L. M., Gutsche, A., Turak, E., and Cao, L., 2013. Identifying the world's most climate change vulnerable species: a systematic trait-based assessment of all birds, amphibians and corals. *PloS one*, 8 (6), e65427.

Frey, J., 2013. *A community-based approach to sustainable ornamental fishing on coral reefs, Bali, Indonesia,* [online]. Manitoba: University of Manitoba

Frey, J. and Berkes, F., 2014. Can partnerships and community-based conservation reverse the decline of coral reef social-ecological systems? *International Journal of the Commons*, 8 (1).

Garcia, B., Fang, M. M., and Lin, J., 2019. Marine plastic pollution in Asia: All hands on deck! *Chinese Journal of Environmental Law*, 3 (1), 11–46.

Garrod, B., Wilson, J. C., and Bruce, D. M., 2003. Defining marine ecotourism: a Delphi study. *Marine Ecotourism: Issues and Experiences. Channel View, Clevedon, Australia*, 17–36.

GBRMPA, 2018. *From blueprint to action: Great Barrier Reef blueprint for resilience: progress report,* [online]. Queensland: GBRMPA.

Germanov, E. S., Marshall, A. D., Bejder, L., Fossi, M. C., and Loneragan, N. R., 2018. Microplastics: no small problem for filter-feeding megafauna. *Trends in ecology & evolution*, 33 (4), 227–232.

Gerungan, A. and Chia, K. W., 2020. Scuba diving operators' perspective of scuba diving tourism business in Nusa Penida, Indonesia. *Journal of Outdoor Recreation and Tourism*, 31, 100328.

Giesler, K., 2018. *The Plastic Problem: Plastic Pollution in Bali,* [online]. Vermont: SIT.

Ginaya, G., 2018. The Balinese calendar system: From its epistemological perspective to axiological practices. *International journal of linguistics, literature and culture*, 4 (3), 24–37.

Glaeser, B. and Glaser, M., 2010. Global change and coastal threats: The Indonesian case. An attempt in multilevel social-ecological research. *Human Ecology Review*, 135–147.

Glaser, M., Baitoningsih, W., Ferse, S. C. A., Neil, M., and Deswandi, R., 2010. Whose sustainability? Top– down participation and emergent rules in marine protected area management in Indonesia. *Marine Policy*, 34 (6), 1215–1225.

Goreau, T. J. F. and Hayes, R. L., 2021. Global warming triggers coral reef bleaching tipping point. *Ambio*, 1–4.

Grorud-Colvert, K., Claudet, J., Tissot, B. N., Caselle, J. E., Carr, M. H., Day, J. C., Friedlander, A. M., Lester, S. E., De Loma, T. L., and Malone, D., 2014. Marine protected area networks: assessing whether the whole is greater than the sum of its parts. *PloS one*, 9 (8), e102298.

Grottoli, A. G., Warner, M. E., Levas, S. J., Aschaffenburg, M. D., Schoepf, V., McGinley, M., Baumann, J., and Matsui, Y., 2014. The cumulative impact of annual coral bleaching can turn some coral species winners into losers. *Global Change Biology*, 20 (12), 3823–3833.

Habel, J. C., Gossner, M. M., Meyer, S. T., Eggermont, H., Lens, L., Dengler, J., and Weisser, W. W., 2013. Mind the gaps when using science to address conservation concerns. *Biodiversity and Conservation*, 22 (10), 2413–2427.

Haddock-Fraser, J. and Hampton, M. P., 2012. Multistakeholder values on the sustainability of dive tourism: case studies of Sipadan and Perhentian Islands, Malaysia. *Tourism Analysis*, 17 (1), 27–41.

Halim, A., 2002. Adoption of cyanide fishing practice in Indonesia. *Ocean & coastal management*, 45 (4–5), 313–323.

Hani, M. S., Jompa, J., Nessa, M. N., and White, A. T., 2019. Manta ray watching tourism in Eastern Indonesia: Is it sustainable? *Earth and Environmental Science*, 253 (1), 12042.

Harris, L. E., 2009. Artificial reefs for ecosystem restoration and coastal erosion protection with aquaculture and recreational amenities. *Reef Journal*, 1 (1), 235–246.

Hartati, S. T., 2017. Rehabilitasi wilayah pesisir melalui pengembangan terumbu buatan. *Bawal Widya Riset Perikanan Tangkap*, 2 (1), 35–43.

Herbert, R. J. H., Collins, K., Mallinson, J., Hall, A. E., Pegg, J., Ross, K., Clarke, L., and Clements, T., 2017. Epibenthic and mobile species colonisation of a geotextile artificial surf reef on the south coast of England. *PloS one*, 12 (9), e0184100.

Heron, S. F., Eakin, C. M., Douvere, F., Anderson, K. L., Day, J. C., Geiger, E., Hoegh-Guldberg, O., Van Hooidonk, R., Hughes, T., and Marshall, P., 2017. Impacts of climate change on World Heritage coral reefs: a first global scientific assessment.

Hilbertz, W. H. and Goreau, T. J., 1996. *Method of enhancing the growth of aquatic organisms, and structures created thereby*, [online]. (United States Patent).

Hines, J. M., Hungerford, H. R., and Tomera, A. N., 1987. Analysis and synthesis of research on responsible environmental behavior: A meta-analysis. *The Journal of environmental education*, 18 (2), 1–8.

Hobbs, J.-P. A. and Frisch, A. J., 2010. Coral disease in the Indian Ocean: taxonomic susceptibility, spatial distribution and the role of host density on the prevalence of white syndrome. *Diseases of Aquatic Organisms*, 89 (1), 1–8.

Hoegh-Guldberg, O., Poloczanska, E. S., Skirving, W., and Dove, S., 2017. Coral reef ecosystems under climate change and ocean acidification. *Frontiers in Marine Science*, 4, 158.

Hughes, T. P., Kerry, J. T., Álvarez-Noriega, M., Álvarez-Romero, J. G., Anderson, K. D., Baird, A. H., Babcock, R. C., Beger, M., Bellwood, D. R., and Berkelmans, R., 2017. Global warming and recurrent mass bleaching of corals. Nature, 543 (7645), 373–377.

Hughes, T. P., Kerry, J. T., Baird, A. H., Connolly, S. R., Dietzel, A., Eakin, C. M., Heron, S. F., Hoey, A. S., Hoogenboom, M. O., and Liu, G., 2018. Global warming transforms coral reef assemblages. Nature, 556 (7702), 492–496.

Israel, D., Gallo, C., and Angel, D. L., 2017. Benthic artificial reefs as a means to reduce the environmental effects of cod mariculture in Skutulsfjörður, Iceland. Marine Biodiversity, 47 (2), 405–411.

IUCN, 2018. *World Database on Protected Areas*, [online]. Gland: IUCN.

Jambeck, J. R., Geyer, R., Wilcox, C., Siegler, T. R., Perryman, M., Andrady, A., Narayan, R., and Law, K. L., 2015. Plastic waste inputs from land into the ocean. *science*, 347 (6223), 768–771.

Jamison, N., 2009. *The role of biorock artificial coral reefs in the sustainable governance of marine protected areas: a case study of Pemuteran, Bali* (doctoral dissertation) *,* [online]. London: University of London.

Johan, O., Bengen, D. G., Zamani, N. P., and Sweet, M. J., 2015. The distribution and abundance of black band disease and white syndrome in Kepulauan Seribu, Indonesia. *HAYATI Journal of Biosciences*, 22 (3), 105–112.

Karim, W., 2019. Status of coral diseases and compromised health syndromes on Pemuteran shallow reefs, North Bali island. *Earth and Environmental Science,* 236 (2), 12048.

Karmini, N. W., 2020. *Ecotourism management based on local wisdom in Tenganan village, Karangasem Bali,* [online]. Denpasar: UNHI

Karunia, A. M., 2021. Pemerintah Anggarkan Rp 1,5 Triliun untuk Restorasi Terumbu Karang di 9 Wilayah Wisata Baru Tahun Ini. Available online: https://money.kompas.com/read/2021/02/15/190611626/pemerintah-anggarkan-rp-15-triliun-untukrestorasi-terumbu-karang-di-9-wilayah..

Keller, K., Steffe, A. S., Lowry, M. B., Murphy, J. J., Smith, J. A., and Suthers, I. M., 2017. Estimating the recreational harvest of fish from a nearshore designed artificial reef using a pragmatic approach. *Fisheries Research*, 187, 158–167.

Kirkbride-Smith, A. E., Wheeler, P. M., and Johnson, M. L., 2016. Artificial reefs and marine protected areas: a study in willingness to pay to access Folkestone Marine Reserve, Barbados, West Indies. *PeerJ*, 4, e2175.

Kitney, S., Stanway, A. R., and Ryan, M. M., 2018. Volunteer tourism motivations of the Marine Conservation Cambodia project. *Current Issues in Tourism*, 21 (10), 1091–1096.

Koeck, B., Pastor, J., Larenie, L., Astruch, P., Saragoni, G., Jarraya, M., and Lenfant, P., 2011. Evaluation of impact of artificial reefs on artisanal fisheries: need for complementary approaches. *Brazilian Journal of Oceanography*, 59 (SPE1), 1–11.

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Koop, K., Booth, D., Broadbent, A., Brodie, J., Bucher, D., Capone, D., Coll, J., Dennison, W., Erdmann, M., and Harrison, P., 2001. ENCORE: the effect of nutrient enrichment on coral reefs. Synthesis of results and conclusions. *Marine pollution bulletin*, 42 (2), 91–120.

Lapointe, B. E., Brewton, R. A., Herren, L. W., Porter, J. W., and Hu, C., 2019. Nitrogen enrichment, altered stoichiometry, and coral reef decline at Looe Key, Florida Keys, USA: a 3-decade study. *Marine Biology*, 166 (8), 1–31.

Lazuardi, M. E., Sudiarta, I. K., Ratha, I., Ampou, E. E., Nugroho, S. C., and Mustika, P. L., 2013. The Status of Coral Reefs in Bali. *In*: *Bali Marine Rapid Assessment Program 2011*. *BioOne*, 69.

Leadley, P. W., Krug, C. B., Alkemade, R., Pereira, H. M., Sumaila, U. R., Walpole, M., Marques, A., Newbold, T., Teh, L. S. L., and van Kolck, J., 2014. *Progress towards the Aichi Biodiversity Targets: An assessment of biodiversity trends, policy scenarios and key actions,* [online]. Montreal: Secretariat of the Convention on Biological Diversity.

Leisher, C., Mangubhai, S., Hess, S., Widodo, H., Soekirman, T., Tjoe, S., Wawiyai, S., Larsen, S. N., Rumetna, L., and Halim, A., 2012. Measuring the benefits and costs of community education and outreach in marine protected areas. *Marine Policy*, 36 (5), 1005–1011.

Lesser, M. P., 2011. Coral bleaching: causes and mechanisms. *In*: *Coral reefs: an ecosystem in transition*. Springer, 405–419.

Lestari, P. and Trihadiningrum, Y., 2019. The impact of improper solid waste management to plastic pollution in Indonesian coast and marine environment. *Marine pollution bulletin*, 149, 110505.

Lewis, S. A., Setiasih, N., Dharmadi, D., O'Malley, M. P., Campbell, S. J., Yusuf, M., and Sianipar, A. B., 2015. Assessing Indonesian manta and devil ray populations through historical landings and fishing community interviews. *PeerJ PrePrints*, 6, e1334v1.

Lewis, S. and Mallela, J., 2018. *A multifactor risk analysis of the record 2016 great barrier reef bleaching,* [online].Canberra: Australian National University.

Lian, T., Chen, D., and Tang, Y., 2017. Genesis of the 2014–2016 El Niño events. *Science China Earth Sciences*, 60 (9), 1589–1600.

LINI, 2021. Seven sites at six villages in Buleleng North Bali construct and install artificial reef structures for their reef restoration program*,* [online].Bali: LINI.

Lison, F., Jiménez-Franco, M. V, Altamirano, A., Haz, A., Calvo, J. F., and Jones, G., 2020. Bat ecology and conservation in semi-arid and arid landscapes: a global systematic review. Mammal Review, 50 (1), 52–67.

Loya, Y., Lubinevsky, H., Rosenfeld, M., and Kramarsky-Winter, E., 2004. Nutrient enrichment caused by in situ fish farms at Eilat, Red Sea is detrimental to coral reproduction. *Marine Pollution Bulletin*, 49 (4), 344–353.

Lundquist, C. J. and Granek, E. F., 2005. Strategies for successful marine conservation: integrating socioeconomic, political, and scientific factors. *Conservation Biology*, 19 (6), 1771–1778.

Madeira, D., Andrade, J., Leal, M. C., Ferreira, V., Rocha, R. J. M., Rosa, R., and Calado, R., 2020. Synergistic Effects of Ocean Warming and Cyanide Poisoning in an Ornamental Tropical Reef Fish. *Frontiers in Marine Science*, 7, 246.

Mahmud, A., Satria, A., and Kinseng, R. A., 2016. TERITORIALISASI DAN KONFLIK NELAYAN DI TAMAN NASIONAL BALI BARAT. *Jurnal Sosial Ekonomi Kelautan dan Perikanan*, 11 (1), 45–54.

Marcos, C., Díaz, D., Fietz, K., Forcada, A., Ford, A., García-Charton, J.A., Goñi, R., Lenfant, P.,Mallol, S., Mouillot, D. and Pérez-Marcos, M., 2021. Reviewing the ecosystem services, societal goods, and benefits of marine protected areas. *Frontiers in Marine Science*, 8, 613819.

Mahyuni, L. P., Yoga, I. M. S., Permana, G. P. L., and Setiawan, I. W. A., 2020. Bagaimana bisnis eco-park menarik minat milenial? sebuah model untuk memahami intensi milenial mengunjungi eco-park. *In*: *FORUM EKONOMI*. 218–231.

Mair, L., Mill, A. C., Robertson, P. A., Rushton, S. P., Shirley, M. D. F., Rodriguez, J. P., and McGowan, P. J. K., 2018. The contribution of scientific research to conservation planning. *Biological Conservation*, 223, 82–96.

Mangubhai, S., Erdmann, M. V, Wilson, J. R., Huffard, C. L., Ballamu, F., Hidayat, N. I., Hitipeuw, C., Lazuardi, M. E., Pada, D., and Purba, G., 2012. Papuan Bird's Head Seascape: Emerging threats and challenges in the global center of marine biodiversity. *Marine pollution bulletin*, 64 (11), 2279–2295.

Marshall, A., Barreto, R., Carlson, J., Fernando, D., Fordham, S., Francis, M. P., Herman, K., Jabado, R. W., Liu, K. M., and Pacoureau, N., 2019. a Mobula alfredi. *The IUCN Red List of Threatened Species,* [online]. Gland: IUCN.

Mastny, L., 2001. A Worldwatch Addendum. *World Watch*, 14 (3), 20–21.

Maynard, J., Van Hooidonk, R., Eakin, C. M., Puotinen, M., Garren, M., Williams, G., Heron, S. F., Lamb, J., Weil, E., and Willis, B., 2015. Projections of climate conditions that increase coral disease susceptibility and pathogen abundance and virulence. *Nature Climate Change*, 5 (7), 688–694.

McLeod, E. and Palmer, M., 2015. Why conservation needs religion. *Coastal Management*, 43 (3), 238–252.

Mills, M., Magris, R. A., Fuentes, M. M. P. B., Bonaldo, R., Herbst, D. F., Lima, M. C. S., Kerber, I. K. G., Gerhardinger, L. C., de Moura, R. L., and Domit, C., 2020. Opportunities to close the gap between science and practice for Marine Protected Areas in Brazil. *Perspectives in Ecology and Conservation*, 18 (3), 161– 168.

Moberg, F. and Folke, C., 1999. Ecological goods and services of coral reef ecosystems. *Ecological economics*, 29 (2), 215–233.

Morikawa, M. K. and Palumbi, S. R., 2019. Using naturally occurring climate resilient corals to construct bleaching-resistant nurseries. *Proceedings of the National Academy of Sciences*, 116 (21), 10586–10591.

Mustika, P. L. and Ratha, I., 2013. Towards the Bali MPA Network. *In*: *Bali Marine Rapid Assessment Program 2011*. BioOne, 131.

NBRC, 2019. *North Bali Reef Conservation 2018 - 2019 Annual Report,* [online]. Bali: North Bali Reef Conservation.

Nõges, P., Argillier, C., Borja, Á., Garmendia, J. M., Hanganu, J., Kodeš, V., Pletterbauer, F., Sagouis, A., and Birk, S., 2016. Quantified biotic and abiotic responses to multiple stress in freshwater, marine and ground waters. *Science of the Total Environment*, 540, 43–52.

Normile, D., 2016. El Niño's warmth devastating reefs worldwide. *Science,* 352 (6281), 15-16.

O'Malley, M. P., Lee-Brooks, K., and Medd, H. B., 2013. The global economic impact of manta ray watching tourism. *PloS one*, 8 (5), e65051.

Onaka, S., Prasetyo, R., Endo, S., and Yoshii, I., 2013. Large-scale coral transplantation in artificial substrates at a shallow lagoon in Kuta Beach, Bali, Indonesia. *Galaxea, Journal of Coral Reef Studies*, 15 (Supplement), 336–342.

Orams, M. B., 2002. Marine ecotourism as a potential agent for sustainable development in Kaikoura, New Zealand. *International Journal of Sustainable Development*, 5 (3), 338–352.

Pandolfi, J. M., Connolly, S. R., Marshall, D. J., and Cohen, A. L., 2011. Projecting coral reef futures under global warming and ocean acidification. *Science*, 333 (6041), 418–422.

Parsons, E. C. M., MacPherson, R., and Villagomez, A., 2017. Marine "conservation": you keep using that word but I don't think it means what you think it means. *Frontiers in Marine Science*, 4, 299.

Pedju, F. M. K., 2018. Stakeholders' perceptions of the impacts of tourism on the social and ecological resilience of marine protected areas in Bali, Indonesia (doctoral thesis)*,* [online]. Auckland: AUT.

Pérez-Ruzafa, A., García-Charton, J.A. and Marcos, C., 2017. North East Atlantic vs. Mediterranean marine protected areas as fisheries management tool. *Frontiers in Marine Science*, 4, 245.

Perkol-Finkel, S., Shashar, N., Barneah, O., Ben-David-Zaslow, R., Oren, U., Reichart, T., Yacobovich, T., Yahel, G., Yahel, R., and Benayahu, Y., 2005. Fouling reefal communities on artificial reefs: does age matter? *Biofouling*, 21 (2), 127–140.

Perkol-Finkel, S., Shashar, N., and Benayahu, Y., 2006. Can artificial reefs mimic natural reef communities? The roles of structural features and age. *Marine environmental research*, 61 (2), 121–135.

Pet-Soede, L. and Erdmann, M., 1998. An overview and comparison of destructive fishing practices in Indonesia. *SPC Live reef fish information bulletin*, 4, 28–36.

Pickering, H. and Whitmarsh, D., 1997. Artificial reefs and fisheries exploitation: a review of the 'attraction versus production'debate, the influence of design and its significance for policy. *Fisheries research*, 31 (1–2), 39–59.

Piskurek, N., 2001. Dive tourism in coral reefs-impacts and conditions for sustainability: a case study from Desa Pemuteran (Bali/ Indonesia). *ACP-EU Fisheries Research Report*, 10, 57–65.

Plass-Johnson, J. G., Schwieder, H., Heiden, J., Weiand, L., Wild, C., Jompa, J., Ferse, S. C. A., and Teichberg, M., 2015. A recent outbreak of crown-of-thorns starfish (Acanthaster planci) in the Spermonde Archipelago, Indonesia. *Regional Environmental Change*, 15 (6), 1157–1162.

Polunin, N. V. C., Halim, M. K., and Kvalvågnaes, K., 1983. Bali Barat: An Indonesian marine protected area and its resources. *Biological Conservation*, 25 (2), 171–191.

Pörtner, H.-O., Karl, D. M., Boyd, P. W., Cheung, W., Lluch-Cota, S. E., Nojiri, Y., Schmidt, D. N., Zavialov, P. O., Alheit, J., and Aristegui, J., 2014. Ocean systems. *In*: *Climate change 2014: impacts, adaptation, and vulnerability. Part A: global and sectoral aspects. contribution of working group II to the fifth assessment report of the intergovernmental panel on climate change*. Cambridge: Cambridge University Press, 411– 484.

Pradisty, N. A., Ampou, E. E., and Hanintyo, R., 2020. Water Quality Assessment in The Occurrence of Acanthaster spp.(Crown-of-Thorns Starfish, CoTS) on Coral Reefs in Menjangan Island, Bali, Indonesia. *Makara Journal of Science*, 24 (3), 8.

Prasetia, D., Supriharyono, M., Anggoro, S., and Sya'Rani, L., 2017. Coral Bleaching on Lembongan Island, Nusa Penida, Bali. *Second International Conference of Innovative Research Across Disciplines.* 134, 66-72.

Pratchett, M. S., Hoey, A. S., and Wilson, S. K., 2014. Reef degradation and the loss of critical ecosystem goods and services provided by coral reef fishes. *Current Opinion in Environmental Sustainability*, 7, 37–43.

Principe, P., Bradley, P., Yee, S., Fisher, W., Johnson, E., Allen, P., and Campbell, D., 2012. *Quantifying coral reef ecosystem services. US Environmental Protection Agency, Office of Research and Development,* [online]. New York: Research Triangle Park.

Puspasari, R., Wiadnyana, N. N., Hartati, S. T., Rachmawati, R., and Yahya, Y., 2020. The Effectiveness of Artificial Reefs in Improving Ecosystem Health to Increase Coral Reef Resilience. *Jurnal Segara*, 16 (2), 115–126.

Raven, J., Caldeira, K., Elderfield, H., Hoegh-Guldberg, O., Liss, P., Riebesell, U., Shepherd, J., Turley, C., and Watson, A., 2005. *Ocean acidification due to increasing atmospheric carbon dioxide*, [online]. London: The Royal Society.

Relini, M., Torchia, G., and Relini, G., 1994. Seasonal variation of fish assemblages in the Loano artificial reef (Ligurian Sea Northwestern-Mediterranean). *Bulletin of Marine Science*, 55 (2–3), 401–417.

Richardson, L. E., Graham, N. A. J., Pratchett, M. S., Eurich, J. G., and Hoey, A. S., 2018. Mass coral bleaching causes biotic homogenization of reef fish assemblages. *Global Change Biology*, 24 (7), 3117–3129.

Roa-Ureta, R. H., Santos, M. N., and Leitão, F., 2019. Modelling long-term fisheries data to resolve the attraction versus production dilemma of artificial reefs. *Ecological Modelling*, 407, 108727.

Robinson, A., Polunin, N., Kvalvågnaes, K., and Halim, M., 1981. Progress in creating a marine reserve system in Indonesia. *Bulletin of Marine Science*, 31 (3), 774–785.

Roche, R. C., Williams, G. J., and Turner, J. R., 2018. Towards developing a mechanistic understanding of coral reef resilience to thermal stress across multiple scales. *Current Climate Change Reports*, 4 (1), 51–64.

Rosado, P. M., Leite, D. C. A., Duarte, G. A. S., Chaloub, R. M., Jospin, G., da Rocha, U. N., Saraiva, J. P., Dini-Andreote, F., Eisen, J. A., and Bourne, D. G., 2019. Marine probiotics: increasing coral resistance to bleaching through microbiome manipulation. *The ISME journal*, 13 (4), 921–936.

Ruchimat, T., Basuki, R., and Welly, M., 2013. Nusa Penida Marine Protected Area (Mpa) Bali-Indonesia: Why Need to be Protected? *Transylvanian Review of Systematical and Ecological Research*, 15 (1), 193–202.

Ruppert, J. L. W., Travers, M. J., Smith, L. L., Fortin, M.-J., and Meekan, M. G., 2013. Caught in the middle: combined impacts of shark removal and coral loss on the fish communities of coral reefs. *PloS one*, 8 (9), e74648.

Salm, R. V, 2005. *Coral reef resilience and resistance to bleaching*, [online]. Gland: IUCN.

Schurer, A. P., Cowtan, K., Hawkins, E., Mann, M. E., Scott, V., and Tett, S. F. B., 2018. Interpretations of the Paris climate target. *Nature Geoscience*, 11 (4), 220–221.

Selgrath, J. C., Peterson, G. D., Thyresson, M., Nyström, M., and Gergel, S. E., 2017. Regime Shifts and Spatial Resilience in a Coral Reef Seascape. *In*: *Learning Landscape Ecology*. Springer, 301–322.

Sembiring, A., Pertiwi, N. P. D., Mahardini, A., Wulandari, R., Kurniasih, E. M., Kuncoro, A. W., Cahyani, N. K. D., Anggoro, A. W., Ulfa, M., and Madduppa, H., 2015. DNA barcoding reveals targeted fisheries for endangered sharks in Indonesia. *Fisheries Research*, 164, 130–134.

SER, 2004. *Society for ecological restoration international science & policy working group*, [online]. Washington: Society for Ecological Restoration.

Shani, A., Polak, O., and Shashar, N., 2012. Artificial reefs and mass marine ecotourism. *Tourism Geographies*, 14 (3), 361–382.

Shuker, I. G. and Cadman, C. A., 2018. *Indonesia-Marine debris hotspot rapid assessment: synthesis report,* [online]. Washington: The World Bank.

Sigit, D. V, Miarsyah, M., Komala, R., Suryanda, A., Fadrikal, R., and Ichsan, I. Z., 2019. Improvement of knowledge and attitude in conservation of mangrove and coral reefs through environmental education community network model. *Journal of Physics,* 1317, 12201.

Soejoto, A., Subroto, W. T., and Suyanto, Y., 2015. Fiscal decentralization policy in promoting Indonesia human development. *International Journal of Economics and Financial Issues*, 5 (3).

Van Soest, H. L., den Elzen, M. G. J., and van Vuuren, D. P., 2021. Net-zero emission targets for major emitting countries consistent with the Paris Agreement. *Nature communications*, 12 (1), 1–9.

Sowman, M. and Sunde, J., 2018. Social impacts of marine protected areas in South Africa on coastal fishing communities. *Ocean & coastal management*, 157, 168–179.

Stafford, R. and Jones, P. J. S., 2019. Viewpoint–Ocean plastic pollution: A convenient but distracting truth? *Marine policy*, 103, 187–191.

Stem, C. J., Lassoie, J. P., Lee, D. R., and Deshler, D. J., 2003. How'eco'is ecotourism? A comparative case study of ecotourism in Costa Rica. *Journal of sustainable tourism*, 11 (4), 322–347.

Stolk, P., Markwell, K., and Jenkins, J., 2005. Perceptions of artificial reefs as scuba diving resources: a study of Australian recreational scuba divers. *Annals of Leisure Research*, 8 (2–3), 153–166.

Stolk, P., Markwell, K., and Jenkins, J. M., 2007. Artificial reefs as recreational scuba diving resources: a critical review of research. *Journal of sustainable tourism*, 15 (4), 331–350.

Strychar, K. B., Coates, M., Sammarco, P. W., Piva, T. J., and Scott, P. T., 2005. Loss of Symbiodinium from bleached soft corals Sarcophyton ehrenbergi, Sinularia sp. and Xenia sp. *Journal of Experimental Marine Biology and Ecology*, 320 (2), 159–177.

Suadi, N. M., 2009. Sharing the commons: resolving the tragedy of the commons through collaborative Management of Coastal Commons at Kedonganan Village. *Bali. J Kyosei Stud*, 3 (1), 263–285.

Subhan, B., Arafat, D., Rahmawati, F., Dasmasela, Y. H., Royhan, Q. M., Madduppa, H., Santoso, P., and Prabowo, B., 2020. Coral disease at Mansuar Island, Raja Ampat, Indonesia. *Earth and Environmental Science,* 429 (1), 12027.

Sugiri, K. G. L. and Mahyuni, L. P., 2019. SUSTAINABLE TOURISM PRACTICES AS A STRATEGY TO ENHANCE CORPORATE BRAND, [online]. *International Journal of Business, Economics and Law.* 20 (5), 2289.

Sully, S., Burkepile, D. E., Donovan, M. K., Hodgson, G., and Van Woesik, R., 2019. A global analysis of coral bleaching over the past two decades. *Nature communications*, 10 (1), 1–5.

Sumadhiharga, O. K., 1977. A preliminary study on the ecology of the coral reef of Pombo Island. *Marine Research In Indonesia*, 17, 29–49.

Suparno, N., Currier, K., Milner, C., Alling, A., and Dustan, P., 2019. Ecological changes in the coral reef communities of Indonesia's Bali Barat National Park, 2011–2016. *Atoll Res Bull*, 620, 1–35.

Sur, C., Abbott, J. M., Ambo-Rappe, R., Asriani, N., Hameed, S. O., Jellison, B. M., Lestari, H. A., Limbong, S. R., Mandasari, M., and Ng, G., 2018. Marine debris on small islands: Insights from an educational outreach program in the Spermonde Archipelago, Indonesia. *Frontiers in Marine Science*, 5, 35.

Susiloningtyas, D., Handayani, T., and Amalia, A. N., 2018. The impact of coral reefs destruction and climate change in Nusa Dua and Nusa Penida, Bali, Indonesia. *Earth and Environmental Science*. 145 (1), 12054.

Suteja, Y., Atmadipoera, A. S., Riani, E., Nurjaya, I. W., Nugroho, D., and Cordova, M. R., 2021. Spatial and temporal distribution of microplastic in surface water of tropical estuary: Case study in Benoa Bay, Bali, Indonesia. *Marine pollution bulletin*, 163, 111979.

Syam, A. R., Edrus, I. N., and Hartati, S. T., 2017. Coral fish population changes in the surrounding artificial reefs of the Lebah coastal waters, Karangasem, Bali. *Indonesian Fisheries Research Journal*, 13 (2), 101– 116.

Szmant, A. M., 2002. Nutrient enrichment on coral reefs: is it a major cause of coral reef decline? *Estuaries*, 25 (4), 743–766.

Tacconi, L., Rodrigues, R. J., and Maryudi, A., 2019. Law enforcement and deforestation: Lessons for Indonesia from Brazil. *Forest policy and economics*, 108, 101943.

Tapsuwan, S. and Rongrongmuang, W., 2015. Climate change perception of the dive tourism industry in Koh Tao Island, Thailand. *Journal of Outdoor Recreation and Tourism*, 11, 58–63.

Tito, C. K. and Ampou, E. E., 2020. Coral reefs ecosystem degradation at Nusa Penida, Bali. *Earth and Environmental Science*. 429 (1), 12053.

Tito, C. K., Ampou, E. E., and Wibawa, T. A., 2019. Stressor-Response of Reef-Building Corals to Climate Change in the Menjangan Island, West Bali National Park, Indonesia. *Earth and Environmental Science*, 246 (1), 12011.

Tomomi, I., 2010. Ecotourism in Bali: Backgrounds, present conditions and challenges (unpublished doctoral thesis), [online]. Kyoto: Ritsumeikan University.

Tompkins, E. L., 2003. *Development pressures and management considerations in small Caribbean islands' coastal zones,* [online]. Norwich: CSERGE.

Trialfhianty, T. I., 2017. The role of the community in supporting coral reef restoration in Pemuteran, Bali, Indonesia. *Journal of Coastal Conservation*, 21 (6), 873–882.

Turak, E. and Devantier, L., 2013. Biodiversity and conservation priorities of reef-building corals in Bali, Indonesia. *Bali Marine Rapid Assessment Program 2011*, BioOne, 78.

Vega Thurber, R. L., Burkepile, D. E., Fuchs, C., Shantz, A. A., McMinds, R., and Zaneveld, J. R., 2014. Chronic nutrient enrichment increases prevalence and severity of coral disease and bleaching. *Global change biology*, 20 (2), 544–554.

Velis, C. and Cook, E., 2020. Mismanagement of plastic waste through open burning in the Global South: A systematic review of risks to occupational and public health. *EngrXiv Preprints.*

Vercelloni, J., Liquet, B., Kennedy, E. V, González-Rivero, M., Caley, M. J., Peterson, E. E., Puotinen, M., Hoegh-Guldberg, O., and Mengersen, K., 2020. Forecasting intensifying disturbance effects on coral reefs. *Global Change Biology*, 26 (5), 2785–2797.

Villarrubia-Gómez, P., Cornell, S. E., and Fabres, J., 2018. Marine plastic pollution as a planetary boundary threat–The drifting piece in the sustainability puzzle. *Marine Policy*, 96, 213–220.

Voss, J. D. and Richardson, L. L., 2006. Nutrient enrichment enhances black band disease progression in corals. *Coral Reefs*, 25 (4), 569–576.

Wall, G., 1997. Is ecotourism sustainable? *Environmental management*, 21 (4), 483–491.

Walton, C. J., Hayes, N. K., and Gilliam, D. S., 2018. Impacts of a regional, multi-year, multi-species coral disease outbreak in Southeast Florida. *Frontiers in Marine Science*, 5, 323.

Wardana, A. A. G. O. P., Utama, M. S., Yasa, I. N. M., and Budiasa, I. G. S., 2018. Effect of Cummunity Participation, Tourism Infrastructure, Tourist Visit to The Tourism Industry Performance and The Quality

Living Community in Bali Indonesia. *International Journal of Sustainability, Education, and Global Creative Economic (IJSEGCE)*, 1 (1), 79–86.

Wearing, S., 2001. *Volunteer tourism: Experiences that make a difference,* [online]. New York: CABI.

Weeks, R., Aliño, P. M., Atkinson, S., Beldia, P., Binson, A., Campos, W. L., Djohani, R., Green, A. L., Hamilton, R., and Horigue, V., 2014. Developing marine protected area networks in the Coral Triangle: good practices for expanding the Coral Triangle Marine Protected Area System. *Coastal Management*, 42 (2), 183–205.

Wenker, R. P. and Stevens, B. G., 2020. Sea whip coral Leptogorgia virgulata in the Mid-Atlantic Bight: Colony complexity, age, and growth. *PeerJ*, 8, e8372.

Wicaksana, I. B. A., 2020. 2020 Bali's Coral Conservation: from pandemic challenge to Government coral garden project. *Bali Tourism Journal*, 4 (2), 35–39.

Wijaya, A., Chrysolite, H., Ge, M., Wibowo, C. K., Pradana, A., Utami, A., and Austin, K., 2017. How can Indonesia achieve its climate change mitigation goal? An analysis of potential emissions reductions from energy and land-use policies. *World Resources Institute. World Resour Inst Work Pap*, 1–36.

Williams, D. E. and Miller, M. W., 2005. Coral disease outbreak: pattern, prevalence and transmission in Acropora cervicornis. *Marine Ecology Progress Series*, 301, 119–128.

Yunitawati, D. and Clifton, J., 2019. Governance in the early stages of marine protected area development: A case study of Nusa Penida District Marine Conservation Area, Indonesia. *Marine Policy*, 103653.

Zamani, N. P. and Madduppa, H. H., 2011. A standard criteria for assesing the health of coral reefs: implication for management and conservation. *Journal of Indonesia Coral Reefs*, 1 (2), 137–146.

Zhao, Q., Stephenson, F., Lundquist, C., Kaschner, K., Jayathilake, D., and Costello, M. J., 2020. Where Marine Protected Areas would best represent 30% of ocean biodiversity. *Biological Conservation*, 244, 108536.

Chapter 3: Convergence of tropical fish communities between artificial and natural reefs over time

Objectives: To assess how fish communities changed over 3 years since the large-scale deployment of ARs in north Bali, and to compare their communities to nearby natural reefs.

Contribution to new knowledge: This study is one of few initial evaluations of the use of artificial reefs (ARs) in Indonesia that assesses their potential to provide localised increases in fish abundance and biodiversity. The results of this study may be useful for communities particularly reliant on the ecosystem services provided coral reefs, especially those that have experienced a decline in the health of their natural reefs, and not only apply to reefs in Indonesia, but internationally too. Furthermore, this research utilises the 3 year data set which I have collected on ARs in north Bali. Ries (2021b) discussed that long term (3 years or more), robust data sets on AR fish communities are rare. From our literature review we have found there to be even less long-term data on tropical AR projects in countries like Indonesia and it is therefore important for longer term studies to be conducted in the region, and thus make key contributions to the field.

How this fits in the PhD: This chapter assesses how assemblages of fish on ARs can converge with those of nearby natural reefs, and also compares these communities to those on a nearby degraded sand flat. It therefore addresses the 'restoration' part of the PhD question (mapping onto objective 2).

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Abstract

In recent decades, artificial reefs have gained support as a restoration tool which increases fish biomass and diversity, especially in areas with limited habitat complexity and larval supply, such as degraded coral reefs. Indonesia is located within the 'coral triangle' and hosts the well-known tourism island of Bali, which has been identified as one of the countries coral reef conservation priorities due to its highly biodiverse reefs. Despite this, in recent decades, coral reef health in Indonesia has declined substantially due to threats such as climate change-induced bleaching, overfishing, destructive fishing practices and pollution, which have resulted in the widespread loss of coral reefs across the nation. To counter this trend, non-governmental organisations, often managed by local communities, have begun constructing artificial reefs as a tool to restore previously degraded reef habitats on a localised scale. Previous studies have highlighted that artificial reefs in Bali have been successful in enhancing habitats and restoring reef fish biodiversity, although few have assessed this over long term (3 years or more) time scales. Furthermore, there is a general lack of long-term, robust data sets on artificial mobile communities in the tropics (not just Indonesia), especially in terms of how they converge with the assemblages of natural coral reefs over time. To address this gap in the research, we conducted unbaited Remote Underwater Video surveys to assess mobile assemblages over a 3 year period (2020, 2021 and 2022) in Tianyar Bay, north Bali, in 3 habitat types: (1) a restoration site which had deployed ARs over a 5 year period (herein 'AR'), (2) a (relatively healthy) natural coral reef (herein 'CR', and (3) a previously degraded reef covered by sand (herein 'sand flat' ('SF')) as a control site. Number of fish species was compared between habitats using a generalised mixed model nested ANOVA, and differences in mobile community structure was analysed using PERMANOVA. Our first major finding was that AR mobile communities in 2021 and 2022 were substantially different from those in 2020, which we propose was because the individual ages of ARs became less important over time, and as their communities began to stabilise they may considered as 'one reef', instead of clusters of different aged units. Our second major finding was that the AR and CR communities became more similar over the 3 year sampling period, and by the final sampling period, they had the same average number of species. Despite this, the AR still displayed significant differences to the CR in terms of community structure, even in the final year of sampling. Over time, as benthic communities become more similar, it is likely that the

AR and CR communities will further converge, although a longer-term sampling period is needed to confirm this.

Key words: Coral reef restoration, Artificial reefs, Fish community structure, Remote underwater video, Bali, Indonesia.

Introduction

Coral reef degradation

Coral reefs have been greatly altered by anthropogenic threats over recent decades (Lesser 2011; Claar et al. 2018; Andrello et al. 2021), resulting in substantial declines in the health of mobile and benthic communities (Heron et al. 2017; Hughes et al. 2018). Although climate change induced bleaching is thought to be the greatest threat to corals globally (Kleypas and Eakin 2007; Van Hooidonk et al. 2016; Boström-Einarsson et al. 2020), although other threats, such as destructive fishing practices and pollution have been shown to further degrade reefs and reduce their resilience to bleaching (Selgrath et al. 2017, Boakes et al. 2022a, Lamont et al. 2022). The ecosystem services provided by coral reefs are estimated at a value of over US $1 trillion globally (Costanza et al. 2014), however the decline in coral reef health has impacted the provisioning of these vital services, resulting in reduced coastal protection from storms, loss of incomes from fishing and tourism, decreased availability of seafood and interrupted biogeochemical cycling processes (Bell et al. 2006; Brander et al. 2007).

Ben-Romdhane et al. (2020) discussed that the most important tool to protect coral reefs is the aggressive and large-scale reduction of greenhouse gas emissions, as a measure to prevent continued coral bleaching. Other 'reactive' conservation measures, notably the establishment of marine protected areas (MPAs), are widely used to conserve coral reefs through restricting potentially threatening human activity such as fishing (Edgar et al. 2014). Reactive tools continue to be effective in conserving coral reefs (Christie and White 2007; Ruchimat et al. 2013), however research has shown that these measures alone do not suffice in adequately protecting reefs against threats such as ocean warming, acidification, coral disease and pollution (Graham et al. 2020; Lamont et al. 2022). Additionally, reefs that are already degraded may be unable to recover from passive tools like this, especially if regime shifts to algal states have already occurred (Graham et al. 2015; Kenyon et al. 2020). In these cases, multiple small - medium scale 'pro-active' conservation tools may be considered by academic, governmental, non-profit and private sector organisations to restore degraded coral reefs (Lamont et al. 2022). Some of these methods may include coral transplantation (Endo et al. 2008; Onaka et al. 2013; Barton et al. 2017; Baria-Rodriguez et al. 2019), establishing coral nurseries with species that may have some resistance to bleaching (Camp et al. 2017; Morikawa and Palumbi 2019), genetically modifying reef building corals (Cleves et al. 2018) and constructing artificial reefs ((Mills et al. 2017, Folpp et al. 2020, Boakes et al. 2022b). Research has shown that these methods may be unable to conserve function and processes on a large scale (Pörtner et al. 2014), however they may provide some degree of protection over time at a localised level, and in some cases, restore ecosystem services in areas which have lost reefs entirely (Boakes et al., 2022b).

Artificial reefs

An artificial reef (AR) can be defined as any solid man-made structure which has been submerged in the marine environment (Bohnsack 1989). ARs have multiple functions in coastal management, such as increasing fisheries yield, boosting dive tourism and preventing bottom trawling (Baine 2001). In recent decades, they have also gained support as a reef restoration tool to enhance habitats (Paxton et al. 2020; Boakes et al., 2022b). Research has shown that ARs can boost fish communities when they are deployed in areas with previously limited habitat complexity and larval supply (Perkol-Finkel et al. 2006; Herbert et al. 2017). Well-designed ARs offer structural complexity, often in the form of multiple hiding spaces and exits, high surface areas and hollow interior spaces (Marinaro 1995; Lemoine et al. 2019). This allows greater colonisation of biological communities, as spawning adults benefit from a textured surface to lay their eggs, whilst juveniles are provided with shelter from predation (Herbert et al. 2017). In terms of fish assemblages, this additional habitat complexity has been shown to facilitate increased abundances at their range edges (Paxton et al. 2019), increase larval and juvenile recruitment, survival and growth (Mercader et al. 2017) and host elevated predator densities (Paxton, Newton, et al. 2020).

There is mixed evidence on whether ARs support similar fish communities to natural reefs (Reis et al. 2021a). In certain systems, ARs have been shown to support equivalent abundance and types of fish as natural reefs (Lemoine et al. 2019, Boakes et al. 2022b). Some studies have highlighted that ARs can support higher abundances and species richness compared to natural reefs (Arena et al. 2007; Paxton et al. 2017) and others have shown that they support fewer (Carr and Hixon 1997; Froehlich and Kline 2015). Research has also highlighted that ARs may create additional threats to the ecosystems where they are deployed, for example through causing changes in food-web structure, connectivity and larval dispersal patterns between natural and artificial habitats, as well as the introduction of pollutants, diseases and/or marine pests (Pears and Williams 2005).

Natural and artificial reefs in Bali, Indonesia

Indonesia is a country that sits within a global biodiversity hotspot known as the 'coral triangle' (Allen 2008) and makes up approximately 12.5 % of the world's total coral reef (Susiloningtyas et al. 2018). Bali is an island in Indonesia (figure 3.1) and has the second highest documented reef fish species richness in the Asia-pacific (Mustika and Ratha, 2013), with at least 805 documented fish species (Allen and Erdmann, 2013). Coral reefs of high conservation value have been identified in abundance along the east and north coasts of Bali (Turak and DeVantier 2011) and the island's reefs represent a large proportion of the US$3 billion annual tourism value from Indonesian coral reefs (Spalding et al. 2017).

Research has shown that 86% of Indonesia's coral reefs face medium or high levels of threat (Burke et al. 2012), with recent studies on Bali's reef in 2017 showed that that 50% of the island's coral reefs are in good health, whilst 20% are declining and 30% are poor (Marine and Fisheries Office 2017 data, as cited by Wicaksana (2020)). Boakes et al. (2022a) highlighted that the main threats to its reefs include coral bleaching (Prasetia et al. 2017; Suparno et al. 2019), destructive fishing practices (Doherty et al. 2013; Frey and Berkes 2014), marine pollution (Germanov et al. 2018; Argeswara et al. 2021) and tourism, often as a result of unsustainable dive and scuba practice (Gerungan and Chia 2020).

The importance of Bali's reefs, as highlighted by the previously cited literature, make it a global conservation priority (Turak and Devantier 2013; Wicaksana 2020). Indonesia has more coral reef restoration programs than any other country in the world, with hundreds of programmes having been established over the past 2 decades (Lamont et al. 2022). Coral transplantation is one of the most commonly used tools for coral restoration in Bali (and wider Indonesia; Onaka et al. 2013, Lamont et al. 2022)). Additionally, three marine protected areas have been officially designated in Bali in Nusa Penida, West Bali National Park and Pemutaran, but vary in their associated regulations, zonation strategies and local compliance (Boakes et al. 2022a). At the time of writing this paper, no coral reefs on the east or north coasts of Bali had been given official MPA status, although multiple locally established no-take zones existed across these coastlines, and had strong potential for the development of a network of MPAs if given sufficient logistic resources and long-term government support (Turak and DeVantier 2011, Boakes et al. 2022a).

Aside from coral transplantation and MPA designation, the construction of ARs is commonly used as a coral restoration tool in Bali, with tens of thousands units already deployed in degraded reefs on the coastlines of Bali's largest tourism hubs, such as Nusa Dua, to some of Bali's poorest regions like Buleleng and Karangasem (Puspasari et al. 2020; Wicaksana 2020). The deployment of AR structures in Bali vary greatly (in terms of design, cost, material etc), with some of the most common including concrete substrate blocks, Reef Ball© and Biorock™ (Boakes et al. 2022a). These ARs have been constructed by international NGOs, community groups, village governments and the central government. The use of ARs as a habitat enhancement tool is becoming widespread within Indonesia. One of the most popular AR construction methods in Bali is the utilisation of local communities to build handmade concrete structures, as used by the 'Indonesian Coral Reef Garden' (ICRG) initiative, which deployed 95,768 AR structures of varying designs in five areas in Bali (ICRG 2021). Previous studies have highlighted that ARs in Bali have been successful in terms of habitat enhancement and restoring reef biodiversity (e.g.(Syam et al. 2017, Puspasari et al. 2020, Boakes et al. 2022b), although few have assessed changes to communities over medium - long term time scales.

Our previous research in Tianyar, north Bali, provided evidence that ARs can support ecologically equivalent fish communities to natural reef on a localised scale (Boakes et al. 2022b). We also showed that, when compared with a nearby degraded sand habitat, ARs displayed a significantly higher number of species, although this study only assessed a short-term data set collected over one season. Ries (2021b) discussed that long term (3 years or more), robust data sets on AR fish communities are rare. The limited available mid-long term data sets particularly in terms of community stability are generally inconclusive, with Mills et al. (2017) highlighting optimal colonisation rates occur on ARs after 1 year of deployment, whereas (Kojansow et al. 2013) showing the same to occur between then 6th – 7th year. From our literature review we have found there to be even less on tropical AR projects in countries like Indonesia and it is therefore important for longer term studies to be conducted in the region, and thus make key contributions to the field. This current study is a follow-up of our previous research Boakes et al. (2022b), but now utilises our medium-term (3 year) data set to assess the development of AR fish assemblages over time. More specifically, this study is aiming to assess how fish communities changed over 3 years since the large-scale deployment of ARs in north Bali, and to compare their communities to nearby natural reefs.

Methods

Location

This research was conducted in Tianyar Bay, Karangasem Regency, north Bali, Indonesia (8°11'27.5"S, 115°29'42.9"E). Tianyar Bay was located within a No-Take Zone (NTZ) Marine Protected Area (MPA) which was established in 2017 by a fisher community locally known as Yowana Bhakti Segara (www.northbalireefconservation.com; 8°11'27.5"S 115°29'42.9"E). At the time of writing,

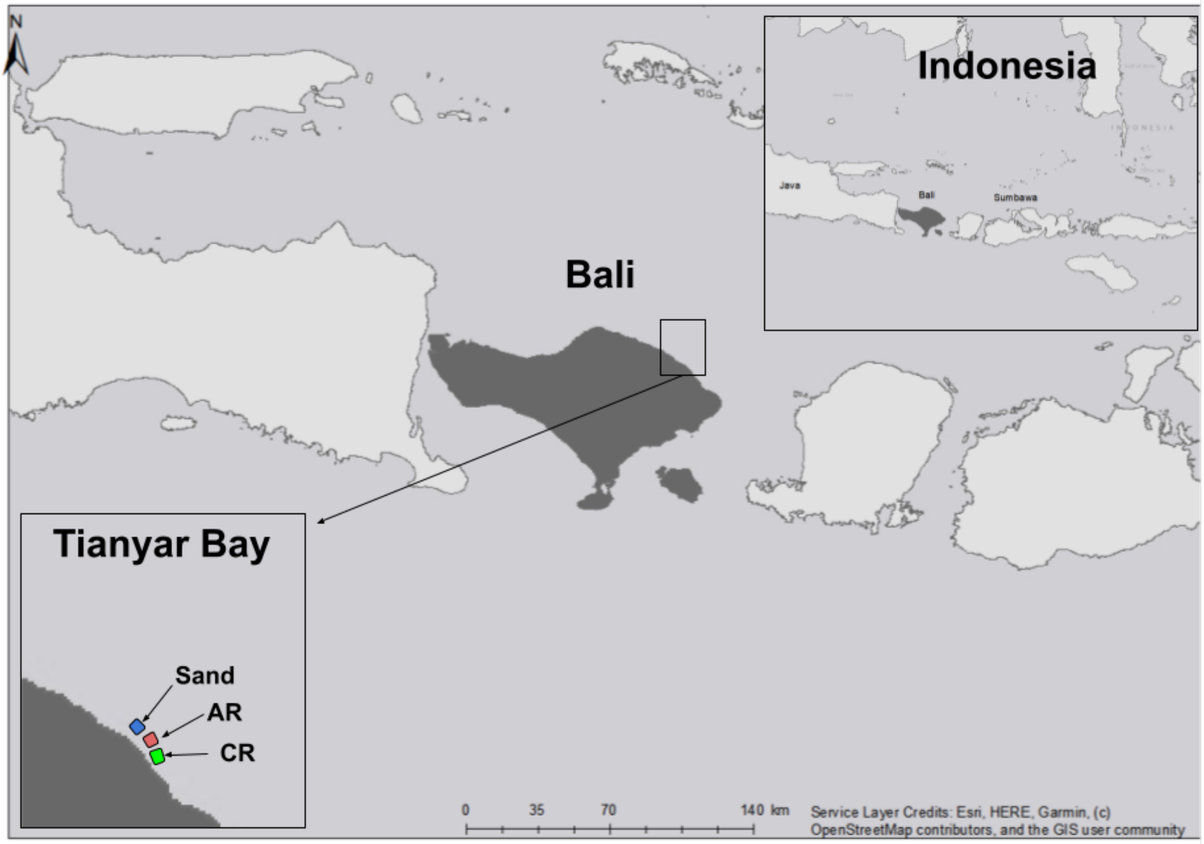

Figure 3. 1: Location of the three sampling sites (Sand Flat, Artificial Reef (AR) and Coral reef (CR)) within Tianyar Bay, Bali, Indonesia *(Created using ArcGIS OpenStreetMap powered by Esri).*

the MPA continued to be well-enforced, and the localised threats which were thought to previously degrade the reef appeared to had stopped, likely in large part due to establishment of this MPA. The MPA was enforced 24/7 by local fishers through beach patrols.

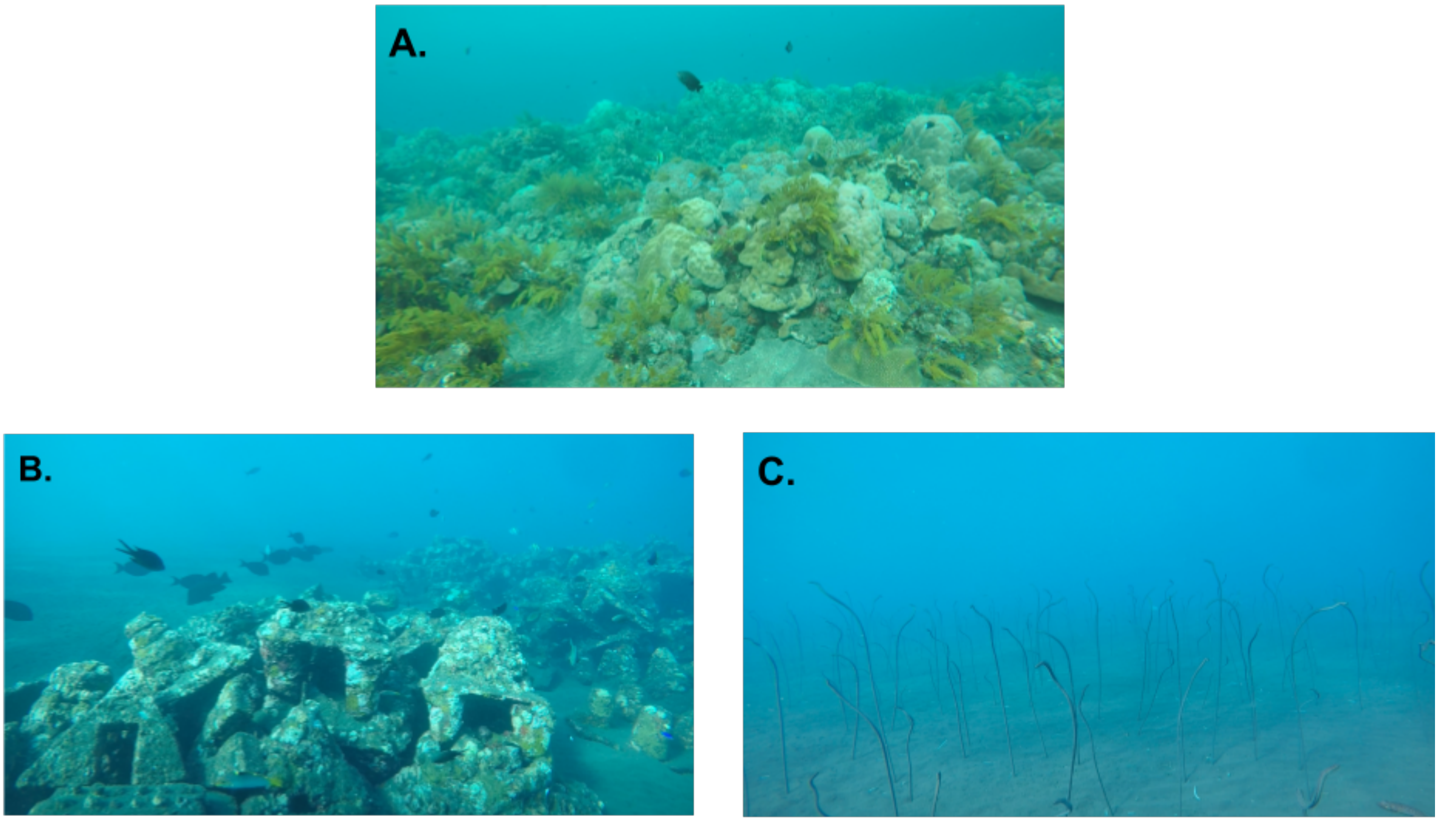

Figure 3. 2: Screenshots from Remote Underwater Video recordings at each of the three habitat types (coral reef (A), artificial reef (B) and sand flat (C)).

Within Tianyar Bay, three habitat types were surveyed; a sand flat (SF), an artificial reef (AR) and a coral reef (CR), as highlighted by figure 3.2. The SF was originally part of the natural coral reef, which was found to have been destroyed due to heavy boat traffic, anchoring and coral harvesting several decades ago (Boakes et al. 2023a), which were later made illegal in the area. Through communication with the local community, it was understood that these destructive activities were previously concentrated in this particular area, explaining why the SF was destroyed, but the CR remained relatively healthy. The combined threats in the SF area resulted in large areas of destroyed reef, covered mostly by sand and relatively devoid of life, hence the name 'sand flat' was given. The NGO North Bali Reef Conservation (NBRC) was established in 2017 with the aim of working with local fishers to restore part of this area by constructing artificial reefs and creating a locally-enforced an MPA. Since the NGO was founded, the community of Tianyar Village, have deployed approximately 20,000 AR units (at the time of this paper), which cover approximately a 2 hectare area. The 'AR' in researched for this study was the artificial reef site deployed by NBRC, and the 'SF' area was the degraded reef which had not yet been restored using artificial reefs . The CR was the natural coral reef adjacent to the AR (figure 3.1). Carr and Hixon (1997) describe how effective monitoring of AR performance should be conducted using comparisons with nearby undisturbed natural reefs (as per this study), by utilising a nearby natural CR. This study aimed to compare AR mobile communities to the nearby CR and SF, which were each approximately 250m apart.

ARs were deployed by local fishers and international volunteers between a depth of 5 – 10m and constructed using a three part mix of cement, calcium and sand. This produced what are known as 'roti buaya' (English translation: 'crocodile bread'), 0.75 × 0.5 m rectangular structures with rough textured surface to allow natural recruitment of coral and settlement of other species. The units were deployed on areas of flat sand or bare rubble, both lacking in physical complexity. They were installed in groups which ranged from 40 - 50 structures approximately 10m spacing between each group. Each group covered an area of approximately 10 - 20 m^2, where structures were stacked haphazardly (in a similar configuration between groups), with the aim of providing optimal protective space, such as holes, tunnels and caves, thus creating additional habitat for sheltering mobile / semi-mobile species.

All three habitat types were surveyed over a 3 year period during the months of July to September in 2020, 2021 and 2022 in Bali's dry season. The different habitat types were all studied within the same depth range (5 – 10 m) and had prevailing SW wind directions and easterly currents. Permission was given by NBRC to conduct the surveys and a research permit was obtained from BRIN (Indonesia's ministry of research). Three sample sites (herein sites) were established (in each of the three habitat types (herein habitats)) for monitoring. For the SF and CR, sub-sites were chosen haphazardly (approximately 50 m apart from each other), and to allow easy identification, each site was marked with a 30 cm^2 cement base attached to a metal frame and sign. For the AR, sub-sites were chosen based on age of the structures, with AR1 = 1 year old, AR2 = 2 years old and AR = 3 years old when monitoring started in 2020. The monitoring then continued to survey the same AR sub sites over the 3 year period (Table 3.1) to allow comparison of reef communities over time. To allow the divers to identify where to take the samples, each sub-site was marked out using a permanent weighted unit with a sign highlighting the sub-sites code (e.g. AR1, AR2, AR3 at the 3 artificial reef sites). Each of these markers were deployed 2 meters away from the desired subject.

Table 3. 1: Age of the three different artificial reef sub-sites at given surveying periods (2020, 2021 and 2022) over the 3 year data collection period.

Age of Artificial Reefs (years)			
	Year 1 (2020)	Year 2 (2021)	Year 3 (2022)
AR1	1	2	3
AR2	2	3	4
AR3	3	4	5

Data collection

Remote Underwater Video (RUV), a cost effective, safe and non-destructive technique commonly used to survey demersal fish assemblages (Langlois et al. 2020), and was used to analyse mobile communities in different habitats in Tianyar Bay. Contrary to common practice, it was decided that the RUV unit would be un-baited to eliminate the issue of bias by not intentionally attracting fish into the area with bait (as discussed by Bernard and Götz (2012)). A GoPro Hero 7 HD 1080p underwater camera was fixed to a weighted unit which was deployed by divers who placed the camera on flat empty area facing the desired habitat. Recordings were 25 minutes in duration, allowing for an initial 5 minute settlement period and 20 minutes of analysis time, following Boakes et al. (2022b) and Tweedie et al. (2023).

RUV surveys were taken only on clear non-cloudy mornings (between 7–9am on varying tidal conditions), when the environmental conditions (waves and wind) were calm, and water visibility was at least 15m (measured using underwater distance markers). Samples were taken from the same sub-site twice (N = 2) over the monitoring season, giving a total of 18 samples across all three habitat types per year (64 in total over the 3 year research period). To account for the potential variability in conditions over the 3 month sampling period, recordings were taken evenly across all locations and sites over time (for example: day 1 = FSB site 1, day 2 = NAR site 1, day 3 = CR site 1, day 4 = FSB site 2 etc).

Video analysis

The initial 5 minutes of every 25 minute recording were discarded due to potential disturbances to nearby mobile species (and possible behavioural changes) which may have been caused by snorkellers deploying the RUV unit (following Hall et al. (2021)). Each video was analysed using Quicktime Media Player and species were only recorded when accurate identification was possible. Mobile species were identified to species level using the guide 'Tropical Pacific Reef Fish Identification' (Allen et al. 2003), and in circumstances of uncertainty, advice was sought from nearby local experts from the ' LINI Les Aquaculture Training Centre (LATC)'.
As a measure of biodiversity, the maximum number of species (NoS) identified over the full 20 minute recording was noted (following Schramm et al. (2020) and Boakes et al. (2022b)). As a measure of abundance of these species, the maximum number of individuals seen in any frame (MaxN; following Whitmarsh et al. (2017) was recorded.

Statisticial analysis

A generalised mixed model nested ANOVA was run for NoS as a dependent variable using the glmer function in the lme4 package in R (Bates et al. 2014). Sub-site was a random factor in the ANOVA, which was nested within the habitat type. A Poisson link function was used to account for the use of count data, and examination of fitted vs. residual plots indicated the data were appropriate for this statistical model (following Zuur et al. (2009)). Significance was tested by dropping the main effect term and as comparing models, as detailed in Howlett et al. (2016). Differences between habitat types were examined using post-hoc tests with Tukey corrections (using the emmeans package—(Lenth 2021)).

PERMANOVA was run using R (following Anderson (2001)) to assess the difference in fish assemblages (MaxN for mobile species) between habitat types. Data was square-root transformed prior to use, to avoid the weighting of common species over rare. A Bray-Curtis resemblance matrix was used with 9999 permutations and PERMANOVA run with unrestricted permutation of raw data. Then, following Boakes et al. (2023b) Principal Coordinate Analysis (PCA) was used to illustrate variation between habitat types and to highlight key species which differentiated the different ecological communities at the different habitat types.

Results

The AR had an average NoS of 23.7, 39.3 and 42.9 in 2020, 2021 in years 2022 (respectively), with most common families including surgeonfish, damselfish and snapper. The coral reef had an average NoS of 28.9, 44.6 and 42.9 in 2020, 2021 and 2022 (respectively), with most common families including butterflyfish, damselfish and wrasse. The sand flat had an average NoS of 6.2, 4.3 and 4.9 in 2020, 2021 and 2022 (respectively), with its mobile populations being made up mostly spotted garden eel.

Figure 3.3 showed that each sampling year, the AR and CR had a substantially higher average NoS than the sand habitat. Mean NoS was greatest on the CR in 2020 and 2021, but similar to AR in 2022. Average NoS on the AR was shown to increase each year and was the same as on the CR by 2022. The CR appeared to fluctuate in terms of average NoS over the sampling period, whereas the sand habitat increased slightly each year.

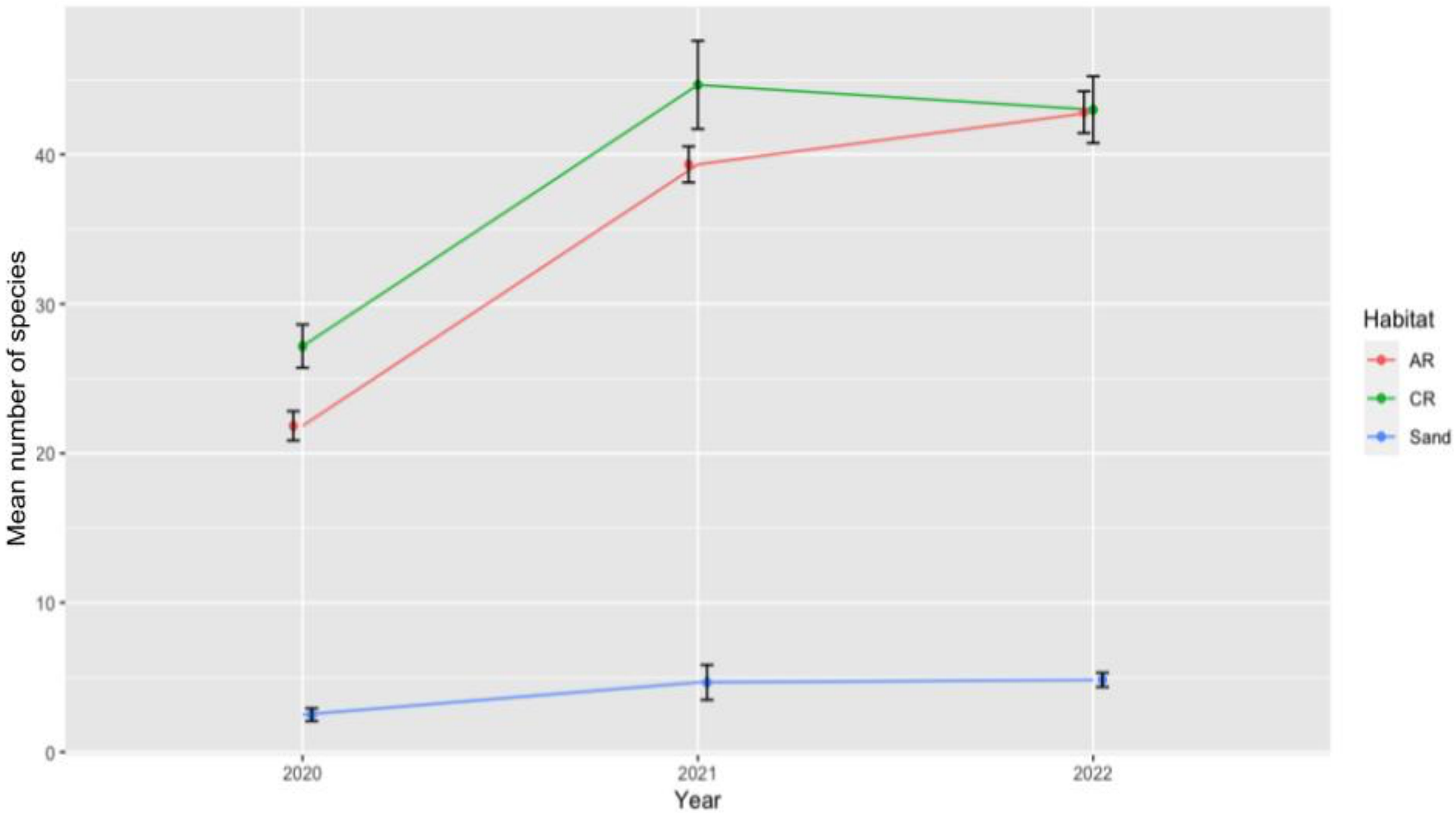

Figure 3. 3: Interaction plot (means +/-SE) of number of species in each habitat (AR, CR, sand) over a 3 year sampling period (2020, 2021, 2022).

Two way PERMANOVA showed significant interactions for habitat type and year ($F_{4,45}$=2.68, p=0.002), with pairwise comparisons showing significant differences between fish communities for all habitat types between all years (p<0.05) in all cases except two (sand 2020: sand 2021 p=0.061; AR 2021: AR 2022p=0.102). Figure 3.3 highlighted the convergence of AR 2022 and CR 2022 in terms of average NoS, however the two way PERMANOVA highlighted that there was still a significant difference in fish community structure between AR 2022 and CR 2022 (p=0.002).

Figure 3. 4: Principal Coordinate Analysis (PCA) plot for fish community structure between habitat types over a 3 year sampling period (2020, 2021, 2022), with overlaid arrows based on r > .45 and p <0.001.

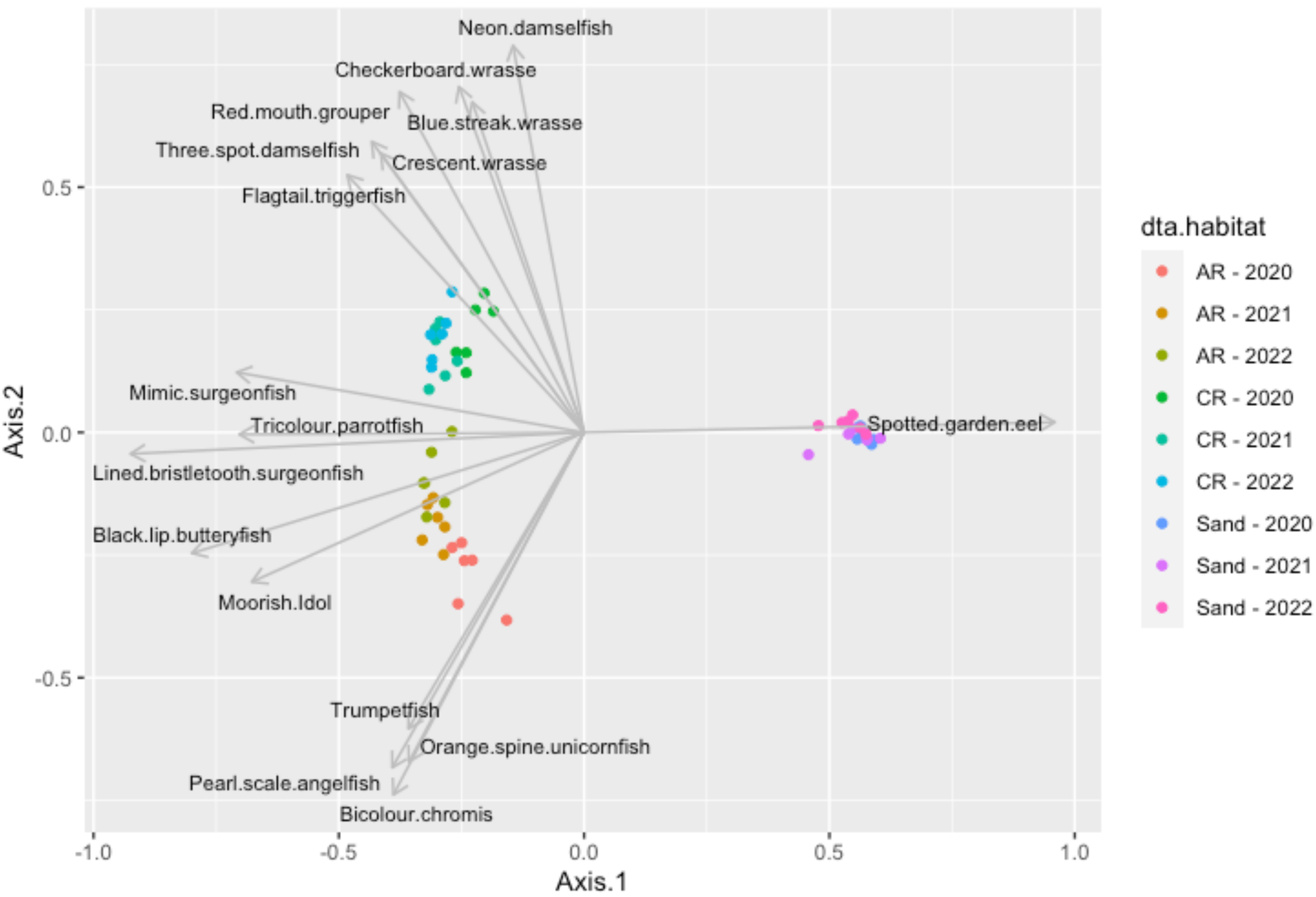

Table 3. 2: The presence (P; green) and absence (A; red) of the species highlighted in figure 3.4 for each habitat over the 3 year sampling period.

Species	**Artificial Reef**			**Coral Reef**			**Sand**		
	2020	2021	2022	2020	2021	2022	2020	2021	2022
Neon damselfish	P	P	A	P	A	A	A	A	A
Checkerboard wrasse	P	P	P	A	P	P	A	A	A
Blue streak wrasse	P	P	P	P	A	P	A	A	A
Crescent wrasse	P	P	P	P	P	P	A	A	A
Three spot damselfish	P	P	P	P	P	P	A	A	A
Flagtail triggerfish	P	P	P	P	P	P	A	A	A
Red mouth grouper	P	P	P	A	P	P	A	A	A
Mimic surgeonfish	P	P	P	P	P	P	A	A	A
Tricolour parrotfish	P	P	P	P	P	P	A	A	A
Lined bristletooth surgeonfish	P	P	P	P	P	P	A	A	A
Spotted garden eel	A	A	A	A	P	A	P	P	P
Black lip butterflyfish	P	P	P	P	A	P	A	A	A
Moorish idol	P	P	P	P	P	P	A	A	A
Trumpetfish	A	P	P	P	P	P	A	A	A
Pearl scale angelfish	P	P	P	P	P	P	A	A	A
Orange spine unicornfish	A	P	P	P	P	P	A	A	A
Bicolour chromis	A	P	P	P	P	P	A	A	A

Figure 3.4 and table 3.2 highlighted that over all time scales, communities on the AR and CR were more similar (close together in the plot), when compared to the sand. There was a clear difference in communities between all three habitats, with certain species driving these differences. Figure 3.4 showed that trumpetfish were more abundant on the AR, spotted garden eel were only seen on the sand and multiple wrasse species were more common on the CR. Figure 3.4 also highlighted that community structure in the sand and coral reef did not differ greatly over the time series, with overlaps of all three for both habitat types over the 3 years. In contrast, it showed substantial differences between the AR communities over the time series, which were shown to get more similar to the CR each year.

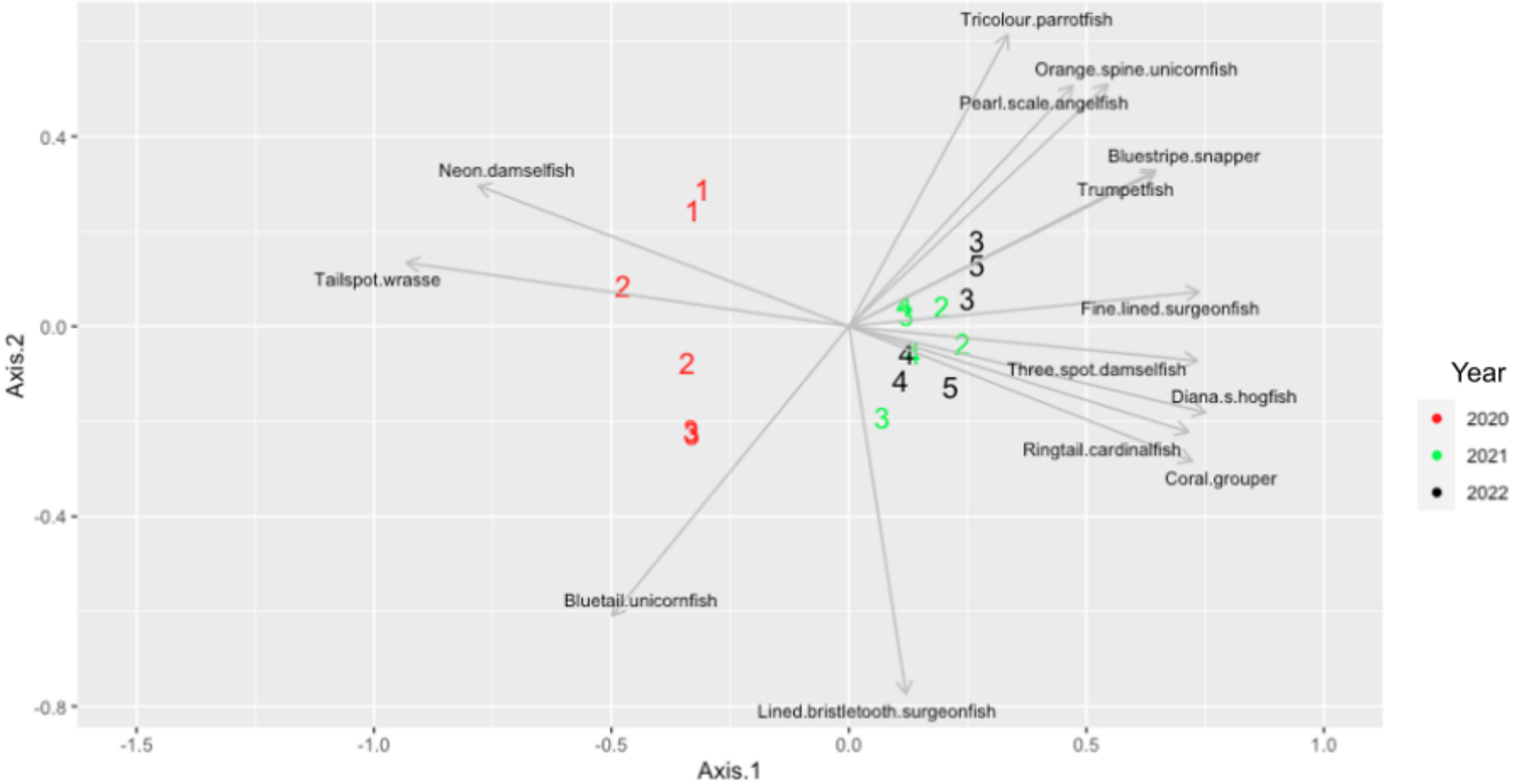

Figure 3. 5: Principal Coordinate Analysis (PCA) plot highlighting the differences in mobile communities between artificial reefs of different ages over a 3 year sampling period (2020, 2021, 2022), with arrows based on r > .45 and p <0.001. *Note: the numbers represented the specific age of the artificial reef site at the time it was surveyed (explained further in table 3.1).*

Figure 3.5 showed that AR mobile communities in 2021 (green) and 2022 (black) were substantially different from the first surveying year (2020; red), with few overlaps between years and several species which were unique to the AR in the first year (notably the neon damselfish and tailspot wrasse). It was also highlighted that the age of the ARs in 2020 made a substantial difference in determining its associated mobile communities, as highlighted by difference in communities (distance apart) between the structures of 1 – 3 years in 2020 . However, as the ARs became older over time (in 2021 and 2022; green and black respectively) , artificial reefs of all ages (2 – 5 years) were shown to overlap, highlighting that their communities were similar, regardless of the age of AR studied.

Figure 3.6 showed that over the three study periods, fish communities on the AR and sand habitat, as well as on the AR and CR became more similar. It also showed that fish communities on the CR and sand became less similar.

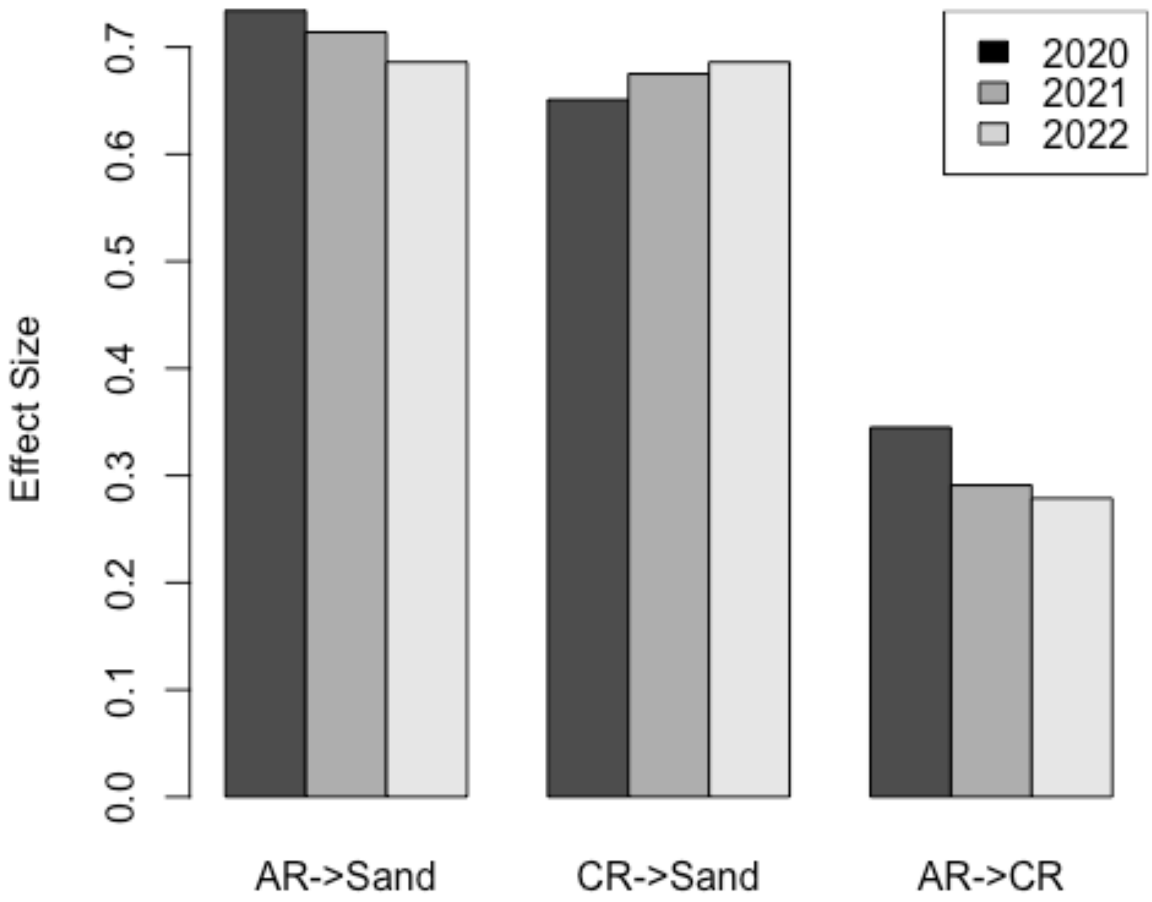

Figure 3. 6: Effect size of difference in fish community structure over a 3 year sampling period (2020, 2021, 2022).

Figure 3.7 highlighted the ‘risky’ foraging behaviours of neon damselfish, which was circled in red and was shown to forage for plankton relatively high up in the water column with little protection from the AR structures. This species was which was present in large populations on the AR in year one, medium-low populations in year two and absent in year three (table 3.2). In contrast, the year 3 screenshot highlighted that bicolor chromis damselfish, which appeared to have a ‘safer’ foraging behaviour that involved staying closer to the protection offered from the AR. In contrast, there were some species which were observed to have the opposite pattern, such as the bicolor chromis damselfish, which were shown to be absent on the AR in year one, have medium-sized populations in year two and large populations in year three (table 3.2).

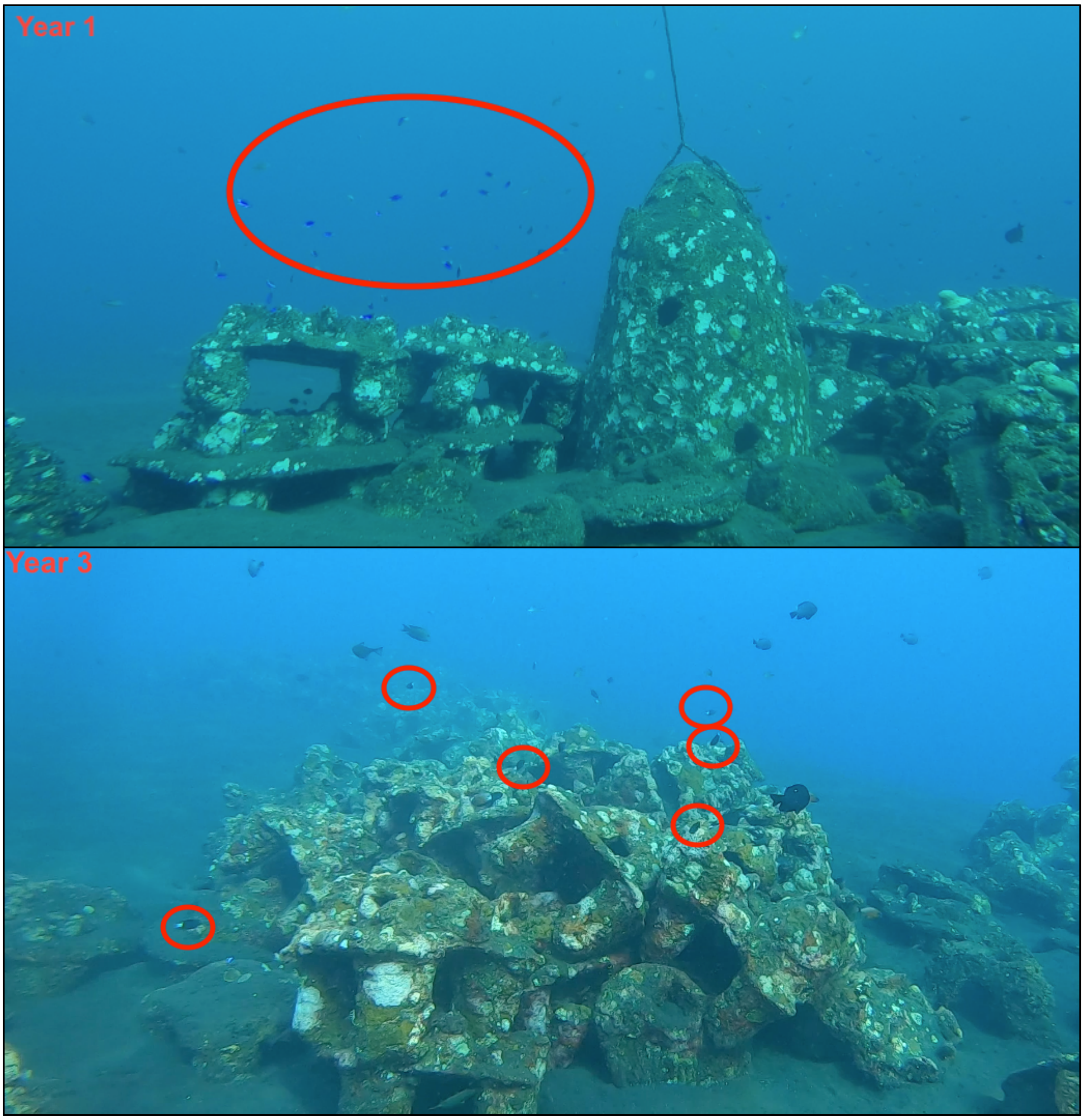

Figure 3. 7: Remote Underwater Video screenshots highlighting the difference in foraging behaviours of two damselfish species on the artificial reef in years one and three.

Discussion

Summary of key findings:

1 - Changes in artificial reef fish communities over time:

AR mobile communities in 2021 and 2022 were substantially different from those in 2020, with the age of the ARs in 2020 making a substantial difference in determining its associated mobile communities. However, ARs in the second and third year (2021 and 2022), were shown to be similar highlighting that specific AR age was not an important factor in determining fish communities.

2 - Convergence and divergence of fish communities between habitats over time:

Over time, we found that communities on the AR and sand habitat, as well as on the AR and CR became more similar. We also found that over time, communities on the CR and sand habitat became less similar. By the final year of sampling, the AR and CR were shown to be the same in terms of NoS and displayed some similarities in terms of community structure. Despite this, two-way PERMANOVA showed that fish community structure between habitat types were still significantly different over all sampling years.

1. Changes in artificial reef communities over time

Our research found a significant difference in AR fish communities between 2020 - 2021, but not between 2021 – 2022, and showed the greatest increase in NoS between years 2020 – 2021. AR fish communities are known to increase in complexity over time (Scarcella et al. 2015; Folpp et al. 2020; Hammond et al. 2020), being shown to rapidly increase in complexity after their initial deployment, and then beginning to stabilise over time (Bohnsack et al. 1994; Leitao et al. 2008; Lowry et al. 2014). In terms of the results from our study, it is possible that between 2020 – 2021, the ARs were experiencing an initial rapid increase in colonisation of fish communities (likely explaining the significant difference between the two), which had begun to stabilise between 2021-2022 (likely explaining the non-significant difference between the two). Medium to long term (3 years or more), robust data sets on AR fish communities are rare (Reis et al. 2021b), and from the limited studies that do assess the stabilisation of AR fish communities over time, there appears to be a range of results. For example, AR communities in a study by Mills et al. (2017) experienced optimal colonisation after 1 year of deployment, whereas Paxton et al. (2018) showed that fish community composition on a newly deployed AR converged with that of a nearby 20 year old AR within five months. Furthermore, Thanner et al. (2006) highlighted that fish communities had not reached stability after 5 years, and an 8 year study (2001-2009) by (Kojansow et al. 2013) highlighted that colonisation rate of coral reef fish on ReefBall® units peaked between then 6th – 7th year.

Various factors affect the time at which it takes for AR fish communities to become stable, such as structural complexity/design of the structures (Boakes et al. 2022a), deployment location (especially in terms of connectivity and proximity to natural reefs (Ambrose and Anderson 1990; Dempster 2005) and supply of juvenile fish (Mellin and Ponton 2009; Lowry et al. 2014). Therefore, direct comparisons between studies of different ARs is challenging, given that each AR site differs in terms of these external factors. It is worth noting that many of the medium to long-term studies researching AR diversity are based in temperate regions such as southern Australia (Folpp et al. 2013, Lowry et al. 2014, Mills et al. 2017, Hammond et al. 2020), which are likely less biodiverse than a tropical system in Indonesia. It may take longer for communities to stabilise in a system of high biodiversity than that of a low one, as more time is required for all species (including the slow recruitment, specialist species) to colonise it *(personal communication with AIMS scientist David Lennon)*. Therefore, this study cannot make direct comparisons with many of the others that have already researched this.

Our results highlighted that in the first year of surveying (2020), the age of the AR structures played a substantial role in determining their associated mobile communities, with a substantial difference in communities on ARs of 1 year to those on ARs of 3 years. However, as the AR structures became older (in 2021 and 2022), their individual ages became less important, with similar communities shown on ARs of 3 years to those on ARs of 5 years. This suggested that once fish communities start to become stable, an AR may be considered as "one reef', instead of many different reefs of different ages. These findings were discussed with Jerry Kojansow, a marine biologist conducting a long term study on fish assemblages on Reef Ball © ARs in Sulawesi, Indonesia. Through personal communication, it was found that Kojansow had noticed a similar trend. He commented that this may be because in the early years after deployment, ARs attract herbivorous fish which graze on the algae that has just grown on the surface of the structure. As benthic communities begin to get more complex after several years of deployment (notably through recruitment of hard corals which replace the initial algal growth), it is expected that fish communities will begin to stabilise. This provides a potential explanation as to why the individual ages of the ARs became less important in determining their fish assemblages, after several years of deployment.

Additionally, by analysing specific changes in fish communities of different trophic levels over time, we can begin to understand how communities may affect ecosystem functioning of ARs (Boakes et al. 2023b). One of several examples of low trophic level changes to the ARs in this study included the neon damselfish, which was present in large populations on the AR in year one, medium-low populations in year two and absent in year three. In contrast, there were some species which appeared to show the opposite pattern, such as the bicolor chromis damselfish, which were absent on the AR in year one, had medium populations in year two and large populations in year three. Research has highlighted the importance of predation in terms of structuring reef communities (Talbot et al. 1978; Hixon and Beets 1993), and Leitão et al (2008) showed that predators on ARs play an important role in controlling populations of low trophic level and/or juvenile species, like damselfish. The individual behaviours of different species often influences their chance of predation (Lowry et al. 2014), with some reef fish, such as the Eastern striped grunter, being associated with higher mortality rates due to risk-taking behaviours (Biro and Booth 2009). Our results showed that neon damselfish displayed a 'risky' foraging behaviour higher up in the water column, whereas the bicolor chromis had a safer foraging behaviour and stayed closer to the protection of the structures. The difference in foraging behaviours between these two damselfish species likely influenced their predation rates and provided an additional potential explanation for the changes in fish communities over time.

Our results also showed changes in communities of medium sized predatory fish on the ARs during the 3 year time series, including multiple species of grouper and snapper, which exhibited similar patterns to the bicolour chromis, with low population numbers in year one, medium in year two and large in year three. Snapper and grouper have been described as high-level, meso-predators (Frisch et al. 2016) and are known to feed on cephalopods, crustaceans and small fish (St John 1999; Szedlmayer and Lee 2004). The findings of this research is supported by other studies, showing that ARs provide important habitats to predatory fish (Herrera et al. 2002, Paxton, Newton, et al. 2020), which are thought to be attracted to the foraging opportunities associated with ARs, rather than the habitat complexity offered by the structures (Edwards and Smith 2005). Additionally, most of the predators present on the ARs in this study (including the multiple species of grouper and snapper) were resident predators (demersal species more commonly associated with the reef structure or seafloor (Paxton, Newton, et al. 2020)), as opposed to transient predators (such as jack or trevally). Residents like these exhibit high degrees of site fidelity and residence time (Dance et al. 2011; Topping and Szedlmayer 2011), and are thought to colonise ARs slowly because they are less mobile and spend less time travelling between habitats, especially compared to transient predators (Paxton, Newton, et al. 2020). This provides a potential explanation for the gradual development of snapper and grouper assemblages onto the AR in this study.

It must also be noted that Vaughan et al. (2021) highlighted that reefs display substantial seasonal variation in fish assemblages, with overall fish abundance being notably higher in warmer summer months. Other research has also highlighted the importance of seasonal variation in driving differences in fish community composition on tropical ARs, with seasonal aspects of reproduction (e.g. spawning) and the availability of food resources potentially dictating the migration of fish to and from the reef (Paxton et al. 2019) All data in this study was collected between the months of July to September each year, which are on average, the three coldest months in terms of water temperature in Bali (data from SeaTemperature.org (2022)), and thus has not accounted for the potential seasonal variation in fish communities. Research on this topic is very limited and more related studies would help increase our understanding the potentially important role of seasonal variation in the development of AR communities.

2. Convergence and divergence of community structure between habitats over time

Our results showed that over the 3 year sampling period, AR and CR communities were becoming more similar to each other, and by the final sampling period (2022), they had the same average NoS. These findings were supported by other studies, such as Carr and Hixon (1997) and Paxton et al. (2020), who highlighted the similarities between ARs and natural reefs in terms of biodiversity. Over time, it is likely that the AR and CR communities will further converge, although a longer-term sampling period is needed to confirm this. Additionally, AR diversity and/or abundance can over time, become higher on ARs than nearby natural reefs in cases where they offer a greater availability of sheltering habitat (Reed et al. 2006, Arena et al. 2007, Hackradt et al. 2011, Paxton et al. 2017). Despite this, our findings showed that each year, AR community structure was significantly different to the CR, with multiple 'specialist' species driving these differences (for example, trumpetfish on the AR, and several wrasse species on the CR). Most studies on this topic have highlighted that although fish assemblages on ARs can be abundant and biodiverse, they rarely mimic natural communities, as shown by multiple long-term studies, such as Thanner et al. (2006), Folpp et al. (2013), Lowry et al. (2014) and Becker et al. (2017). These studies concluded that mobile communities on ARs often remain distinct from nearby natural reefs due to differences in material, vertical relief, proximity to natural recruitment resources, availability of cryptic habitat and habitat complexity.

When comparing the AR to the nearby SF habitat, our results showed that each year, the AR had significantly different communities and a substantially higher NoS. It can therefore be highlighted that the ARs in this study generated near immediate increases in fish (in terms of biodiversity and abundance), as supported by the conclusions of most studies on ARs (e.g. Bohnsack et al. (1994), Leitao et al. (2008), Lowry et al. (2014), Paxton et al. (2020)). What was less certain was the extent to which the ARs in this study supported production of new biodiversity, instead of merely attracting it from nearby natural reefs (known as the 'attraction vs production debate', as discussed by (Pickering and Whitmarsh 1997). This debate remains topical in current AR literature (Kirkbride-Smith et al. 2016; Roa-Ureta et al. 2019), although most research generally agrees that well-designed and suitably-located ARs (which have an unrestricted supply of larvae) do increase biomass production (Cresson et al. 2014; Folpp et al. 2020). During data collection period, it was noted that there was a high proportion of juveniles (such as snapper, trevally and parrotfish), as well as several species of damselfish which were observed laying eggs on the structures, suggesting that the AR may have been supporting production of new biodiversity. Folpp et al. (2020) showed that initial colonisation of fish ARs is likely due to attraction of fish from other habitats, but over time ARs can lead to increased production, and can even increase abundance of fish on other non-artificial nearby habitats. Our results highlighted that the AR and sand habitat communities became more similar over the sampling period, which based on the findings of Folpp et al. (2020), may be due to the AR increasing complexity of the communities on the nearby sand habitat. This assumption would explain why the average NoS in the sand habitat increased slightly each year. If the AR was indeed increasing complexity of communities on the nearby sand habitat, this may also provide an explanation as to why the CR and sand communities became more different over the study period.

It must also be noted that research by Hijbers (2015) showed that some fish frequently move between multiple habitat types, including between sand flats and coral reefs at different times for different purposes. Our results showed that multiple species present were present across the CR, AR and SF, likely because each habitat was utilised by fish different purposes. All habitats were approximately 250m apart from each other, thus it is possible fish could be moving between these all three throughout the RUV recordings, which could confound the results of this study. However, significant differences in fish communities were highlighted between the habitat types at different multiple times during the 3 year monitoring period (especially between the SF and other habitat types). It can therefore be said that any non-independence of species at a site as a consequence of movement from another habitat type, had little influence and did not weaken the significance of the findings. This study used non-baited-RUV, as opposed to baited-RUV in an attempt to prevent an exaggerated movement of species attracted by bait (as discussed by Bernard and Götz (2012)). As such, any non-independence of sites as a result of species movement between site the CR, AR and SF can be dismissed as a possible confounding factor in this study. It is however possible that there is as some degree of similarity of species between habitat types may due to the movement between spatially close habitats.

Conclusion

We aimed to assess how fish communities changed on an artificial reef over a 3 year time scale, and then compare these communities to those on nearby natural reefs. Our first major finding was that AR mobile communities in 2021 and 2022 were substantially different from those in 2020, with the age of the ARs in 2020 making a substantial difference in determining their associated mobile communities. However, ARs in the second and third year (2021 and 2022), were similar, highlighting that specific AR age was not an important factor in determining fish communities. It is possible that between 2020 – 2021, the ARs were experiencing an initial rapid increase in colonisation of fish communities, which had started to stabilise between 2021-2022 due various predatory and competitive processes. We propose that over time, the individual ages of ARs became less important, and as their communities begin to stabilise, ARs may be considered to function as 'one reef' instead of clusters of different aged ARs.

Our second major finding was that over the 3 year sampling period, AR and CR communities became more similar, and by the final sampling period, they had the same average NoS. However, even after this final year of sampling, the AR still displayed small (yet distinct and significant) differences to the CR, with multiple 'specialist' species driving these differences. Over time, as benthic communities become more similar, it is likely that the AR and CR communities will further converge, although a longer-term sampling period is needed to confirm this.

This work is one of one of few studies which utilise a long-term dataset to evaluate the potential of ARs in Indonesia to provide localised increases in fish abundance and biodiversity. The results of this study may be useful for communities particularly reliant on the ecosystem services provided coral reefs, especially those that have experienced a decline in the health of their natural reefs, and not only apply to reefs in Indonesia, but internationally too. Future work is needed to assess that if, alongside supporting fish abundance and biodiversity, ARs can support the same functional processes as their natural counterparts.

References

Allen, G. R., 2008. Conservation hotspots of biodiversity and endemism for Indo-Pacific coral reef fishes. *Aquatic conservation marine and freshwater ecosystems*.

Allen, G. R., Steene, R., Humann, P. and Deloach, N., 2003. *Reef fish identification: tropical Pacific*. New World Publications Incorporated.

Ambrose, R. F. and Anderson, T. W., 1990. Influence of an artificial reef on the surrounding infaunal community. *Marine Biology*, 107 (1), 41–52.

Anderson, M. J., 2001. A new method for non-parametric multivariate analysis of variance. *Austral ecology*, 26 (1), 32–46.

Andrello, M., Darling, E. S., Wenger, A., Suárez-Castro, A. F., Gelfand, S. and Ahmadia, G. N., 2021. A global map of human pressures on tropical coral reefs. *Conservation Letters*, e12858.

Arena, P. T., Jordan, L. K. B. and Spieler, R. E., 2007. Fish assemblages on sunken vessels and natural reefs in southeast Florida, USA. *In*: *Biodiversity in enclosed seas and artificial marine habitats*. Springer, 157–171.

Argeswara, J., Hendrawan, I. G., Dharma, I. G. B. S. and Germanov, E., 2021. What's in the soup? Visual characterization and polymer analysis of microplastics from an Indonesian manta ray feeding ground. *Marine Pollution Bulletin*, 168, 112427.

Baine, M., 2001. Artificial reefs: a review of their design, application, management and performance. *Ocean & coastal management*, 44 (3–4), 241–259.

Baria-Rodriguez, M. V., dela Cruz, D. W., Dizon, R. M., Yap, H. T. and Villanueva, R. D., 2019. Performance and cost-effectiveness of sexually produced Acropora granulosa juveniles compared with asexually generated coral fragments in restoring degraded reef areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 29 (6), 891–900.

Barton, J. A., Willis, B. L. and Hutson, K. S., 2017. Coral propagation: a review of techniques for ornamental trade and reef restoration. *Reviews in Aquaculture*, 9 (3), 238–256.

Bates, D., Mächler, M., Bolker, B. and Walker, S., 2014. Fitting linear mixed-effects models using lme4. *arXiv preprint arXiv:1406.5823*.

Becker, A., Taylor, M. D. and Lowry, M. B., 2017. Monitoring of reef associated and pelagic fish communities on Australia's first purpose built offshore artificial reef. *ICES Journal of Marine Science*, 74 (1), 277–285.

Bell, J. D., Ratner, B. D., Stobutzki, I. and Oliver, J., 2006. Addressing the coral reef crisis in developing countries.

Ben-Romdhane, H., Jabado, R. W., Grandcourt, E. M., Perry, R. J. O., Al Blooshi, A. Y., Marpu, P. R., Ouarda, T. B. M. J. and Ghedira, H., 2020. Coral reefs of Abu Dhabi, United Arab Emirates: Analysis of management approaches in light of international best practices and a changing climate. *Frontiers in Marine Science*, 7, 541.

Bernard, A. T. F. and Götz, A., 2012. Bait increases the precision in count data from remote underwater video for most subtidal reef fish in the warm-temperate Agulhas bioregion. *Marine Ecology Progress Series*, 471, 235–252.

Biro, P. A. and Booth, D. J., 2009. Extreme boldness precedes starvation mortality in six-lined trumpeter (Pelates sexlineatus). *Hydrobiologia*, 635 (1), 395–398.

Boakes, Z., Hall, A. E., Ampou, E. E., Jones, G. C. A., Suryaputra, I. G. N. A., Mahyuni, L. P., Prasetyo, R. and Stafford, R., 2022. Coral reef conservation in Bali in light of international best practice, a literature review. *Journal for Nature Conservation*, 126190.

Boakes, Z., Hall, A., Jones, G., Prasetyo, R., Stafford, R. and Yahya, Y., 2022. Artificial coral reefs as a localised approach to increase fish biodiversity and abundance along the North Bali coastline. *AIMS Geosciences*, 8 (2), 303–325.

Boakes, Z., Mahyuni, L. P., Hall, A. E., Cvitanovic, M. and Stafford, R., 2023. Can Coral Reef Restoration Programmes Facilitate Changes in Environmental Attitudes? A Case Study on a Rural Fisher Community in North Bali, Indonesia. *Human Ecology*, 1–15.

Boakes, Z., Suryaputra, I. G. N. A., Hall, A. E., Franklin, D. J. and Stafford, R., 2023. Nutrient dynamics, carbon storage and community composition on artificial and natural reefs in Bali, Indonesia. *Marine Biology*, 170 (10), 130.

Bohnsack, J. A., 1989. Are high densities of fishes at artificial reefs the result of habitat limitation or behavioral preference? *Bulletin of marine science*, 44 (2), 631–645.

Bohnsack, J. A., Harper, D. E., McClellan, D. B. and Hulsbeck, M., 1994. Effects of reef size on colonization and assemblage structure of fishes at artificial reefs off southeastern Florida, USA. *Bulletin of Marine Science*, 55 (2–3), 796–823.

Boström-Einarsson, L., Babcock, R. C., Bayraktarov, E., Ceccarelli, D., Cook, N., Ferse, S. C. A., Hancock, B., Harrison, P., Hein, M. and Shaver, E., 2020. Coral restoration–A systematic review of current methods, successes, failures and future directions. *PloS one*, 15 (1), e0226631.

Brander, L. M., Van Beukering, P. and Cesar, H. S. J., 2007. The recreational value of coral reefs: a meta-analysis. *Ecological Economics*, 63 (1), 209–218.

Burke, L., Reytar, K., Spalding, M. and Perry, A., 2012. *Reefs at risk revisited in the Coral Triangle*.

Camp, E. F., Nitschke, M. R., Rodolfo-Metalpa, R., Houlbreque, F., Gardner, S. G., Smith, D. J., Zampighi, M. and Suggett, D. J., 2017. Reef-building corals thrive within hot-acidified and deoxygenated waters. *Scientific reports*, 7 (1), 1–9.

Carr, M. H. and Hixon, M. A., 1997. Artificial reefs: the importance of comparisons with natural reefs. *Fisheries*, 22 (4), 28–33.

Christie, P. and White, A. T., 2007. Best practices for improved governance of coral reef marine protected areas. *Coral Reefs*, 26 (4), 1047–1056.

Claar, D. C., Szostek, L., McDevitt-Irwin, J. M., Schanze, J. J. and Baum, J. K., 2018. Global patterns and impacts of El Niño events on coral reefs: A meta-analysis. *PloS one*, 13 (2), e0190957.

Cleves, P. A., Strader, M. E., Bay, L. K., Pringle, J. R. and Matz, M. V, 2018. CRISPR/Cas9-mediated genome editing in a reef-building coral. *Proceedings of the National Academy of Sciences*, 115 (20), 5235–5240.

Costanza, R., De Groot, R., Sutton, P., Van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S. and Turner, R. K., 2014. Changes in the global value of ecosystem services. *Global environmental change*, 26, 152–158.

Cresson, P., Ruitton, S. and Harmelin-Vivien, M., 2014. Artificial reefs do increase secondary biomass production: mechanisms evidenced by stable isotopes. *Marine Ecology Progress Series*, 509, 15–26.

Dance, M. A., Patterson III, W. F. and Addis, D. T., 2011. Fish community and trophic structure at artificial reef sites in the northeastern Gulf of Mexico. *Bulletin of Marine Science*, 87 (3), 301–324.

Dempster, T., 2005. Temporal variability of pelagic fish assemblages around fish aggregation devices: biological and physical influences. *Journal of Fish biology*, 66 (5), 1237–1260.

Doherty, O., Milner, C., Dustan, P., Campbell, S., Pardede, S., Kartawijaya, T. and Alling, A., 2013. Report on Menjangan Island's coral reef: A Bali Barat National Park marine protected area. *Atoll Research Bulletin*, 19 (599), 1–18.

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., Barrett, N. S., Becerro, M. A., Bernard, A. T. F. and Berkhout, J., 2014. Global conservation outcomes depend on marine protected areas with five key features. *Nature*, 506 (7487), 216–220.

Edwards, R. A. and Smith, S. D. A., 2005. Subtidal assemblages associated with a geotextile reef in south-east Queensland, Australia. *Marine and Freshwater Research*, 56 (2), 133–142.

Endo, S., Prasetyo, R., Nishihira, M. and Onaka, S., 2008. Experimental coral transplantation in the moat of Kuta beach, Bali Island, Indonesia. *In*: *PROCEEDINGS OF COASTAL ENGINEERING, JSCE*. Japan Society of Civil Engineers, 1121–1125.

Folpp, H., Lowry, M., Gregson, M. and Suthers, I. M., 2013. Fish assemblages on estuarine artificial reefs: natural rocky-reef mimics or discrete assemblages? *PLoS One*, 8 (6), e63505.

Folpp, H. R., Schilling, H. T., Clark, G. F., Lowry, M. B., Maslen, B., Gregson, M. and Suthers, I. M., 2020. Artificial reefs increase fish abundance in habitat-limited estuaries. *Journal of Applied Ecology*, 57 (9), 1752–1761.

Frey, J. and Berkes, F., 2014. Can partnerships and community-based conservation reverse the decline of coral reef social-ecological systems? *International Journal of the Commons*, 8 (1).
Frisch, A. J., Ireland, M., Rizzari, J. R., Lönnstedt, O. M., Magnenat, K. A., Mirbach, C. E. and Hobbs, J.-P. A., 2016. Reassessing the trophic role of reef sharks as apex predators on coral reefs. *Coral Reefs*, 35 (2), 459–472.
Froehlich, C. Y. M. and Kline, R. J., 2015. Using fish population metrics to compare the effects of artificial reef density. *PloS one*, 10 (9), e0139444.
Germanov, E. S., Marshall, A. D., Bejder, L., Fossi, M. C. and Loneragan, N. R., 2018. Microplastics: no small problem for filter-feeding megafauna. *Trends in ecology & evolution*, 33 (4), 227–232.
Gerungan, A. and Chia, K. W., 2020. Scuba diving operators' perspective of scuba diving tourism business in Nusa Penida, Indonesia. *Journal of Outdoor Recreation and Tourism*, 31, 100328.
Graham, N. A. J., Jennings, S., MacNeil, M. A., Mouillot, D. and Wilson, S. K., 2015. Predicting climate-driven regime shifts versus rebound potential in coral reefs. *Nature*, 518 (7537), 94–97.
Graham, N. A. J., Robinson, J. P. W., Smith, S. E., Govinden, R., Gendron, G. and Wilson, S. K., 2020. Changing role of coral reef marine reserves in a warming climate. *Nature Communications*, 11 (1), 1–8.
Hackradt, C. W., Félix-Hackradt, F. C. and García-Charton, J. A., 2011. Influence of habitat structure on fish assemblage of an artificial reef in southern Brazil. *Marine environmental research*, 72 (5), 235–247.
Hall, A. E., Herbert, R. J. H. and Stafford, R., 2021. Temporal and spatial variation in adult and juvenile mobile fauna associated with natural and artificial coastal habitats. *Marine Biology*, 168 (2), 1–13.
Hammond, M., Bond, T., Prince, J., Hovey, R. K. and McLean, D. L., 2020. An assessment of change to fish and benthic communities following installation of an artificial reef. *Regional Studies in Marine Science*, 39, 101408.
Herbert, R. J. H., Collins, K., Mallinson, J., Hall, A. E., Pegg, J., Ross, K., Clarke, L. and Clements, T., 2017. Epibenthic and mobile species colonisation of a geotextile artificial surf reef on the south coast of England. *PloS one*, 12 (9), e0184100.
Heron, S. F., Eakin, C. M., Douvere, F., Anderson, K. L., Day, J. C., Geiger, E., Hoegh-Guldberg, O., Van Hooidonk, R., Hughes, T. and Marshall, P., 2017. Impacts of climate change on World Heritage coral reefs: a first global scientific assessment.
Herrera, R., Espino, F., Garrido, M. and Haroun, R. J., 2002. Observations on fish colonization and predation on two artificial reefs in the Canary Islands. *ICES Journal of Marine Science*, 59 (suppl), S69–S73.
Hixon, M. A. and Beets, J. P., 1993. Predation, prey refuges, and the structure of coral-reef fish assemblages. *Ecological Monographs*, 63 (1), 77–101.
Van Hooidonk, R., Maynard, J., Tamelander, J., Gove, J., Ahmadia, G., Raymundo, L., Williams, G., Heron, S. F. and Planes, S., 2016. Local-scale projections of coral reef futures and implications of the Paris Agreement. *Scientific reports*, 6 (1), 1–8.
Howlett, S. J., Stafford, R., Waller, M., Antha, S. and Mason-Parker, C., 2016. Linking protection with the distribution of grouper and habitat quality in Seychelles. *Journal of Marine Biology*, 2016.
Hughes, T. P., Kerry, J. T., Baird, A. H., Connolly, S. R., Dietzel, A., Eakin, C. M., Heron, S. F., Hoey, A. S., Hoogenboom, M. O. and Liu, G., 2018. Global warming transforms coral reef assemblages. *Nature*, 556 (7702), 492–496.
Huijbers, C. M., Nagelkerken, I. and Layman, C. A., 2015. Fish movement from nursery bays to coral reefs: a matter of size? *Hydrobiologia*, 750 (1), 89–101.
ICRG, 2021. *Coral Reef Restoration Successfully Absorbs 10 Thousand Workers* [online]. Available from: https://www.icrg.id/en/news/detail/coral-reef-restoration-successfully-absorbs-10-thousand-workers.
Kenyon, T. M., Doropoulos, C., Dove, S., Webb, G. E., Newman, S. P., Sim, C. W. H., Arzan, M. and Mumby, P. J., 2020. The effects of rubble mobilisation on coral fragment survival, partial mortality and growth. *Journal of Experimental Marine Biology and Ecology*, 533, 151467.

Kirkbride-Smith, A. E., Wheeler, P. M. and Johnson, M. L., 2016. Artificial reefs and marine protected areas: a study in willingness to pay to access Folkestone Marine Reserve, Barbados, West Indies. *PeerJ*, 4, e2175.

Kleypas, J. A. and Eakin, C. M., 2007. Scientists' perceptions of threats to coral reefs: results of a survey of coral reef researchers. *Bulletin of Marine Science*, 80 (2), 419–436.

KOJANSOW, J., SOMPIE, D., EMOR, D. and Rondonuwu, A. B., 2013. Fish settlement on reefballs artificial reef and natural coral reef at Buyat Bay and surrounding areas, North Sulawesi, Indonesia. *Galaxea, Journal of Coral Reef Studies*, 15 (Supplement), 229–237.

Lamont, T. A. C., Razak, T. B., Djohani, R., Janetski, N., Rapi, S., Mars, F. and Smith, D. J., 2022. Multi-dimensional approaches to scaling up coral reef restoration. *Marine Policy*, 143, 105199.

Langlois, T., Goetze, J., Bond, T., Monk, J., Abesamis, R. A., Asher, J., Barrett, N., Bernard, A. T. F., Bouchet, P. J. and Birt, M. J., 2020. A field and video annotation guide for baited remote underwater stereo-video surveys of demersal fish assemblages. *Methods in Ecology and Evolution*, 11 (11), 1401–1409.

Leitao, F., Santos, M. N., Erzini, K. and Monteiro, C. C., 2008. Fish assemblages and rapid colonization after enlargement of an artificial reef off the Algarve coast (Southern Portugal). *Marine Ecology*, 29 (4), 435–448.

Lemoine, H. R., Paxton, A. B., Anisfeld, S. C., Rosemond, R. C. and Peterson, C. H., 2019. Selecting the optimal artificial reefs to achieve fish habitat enhancement goals. *Biological Conservation*, 238, 108200.

Lenth, R. V, 2021. Estimated marginal means, aka least-squares means [R Package Emmeans Version 1.6. 0]. *Comprehensive R Archive Network (CRAN)*.

Lesser, M. P., 2011. Coral bleaching: causes and mechanisms. *In*: *Coral reefs: an ecosystem in transition*. Springer, 405–419.

Lowry, M. B., Glasby, T. M., Boys, C. A., Folpp, H., Suthers, I. and Gregson, M., 2014. Response of fish communities to the deployment of estuarine artificial reefs for fisheries enhancement. *Fisheries Management and Ecology*, 21 (1), 42–56.

Marinaro, J.-Y., 1995. Artificial reefs in the French Mediterranean: A critical assessment of previous experiments and a proposition in favour of a new reef-planning policy. *Biol. Mar. Mediterr.*, 2 (1), 65–76.

Mellin, C. and Ponton, D., 2009. Assemblages of reef fish settling on artificial substrates: effect of ambient habitat over two temporal scales. *Marine and freshwater research*, 60 (12), 1285–1297.

Mercader, M., Mercière, A., Saragoni, G., Cheminée, A., Crec'hriou, R., Pastor, J., Rider, M., Dubas, R., Lecaillon, G. and Boissery, P., 2017. Small artificial habitats to enhance the nursery function for juvenile fish in a large commercial port of the Mediterranean. *Ecological Engineering*, 105, 78–86.

Mills, K. A., Hamer, P. A. and Quinn, G. P., 2017. Artificial reefs create distinct fish assemblages. *Marine Ecology Progress Series*, 585, 155–173.

Morikawa, M. K. and Palumbi, S. R., 2019. Using naturally occurring climate resilient corals to construct bleaching-resistant nurseries. *Proceedings of the National Academy of Sciences*, 116 (21), 10586–10591.

Onaka, S., Prasetyo, R., Endo, S. and Yoshii, I., 2013. Large-scale coral transplantation in artificial substrates at a shallow lagoon in Kuta Beach, Bali, Indonesia. *Galaxea, Journal of Coral Reef Studies*, 15 (Supplement), 336–342.

Paxton, A. B., Newton, E. A., Adler, A. M., Van Hoeck, R. V, Iversen Jr, E. S., Taylor, J. C., Peterson, C. H. and Silliman, B. R., 2020. Artificial habitats host elevated densities of large reef-associated predators. *PloS one*, 15 (9), e0237374.

Paxton, A. B., Peterson, C. H., Taylor, J. C., Adler, A. M., Pickering, E. A. and Silliman, B. R., 2019. Artificial reefs facilitate tropical fish at their range edge. Communications Biology, Vol. 2, article. 168.

Paxton, A. B., Pickering, E. A., Adler, A. M., Taylor, J. C. and Peterson, C. H., 2017. Flat and complex temperate reefs provide similar support for fish: Evidence for a unimodal species-habitat relationship. *PLoS One*, 12 (9), e0183906.

Paxton, A. B., Revels, L. W., Rosemond, R. C., Van Hoeck, R. V, Lemoine, H. R., Taylor, J. C. and Peterson, C. H., 2018. Convergence of fish community structure between a newly deployed and an established artificial reef along a five-month trajectory. *Ecological Engineering*, 123, 185–192.

Paxton, A. B., Shertzer, K. W., Bacheler, N. M., Kellison, G. T., Riley, K. L. and Taylor, J. C., 2020. Meta-analysis reveals artificial reefs can be effective tools for fish community enhancement but are not one-size-fits-all. *Frontiers in Marine Science*, 7, 282.

Pears, R. J. and Williams, D. M., 2005. *Potential effects of artificial reefs on the Great Barrier Reef: background paper*. CRC Reef Research Centre Townsville,, Australia.

Perkol-Finkel, S., Shashar, N. and Benayahu, Y., 2006. Can artificial reefs mimic natural reef communities? The roles of structural features and age. *Marine environmental research*, 61 (2), 121–135.

Pickering, H. and Whitmarsh, D., 1997. Artificial reefs and fisheries exploitation: a review of the 'attraction versus production'debate, the influence of design and its significance for policy. *Fisheries research*, 31 (1–2), 39–59.

Pörtner, H.-O., Karl, D. M., Boyd, P. W., Cheung, W., Lluch-Cota, S. E., Nojiri, Y., Schmidt, D. N., Zavialov, P. O., Alheit, J. and Aristegui, J., 2014. Ocean systems. *In*: *Climate change 2014: impacts, adaptation, and vulnerability. Part A: global and sectoral aspects. contribution of working group II to the fifth assessment report of the intergovernmental panel on climate change*. Cambridge University Press, 411–484.

Prasetia, D., Supriharyono, M., Anggoro, S. and Sya'Rani, L., 2017. Coral Bleaching on Lembongan Island, Nusa Penida, Bali. *In*: . Atlantis Press.

Puspasari, R., Wiadnyana, N. N., Hartati, S. T., Rachmawati, R. and Yahya, Y., 2020. The Effectiveness of Artificial Reefs in Improving Ecosystem Health to Increase Coral Reef Resilience. *Jurnal Segara*, 16 (2), 115–126.

Reed, D. C., Schroeter, S. C., Huang, D., Anderson, T. W. and Ambrose, R. F., 2006. Quantitative assessment of different artificial reef designs in mitigating losses to kelp forest fishes. *Bulletin of Marine Science*, 78 (1), 133–150.

Reis, B., van der Linden, P., Pinto, I. S., Almada, E., Borges, M. T., Hall, A. E., Stafford, R., Herbert, R. J. H., Lobo-Arteaga, J. and Gaudêncio, M. J., 2021a. Artificial reefs in the North–East Atlantic area: Present situation, knowledge gaps and future perspectives. *Ocean & Coastal Management*, 213, 105854.

Reis, B., van der Linden, P., Pinto, I. S., Almada, E., Borges, M. T., Hall, A. E., Stafford, R., Herbert, R. J. H., Lobo-Arteaga, J. and Gaudêncio, M. J., 2021b. Artificial reefs in the North–East Atlantic area: Present situation, knowledge gaps and future perspectives. *Ocean & Coastal Management*, 213, 105854.

Roa-Ureta, R. H., Santos, M. N. and Leitão, F., 2019. Modelling long-term fisheries data to resolve the attraction versus production dilemma of artificial reefs. *Ecological Modelling*, 407, 108727.

Ruchimat, T., Basuki, R. and Welly, M., 2013. Nusa Penida Marine Protected Area (Mpa) Bali-Indonesia: Why Need to be Protected? *Transylvanian Review of Systematical and Ecological Research*, 15 (1), 193–202.

Scarcella, G., Grati, F., Bolognini, L., Domenichetti, F., Malaspina, S., Manoukian, S., Polidori, P., Spagnolo, A. and Fabi, G., 2015. Time-series analyses of fish abundance from an artificial reef and a reference area in the central-Adriatic Sea. *Journal of Applied Ichthyology*, 31, 74–85.

Schramm, K. D., Harvey, E. S., Goetze, J. S., Travers, M. J., Warnock, B. and Saunders, B. J., 2020. A comparison of stereo-BRUV, diver operated and remote stereo-video transects for assessing reef fish assemblages. *Journal of Experimental Marine Biology and Ecology*, 524, 151273.

SeaTemperature.org, 2022. *SeaTemperature.org Bali Average Sea Temperatures*.

Selgrath, J. C., Peterson, G. D., Thyresson, M., Nyström, M. and Gergel, S. E., 2017. Regime Shifts and Spatial Resilience in a Coral Reef Seascape. *In*: *Learning Landscape Ecology*. Springer, 301–322.

Spalding, M., Burke, L., Wood, S. A., Ashpole, J., Hutchison, J. and Zu Ermgassen, P., 2017. Mapping the global value and distribution of coral reef tourism. *Marine Policy*, 82, 104–113.

St John, J., 1999. Ontogenetic changes in the diet of the coral reef grouper Plectropomus leopardus (Serranidae): patterns in taxa, size and habitat of prey. *Marine Ecology Progress Series*, 180, 233–246.

Suparno, N., Currier, K., Milner, C., Alling, A. and Dustan, P., 2019. Ecological changes in the coral reef communities of Indonesia's Bali Barat National Park, 2011–2016. *Atoll Res Bull*, 620, 1–35.

Susiloningtyas, D., Handayani, T. and Amalia, A. N., 2018. The impact of coral reefs destruction and climate change in Nusa Dua and Nusa Penida, Bali, Indonesia. *In*: . IOP Publishing, 12054.

Syam, A. R., Edrus, I. N. and Hartati, S. T., 2017. Coral fish population changes in the surrounding artificial reefs of the Lebah coastal waters, Karangasem, Bali. *Indonesian Fisheries Research Journal*, 13 (2), 101–116.

Szedlmayer, S. T. and Lee, J. D., 2004. Diet shifts of juvenile red snapper (Lutjanus campechanus) with changes in habitat and fish size.

Talbot, F. H., Russell, B. C. and Anderson, G. R. V, 1978. Coral reef fish communities: unstable, high-diversity systems? *Ecological Monographs*, 48 (4), 425–440.

Thanner, S. E., McIntosh, T. L. and Blair, S. M., 2006. Development of benthic and fish assemblages on artificial reef materials compared to adjacent natural reef assemblages in Miami-Dade County, Florida. *Bulletin of marine Science*, 78 (1), 57–70.

Topping, D. T. and Szedlmayer, S. T., 2011. Site fidelity, residence time and movements of red snapper Lutjanus campechanus estimated with long-term acoustic monitoring. *Marine Ecology Progress Series*, 437, 183–200.

Turak, E. and DeVantier, L., 2011. Chapter 5: Biodiversity and Conservation Priorities of Reef-building Corals in Bali, Indonesia. *Bali Marine Rapid Assessment Program*, 78–130.

Turak, E. and Devantier, L., 2013. Biodiversity and conservation priorities of reef-building corals in Bali, Indonesia. *In*: *Bali Marine Rapid Assessment Program 2011*. BioOne, 78.

Tweedie, J. B., Cockburn, J. M. H. and Villard, P. V, 2023. The Potential Use of Remote Underwater Video (RUV) to Evaluate Small-Bodied Fish Assemblages. *Hydrobiology*, 2 (3), 507–520.

Vaughan, G. O., Shiels, H. A. and Burt, J. A., 2021. Seasonal variation in reef fish assemblages in the environmentally extreme southern Persian/Arabian Gulf. *Coral Reefs*, 40 (2), 405–416.

Whitmarsh, S. K., Fairweather, P. G. and Huveneers, C., 2017. What is Big BRUVver up to? Methods and uses of baited underwater video. *Reviews in Fish Biology and Fisheries*, 27 (1), 53–73.

Wicaksana, I. B. A., 2020. 2020 Bali's Coral Conservation: from pandemic challenge to Government coral garden project. *Bali Tourism Journal*, 4 (2), 35–39.

Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A. and Smith, G. M., 2009. *Mixed effects models and extensions in ecology with R*. Springer.

Chapter 4: Nutrient dynamics, carbon storage and community composition on artificial and natural reefs in Bali, Indonesia.

Objectives: To examine the links between ecosystem function (nutrient dynamics and storage) and community structure between three different habitat types (flat sand bed, artificial reef and coral reef), with the aim of assessing if artificial reefs can mimic natural reefs in terms of ecosystem function.

Contribution to new knowledge: A recently literature review by Vivier et al., (2021) highlighted that there is a limited amount of research which focuses on the ecosystem function of artificial reefs, and suggests that more studies should investigate the complex relationships between environmental parameters like nutrient dynamics and ecosystem structure. To the best of my knowledge, very few published studies exist which focus on AR nutrient cycling in tropical coral reef enviroments, and the results of these studies are generally inconclusive.

How this fits in the PhD: This chapter assesses the extent to which artificial reefs can mimic natural reefs in terms of ecosystem function (mapping onto objective 3). It fits directly with 'restoration' part of my PhD question, as ecosystem function play a vital role within coral reef biology and ecology. Ecosystem function (particularly with regards to carbon sequestration) is also considered an ecosystem service, and therefore also fits within the 'socioeconomic' aspect of my PhD too.

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Boakes, Z., Suryaputra, I. G. N. A., Hall, A. E., Franklin, D. J. and Stafford, R., 2023. Nutrient dynamics, carbon storage and community composition on artificial and natural reefs in Bali, Indonesia. Marine Biology, 170, 130. https://doi.org/10.1007/s00227-023-04283-4

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Abstract

Artificial reefs are now commonly used as a tool to restore degraded coral reefs and have a proven potential to enhance biodiversity. Despite this, there is currently a limited understanding of ecosystem functioning on artificial reefs, and how this compares to natural reefs. We used water sampling (bottom water sampling and pore water sampling), as well as surface sediment sampling and sediment traps, to examine the storage of total organic matter (as a measure of total organic carbon) and dynamics of dissolved inorganic nitrate, nitrite, phosphate and ammonium. These biogeochemical parameters were used as measures of ecosystem functioning, which were compared between an artificial reef and natural coral reef, as well as a degraded sand flat (as a control habitat), in Bali, Indonesia. We also linked the differences in these parameters to observable changes in the community structure of mobile, cryptobenthic and benthic organisms between habitat types. Our key findings showed: 1) There were no significant differences in inorganic nutrients between habitat types for bottom water samples, 2) Pore water phosphate concentrations were significantly higher on the artificial reef than on both other habitats, 3) Total organic matter content in sediments was significantly higher on the coral reef than both other habitat types, and 4) Total organic matter in sediment traps in May and September were higher on coral reefs than other habitats, but no differences were found in November. Overall, in terms of ecosystem functioning (specifically nutrient storage and dynamics), the artificial reef showed differences from the nearby degraded sand flat, and appeared to have some similarities with the coral reef. However, it was shown to not yet be fully functioning as the coral reef, which we hypothesise is due its relatively less complex benthic community and different fish community. We highlight the need for longer-term studies on artificial reef functioning, to assess if these habitats can replace the ecological function of coral reefs at a local level.

Key words: Artificial reef ecosystem functioning, coral reef functionality, total organic matter, total organic carbon, dissolved inorganic nutrients, biological community structure

Introduction

Artificial reefs

Artificial reefs (ARs) are man-made structures deployed within the marine environment thought to have been utilised since the 1600s as a tool used to attract fish for enhancing fish catch (Stone et al. 1979). Only since the rise of the environmental movement in the 1960s and 1970s, have artificial reefs gained attention for their potential in habitat restoration (Paxton et al. 2020). Research in the last decade has highlighted how ARs can quickly restore previously degraded and/or unproductive areas, through providing previously unavailable substrata and habitat complexity (Becker et al. 2017, Israel et al. 2017). The use of ARs as a habitat enhancement tool has been shown to be particularly successful when deployed in previously degraded tropical coral reefs (Lemoine et al. 2019, Paxton et al. 2020, Boakes et al. 2022a), especially in cases where natural recovery would be unlikely or slow (e.g. if regime shifts to algal states have already occurred (Graham et al. 2015, Kenyon et al. 2020)). Due to anthropogenic threats such as climate change induced bleaching, overfishing and pollution (Lesser 2011, Claar et al. 2018, Andrello et al. 2021), global coral reef health has substantially declined in recent decades (Heron et al. 2017, Hughes et al. 2018). ARs continue to be used as a tool to provide some degree of localised protection against this, and in certain cases, restore ecosystem services in tropical areas which have lost natural reefs entirely (Chen et al. 2013, Schulze et al. 2020, Boakes et al. 2022a).

ARs designed for habitat enhancement purposes may incorporate intentionally-built structural complexity (e.g. in the form of multiple hiding spaces and exits, high surface areas and hollow interior spaces such as caves or tunnels (Marinaro 1995, Lemoine et al. 2019)). This supports colonisation of mobile communities as spawning adults use the new substrata to lay their eggs, whilst juveniles are provided with shelter and protection, and utilise the AR as a nursery (Herbert et al. 2017). Artificial reefs may also be built with a rough and/or textured surface which allows the larvae of corals (and other benthos) to attach themselves, thus enhancing benthic recruitment (Bohnsack and Sutherland 1985, Harris 2009). When deployed in previously poor quality and/or degraded habitats, the new substrata and complexity provided by the ARs has been shown to consistently lead to increases in biomass and diversity of reef species (Godoy et al. 2002, Komyakova et al. 2019, Boakes et al 2022a). Whilst a great deal of research has already associated ARs with the restoration of biodiversity, it remains debated whether their functioning is comparable to natural reefs (Carr and Hixon 1997, Paxton et al. 2020), with currently a very limited amount of research on the topic. Our previous study has highlighted that although well-designed AR structures can provide ecologically equivalent mobile faunal communities to a nearby natural coral reefs, exact species composition between ARs and their nearby natural reefs remain distinct 3 years after deployment (Boakes et al. 2022a).

Artificial reef fish communities

Literature has shown that ARs are especially effective at increasing fish biomass of a given area because they supply additional food, enhance feeding efficiency and offer protection from predators (Bohnsack 1989). The increased fish biomass associated with ARs can lead to higher biogenic deposits onto the reef system (Ambrose and Anderson 1990, Rizzo 1990, Fabi et al. 2002, dos Santos et al. 2005, Reeds et al. 2018). Rizzo (1990) and Leitão (2013) showed that when these bio-deposits (e.g. excretion of ammonium, urea and faeces) enter the water column, they may be deposited and stored within sediments arounds ARs. This has been further demonstrated by Falcão et al. (2007), who highlighted that 2 years after deployment of ARs in Portugal, sediments displayed increased concentrations of organic and inorganic compounds by 30–60%, compared to pre-deployment levels. Other studies conducted on ARs in Portugal have demonstrated the link between higher levels of organic carbon (OC) and nitrogen on ARs with higher fish biomass (Vicente et al. 2008). It must be noted that literature investigating the links between fish and AR ecosystem functioning is still limited, with the majority of studies that do assess this being from temperate environments, and very few from tropical reefs. Despite this literature gap, research on the functioning of tropical natural reefs has shown that fish have key functional roles within coral reef systems (e.g. the role of surgeonfish in algal grazing; Bellwood et al. (2019)). Furthermore, reef fish have been highlighted to make substantial contributions to exporting OC to surrounding sediments (Polunin 1996), and restored systems and healthy fish stocks have been linked with significantly higher carbon sequestration rates (Howard et al. 2017, Stafford et al. 2021). More work is needed to specifically understand the roles fish play in the functioning of ARs, and if this is comparable to natural coral reefs.

Artificial reef benthic communities

Alongside providing habitats to fish, ARs are also colonised by corals, as well as fouling organisms such as sponges, tunicates and bryozoans (Perkol-Finkel and Benayahu 2007, Burt et al. 2009). Benthic invertebrates rapidly colonise ARs (Holmström and Kjelleberg 1994, Oren and Benayahu 1997, Mariani 2003) and have been shown to compete with each other for space on the substrata provided by artificial structures (Perkol-Finkel and Benayahu 2007). Despite the recruitment and growth of benthic invertebrates on ARs, studies have highlighted that their communities often remain distinct to those on nearby natural coral reefs (Perkol-Finkel et al. 2005, 2006), likely to some extent because coral reefs are formed over hundreds of years of complex, reef-forming processes (El-Naggar 2020). Currently no research has assessed the role of tropical AR benthic communities on ecosystem functioning, and if they can contribute to similar levels of nutrient uptake and release to neighbouring natural coral reefs. Natural coral reef benthic communities (specifically corals and algae) have been shown to be important in terms of the uptake of dissolved inorganic nutrients (DINs) like nitrate, phosphate and ammonium (Steven and Atkinson 2003, Den Haan et al. 2016). De Goeij et al. (2017) discussed the importance of sponges in terms of reef ecosystem functioning, where they are described as the "*ecosystem driver in the cycling of nutrients and energy on coral reef ecosystems"*. Current research is generally inconclusive with regards to the role of coral reefs in terms of net carbon sequestration, with Howard et al. (2017) highlighting that coral reef ecosystems can be sources or sinks of atmospheric CO_2, depending on the balance between two sets of processes: photosynthesis/respiration and calcification/dissolution. Research by Gattuso et al. (1998), highlighted that on most reefs, the CO_2 taken in by the coral's photosynthetic algae is approximately equal to the CO_2 released as a result of coral, algal and microbial respiration. The same study concluded that many coral reef ecosystems may actually lead to little/ no net carbon removal from the surrounding water column and atmosphere, especially when compared to other marine habitats such as mangroves and seagrasses.

Understanding the relationships between key nutrients and reef biota

Nutrient uptake and release is one of the core processes defining coral reef functioning (Brandl et al., 2019a) and is often used as a key measure for assessing ecosystem functioning (e.g. Lohrer et al. (2010), Trap et al. (2016), Griffiths et al. (2017)). Research has shown that preserving these processes is fundamental in safeguarding the health and resilience of a given system (Isbell et al. 2017) as ecosystem functioning on reefs supports common conservation objectives such as high coral cover, structural complexity and fish abundance (Brandl et al. 2019a). It is important to understand the relationships between key nutrients and reef biota (Bellwood et al. 2019), especially with regards to OC (e.g. Atwood et al. (2018), Nelson et al. (2023)) and dissolved inorganic nutrients (DINs; e.g. Hatcher and Frith (1985), Silbiger et al. (2018)). Despite this, the current link between community structure and coral reef ecosystem functioning remains poorly researched (Brandl et al. 2019b). Given the rapid global decline in coral reef health, as well as their capacity to deliver ecosystem services, there is an ever-increasing need to better understand the functioning of coral reefs (Bellwood et al. 2019), including those which have been restored using ARs. Vivier et al. (2021) highlighted that there is a limited amount of research which has evaluated ecosystem function of ARs and suggested that more studies should investigate the complex relationships between their functioning and reef biota. Our study compared nutrient storage and dynamics on an AR to a neighbouring natural coral reef, as well as to a degraded sand flat in Bali, Indonesia. We aimed to examine, if, based on these biogeochemical parameters, functioning on ARs was comparable to natural coral reefs, or if they displayed more similarities with degraded sand flats. More specifically, we aimed to investigate the differences in storage of TOM, as well as dynamics of inorganic nitrites, nitrites, ammonium and phosphates between these habitat types, and whether these differences were linked to observable changes in the community structure of mobile, cryptobenthic and benthic organisms.

Methods

Location

All data was collected in Tianyar Bay, North Bali, Indonesia (Figure 4.1) across three habitat types.

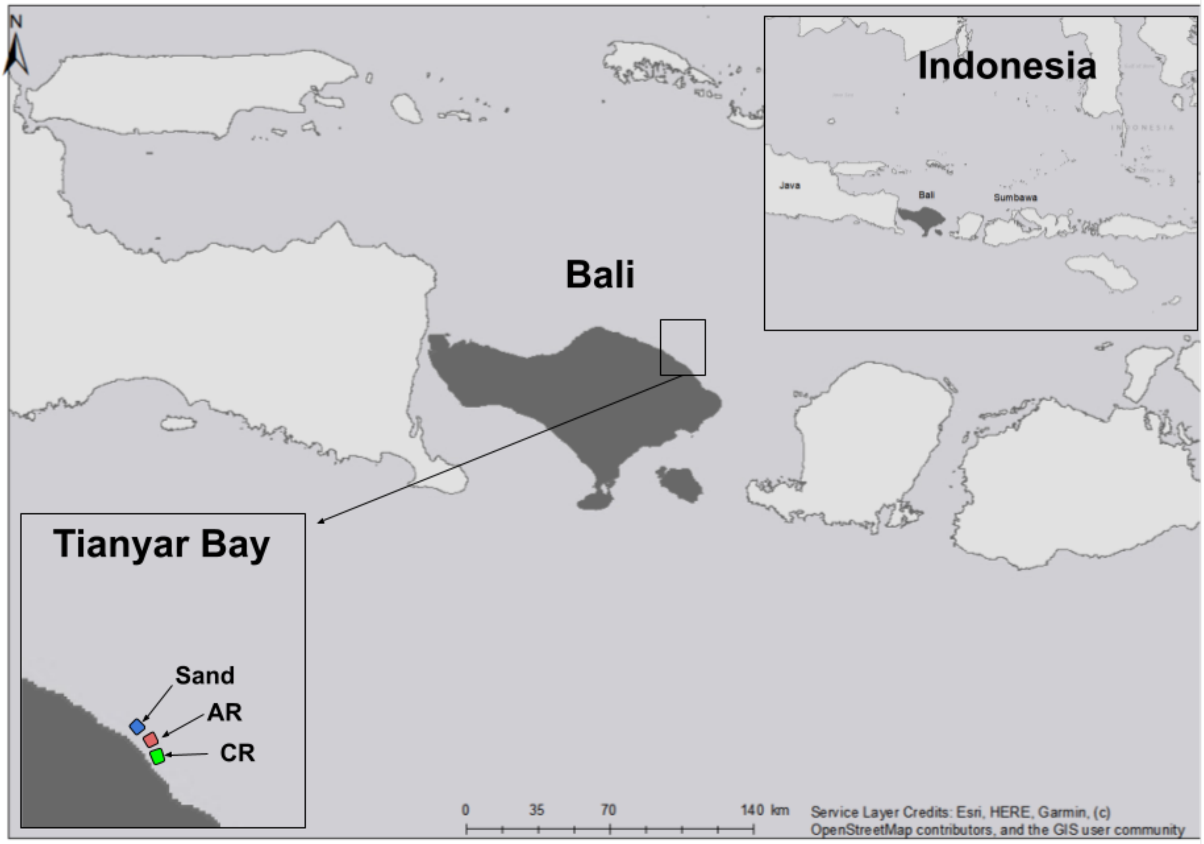

Figure 4. 1: Location of the three sampling sites (Sand Flat, Artificial Reef (AR) and Coral reef (CR)) within Tianyar Bay, Bali, Indonesia *(Created using ArcGIS OpenStreetMap powered by Esri).*

The three surveyed habitats, as shown by figure 4.2, included:

A) **Artificial reefs;** which were constructed by the local community using a three part mix of cement, calcium and sand, producing what were known as 'roti buaya'; 1 x 0.5m table shaped structures with a textured surface (e.g. with bumps, scratches, cracks and crevices) to allow natural recruitment of benthic species. The units were deployed between 5-10m depth on top of sand flats (*see below*). The ARs were installed in clusters, each with 20-30 units, covering an area of approximately $10m^2$. In each group, structures were stacked haphazardly (in a similar configuration between groups, and also locations), with the aim of providing optimal protective space, such as holes, tunnels and caves which provide additional habitat for sheltering fish (see figure 4.2B). The ARs were deployed over a 3 year period and the data collected in this study were from ARs aged between 1 – 3 years.

B) **Flat sand habitat (*Herein 'sand flat'*);** a sand-bottom area with little no/hard substrata and limited biological communities. Through conversations with the local community, it was understood that this sand flat was originally a healthy natural reef, but was destroyed several decades ago due to boat anchoring, coral harvesting and destructive fishing practices. After several decades of erosion and sedimentation, most of the remains of this degraded reef had disappeared and become covered with a layer of sand, hence the name 'sand flat'. It was decided that this habitat type would be included within our study, as it represented a control site (the AR habitat if no structures had been deployed there), therefore directly highlighting the changes in ecological communities as a result of AR deployment.

C) **Coral reef (CR);** a relatively pristine coral reef with a high biodiversity of benthic and mobile species. Through conducting reef health index (RHI) surveys (following Díaz-Pérez et al. (2016)) on this area of reef, it was shown through our research that this area of reef had a RHI score of 4.5 *("good" – "very good"; Boakes et al., unpublished*). Through personal communication with the local community, it was understood that this area of reef had not been previously targeted by the same localised threats as the other degraded habitats in this study (notably the sand flat), likely explaining why it was still in good condition.

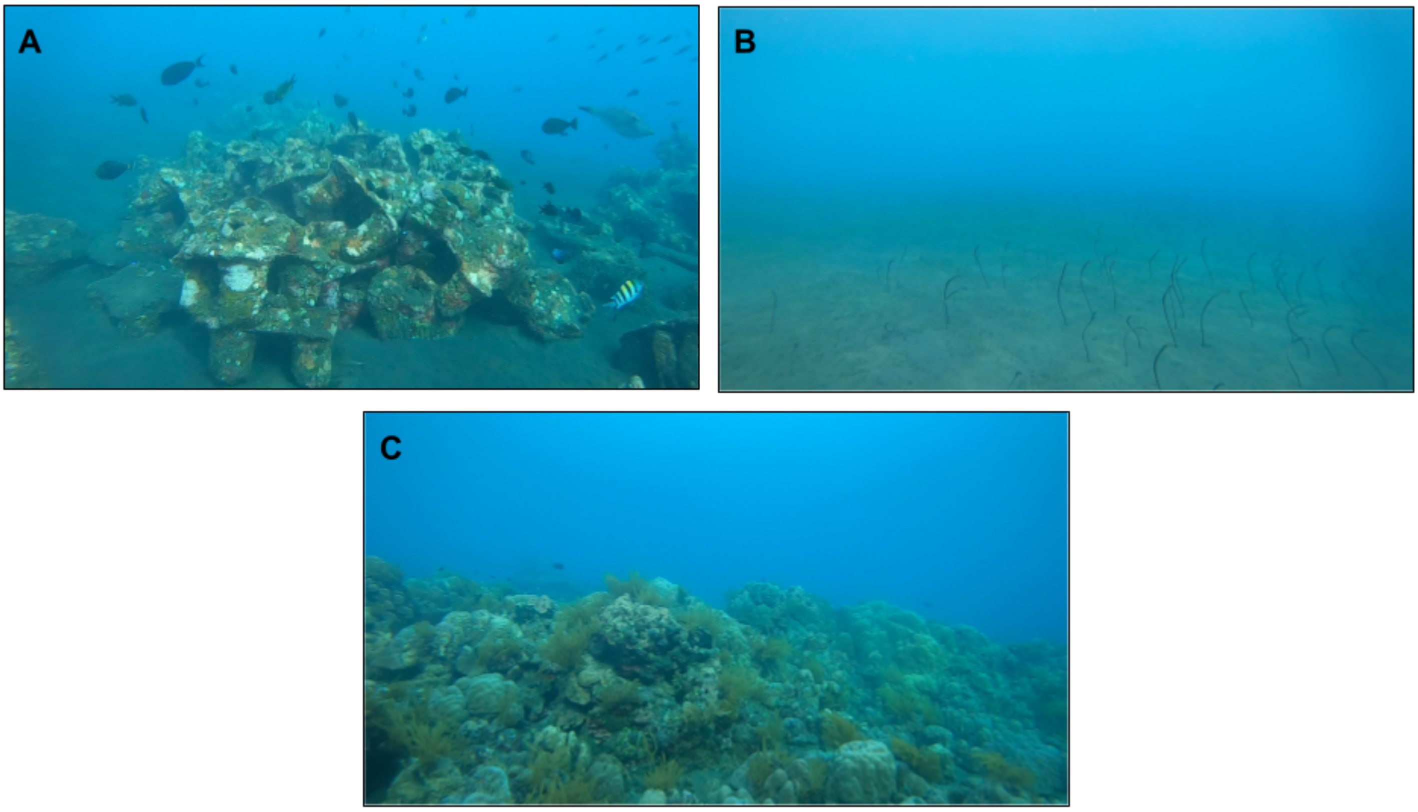

Figure 4. 2: Three habitat types were surveyed, including an artificial reef (A) and flat sand bed (B) and coral reef (C). These images were taken as screenshots from RUV recordings at each habitat type.

Each habitat was approximately 250m apart and surveyed within the same depth range (5-10m). Habitats were surveyed between the months of July 2021 to November 2022, with periodic nutrient sampling and ongoing ecological surveying (except the monsoon season due to poor fieldwork conditions) throughout the whole data collection period (*see Figure S.4.1 data collection schedule in the supplementary material).* All three habitat types were located within a marine protected area (MPA) managed by North Bali Reef Conservation, locally known as Yowana Bhakti Segara (www.northbalireefconservation.com; 8°11'27.5"S 115°29'42.9"E). The MPA (approximately 5 hectares (*personal communication with local fishers)* was established by the local community in

2017. At the time this study was conducted, the MPA continued to be well-enforced, and the localised threats which were thought to previously degrade the reef appeared to have stopped, likely in large part due to establishment of this MPA.

Nutrient sample collection

Overview

As a measure of dynamics of DINs (nitrate, nitrite, phosphate and ammonium), bottom water and pore water samples were taken. As a measure of total organic matter (TOM) storage, surface sediment samples were taken over a two day period and sediment traps were deployed for 8 weeks (replicated three times) over an eight month period.

Inorganic nutrients in bottom water samples

Bottom water samples were collected by SCUBA divers at the same frequency and sampling sites as the sediment samples (as described above). In total, across all three habitat types, 90 samples were taken. Bottom water samples were collected 0.5m above the reef/ sand surface, following Larned (1998) and Wild et al. (2009), which was measured using a 0.5m measurement line. The samples were taken using 250ml washed Nalgene polyethene bottles (following Lafferty et al. (2018)). The bottle caps were opened at the desired sampling site, allowing the water to flow in, and were closed after 1 minute once the bottle was free of air bubbles (following Limbong (2003)). After the dive, the samples were filtered using 0.7um Whatman glass microfiber filters, and then kept refrigerated in cool boxes (same as above) until they were tested in the lab (following Leichter et al. (2003)).

Inorganic nutrients in pore water samples

Fewer pore water samples were taken than bottom water samples because extraction of the pore water took substantially longer, meaning that SCUBA divers were limited in the amount of samples that could be taken per dive. In total, 10 pore water samples were taken in each of the three habitat types (n=30), across a transect where an even number of samples were taken at each depth between 5-9m (two samples were taken at 5m, two at 6m, two at 7m, two at 8m and two at 9m). Pore water was extracted using washed 300ml syringes, attached to a tygon tube with a perforated steel pointed tip. The steel tip was injected 10cm into the sediment (following Precht and Huettel (2004)), then, following the recommendations of Berg and McGlathery (2001), divers pulled back the syringe piston in a slow and steady movement so that pore water was drawn into the tygon tube. Once the syringe was filled to 300ml, it was immediately taken to the surface to avoid contamination with other water, where it was filtered, cooled and analysed using the same methods as the bottom water samples, as discussed above. Samples were kept refrigerated in cool boxes (as above) until they were tested in the lab (following Leichter et al. (2003)).

Total organic matter in surface sediment samples

In each habitat type, 30 sediment samples were taken between a depth of 5-10m, with five samples taken at 5m depth, five at 6m, five at 7m, five at 8m, five at 9m and five at 10m, similar to the methods of Pardo (2014). In total, across all three habitat types, 90 samples were taken. It was ensured that there was at least a 5m distance between each sample. Following the methods of Jewett et al. (2008), surface samples were taken at a sediment depth of 5cm at each sampling site, using washed 250ml polyethylene bottles which were filled with surface sediment and then sealed (following Honjo et al (1988)). On the AR, samples were taken as close as possible to the transect points, whilst also ensuring samples were collected next to or directly below the artificial reef unit. On the CR, samples were taken on sediment areas closest to the transect points. After the samples were collected, they were sealed using screw bottle caps and then stored in dark cool boxes kept between 4 - 5 degrees °C (following Von Wachenfeldt (2008)) for two hours whilst they were taken to the lab for processing. Samples were processed immediately once they had arrived at the lab.

Total organic matter in sediment traps

Following Buesseler et al. (2007), sediment traps were collected and analysed as an measure of TOM sediment deposition. Sediment traps were made from PVC cones (with a height of 20cm and a diameter of 13cm at the mouth and 3cm at the bottom (following Gust et al. (1994)). The specific cone shape (highlighted in figure 4.3) was chosen over standard cylindrical tube sediment traps, in order to allow a greater collection of sediment over short time scales. Using steel wire, the cones were fitted and attached to purpose built tripod stands which lifted the mouth of the traps 30cm above the sand / sea floor (figure 4.3). The sediment traps and their tripod stands were deployed from a boat and then carried to their desired sampling sub-sites by SCUBA divers. Deployment sub-sampling sites were chosen haphazardly on the flat sand bed and the artificial reef, however on the coral reef they were placed only on small, empty sand patches (instead of directly on top of corals). On the AR, the sediment traps were deployed directly next to the AR units (as close as possible to them), which ranged in age between 1 - 3 years.

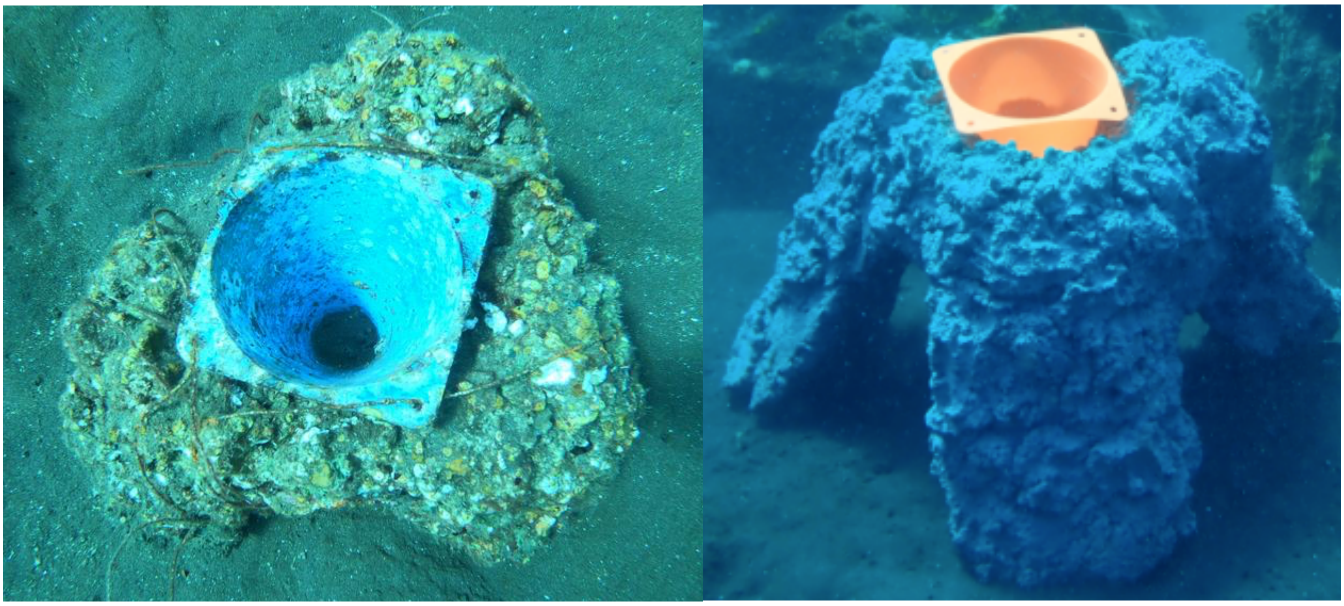

Figure 4. 3: Photographs of two of the sediment traps which were fitted to purpose-built tripods.

In total, 5 sediment traps were deployed in each of the three habitat types. The sediment traps were retrieved by removing the cones from their tripod frames and carefully brought to the surface. After being bought ashore, the samples were transferred to 250ml PET bottles, sealed with screw bottle caps and then stored in dark cool boxes (following Von Wachenfeldt (2008)) whilst they were taken to the lab for processing. Samples were deployed for 8 weeks (following Harrison and Hall (2021) at a time, and then re-deployed two more times (three replicates) at the exact same locations. In total, the sampling period for the sediment traps was 8 months (f*igure S.4.1*), giving a total of 45 samples across all three habitat types over this time.

Lab analysis

Following the methods of Baum et al. (2015), water samples were analysed for nitrate, nitrite, ammonium and phosphate using a Hach DR900 using the cadmium reduction, diazotization, salicylate, and ascorbic acid method, respectively. Detection values followed those of Baalbaki et al. (2019) (e.g. the detection value for nitrate and nitrite was 0.01 mg/L and 0.001 mg/L respectively). Sediment samples (both surface sediments and sediment trap) were tested for total organic matter (TOM) content using the 'loss-on-ignition' method, which compared the weight of the dry mass of the sample to the 'ashed' mass (after it had been combusted at 550 °C for 12h, thus allowing percentage total organic matter (herein % TOM) to be calculated (following Wang et al. (2011)).

Given that the primary component of TOM in ocean sediments is total organic carbon (TOC; Sutherland 1998), these results were used as a measure of TOC within our sediment samples.

Biological community structure

For data collection on benthic, cryptic and mobile communities, three sample sites (herein sites) were established in each of the three habitat types, which were each approximately 50m apart. Sampling sites were chosen haphazardly, and were marked using a coded sign, attached to a frame and a 30cm² concrete base.

Mobile species

Following the methods of Boakes et al. (2022a), remote underwater video (RUV) was used as a measure of mobile community structure by comparing the abundance and diversity of mobile species between sites. Video samples were taken within 6 weeks of water and surface samples being taken and during the time of sediment trap data collection using a GoPro Hero 5 HD 1080p underwater camera between 8-10am on sampling days (of varying tidal conditions), only on calm mornings when underwater visibility was at least 15m (measured using a visibility measuring line). Again, following Boakes et al. (2022b), each habitat type had three sampling sites, which were each recorded twice over the research period, giving a total of 6 samples per habitat type, and 18 in total. RUV videos were recorded for a duration of 25 minutes, allowing for an initial 5 minute settlement period and 20 minutes of analysis time (following Boakes et al. (2022b)), allowing appropriate estimates of community structure to be obtained, only missing low numbers of rarely occurring, often transient, species. From the videos, only clearly identifiable individuals were recorded. Mobile species were identified to species level and in circumstances of uncertainty, advice was sought from local expert, Yunald Yahya (LINI foundation). As a relative measure of abundance, the maximum number of individuals seen in any frame (herein MaxN; following Whitmarsh et al., (2017)) during the 20 minute video (each sampling period) was calculated.

Cryptic species

It was noticed that there were several crypto-benthic (CB) fish (small (<15cm) fish which reside mostly inside the reef substrate and rarely enter the water column above). On the AR and CR, these CB fish resided within the protective space provided by the substrata, and thus were not clearly identifiable from the RUV recordings. Due to the potentially important role of CBs in reef ecosystem functioning (as demonstrated by Brandl et al. (2019a)), it was decided that a 'cryptical crawl' (underwater stationary point count) would be conducted to estimate CB community structure, following Mallet et al. (2014). In the same site as each of the RUV recordings (nine in total), two independent Underwater Visual Census surveys were performed by SCUBA divers in each of the three habitat types (n=6), who recorded the cryptic fish species that were present within the refuge area provided by the substrata (thus making them mostly unrecorded from the RUV analyses). Following Watson (1997), all cryptic fish within a cylindrical column (of a 10m radius) were recorded over a 12 minute sampling period. One diver was responsible for recording fish species on a pre-prepared dive slate, whilst the other ensured all sampling was conducted within the 10m cylindrical column. As with RUV analysis, the maximum number of individuals of each species (MaxN) was recorded.

Benthic species

Photo-quadrat sampling is a method commonly used to determine estimates on benthic community structure on coral reefs (Leujak and Ormond 2007). Following the methods of Clua et al. (2006) and Chaves et al. (2013), 40cm^2 quadrats were placed randomly along fixed 20m line transects running across each site (the same sites used for the RUV samples). Using SCUBA and an Olympus TG-6 camera, 10 photo-quadrats were taken at each of the three sites (90 in total across all three habitat types). Following Leujak and Ormond (2007), photos were taken approximately 2 m away from the substrate, thus fitting the whole quadrat into one photograph. Percentage cover was calculated by dividing each photograph into four 10cm^2 sub-frames (following Mantelatto et al., 2013), allowing benthos to be more accurately estimated. Within each sub-frame, benthos was identified to at least family level (following Schmidt-Roach et al. (2008)) and total percentage coral cover of each sub-frame was analysed using Coral Point Count with Excel extensions, following the guidance of Kohler and Gill (2006). Corals were identified with the help of multiple benthic ID guides and local experts were approached in times of uncertainty.

Statistical analysis

Two-way ANOVA was run to analyse inorganic nutrient concentrations in water samples, with water type (bottom, pore) and habitat type (CR, AR, S) as the two factors, separate analyses were conducted for each nutrient. Two-way ANOVA was also run to analyse % TOM in sediment traps, with sampling period (May, September, November) and habitat type as the two factors. A separate one-way ANOVA was run to analyse TOM between habitat types in surface sediments. Additionally, to compare surface %TOM values with those collected from sediment traps, we included the surface samples as an additional sampling period in the sediment trap database, and ran a two-way ANOVA between habitat and sampling period again (with quasipoisson link functions). In all cases, assumptions of standard ANOVA (normality and homogeneity of variance) were not adequately met. Following Crawley (2007) GLMs with quasipoisson link functions were used with ANOVA p-values calculated using F tests. Additional multiple comparison tests were conducted using the emmeans package in R (Russell and Length, 2021). No p value adjustment was used as the number of comparisons of interest was much lower than the full set of interaction terms in the two-way models.

To explore community structures for mobile, CB and benthic assemblages, PERMANOVA was run (separately for each community) using the Vegan package in R (following Anderson (2001)) to assess the difference in communities (MaxN for mobile species and CBs, and percentage cover for benthos) between habitat types. Data was square-root +1 transformed prior to use, to avoid the excessive weighting of common species over rare. A Bray-Curtis resemblance matrix was used with 9999 permutations and PERMANOVA run with unrestricted permutation of raw data. For mobile species, as RUV recordings were taken from the same sites within each habitat, site was nested within habitat. For other tests, there was independence between all samples. Principal Coordinate Analysis (PCoA) was used to visualise community variation between habitat types and to highlight key species which differentiated the different biological communities at the different habitat types (using criteria of $p \leq 0.001$ and $r > 0.45$ to display discriminating species arrows).

Results

Inorganic nutrients in bottom and pore water samples

For nitrate, nitrite and ammonium, two way ANOVA showed no significant interaction terms and no differences between habitat types ($p>0.05$ in all cases), however pore water had significantly higher concentration than bottom water for all three nutrients (table 4.1).

Table 4. 1: ANOVA results and mean *(+/- SD)* concentrations of inorganic nutrients from water samples

Nutrient	**Factors**	**d.f.**	**F value**	**P value**	**Bottom water mean concentration (mg/L)**	**Pore water mean concentration (mg/L)**
Nitrate	Water type	1, 114	51.6	<0.001	0.0027 *(0.0058)*	0.0130 *(0.0091)*
Nitrite	Water type	1, 114	18.7	<0.001	0.0030 *(0.0013)*	0.0042 *(0.001)*
Phosphate	Water type* habitat type	2, 114	12.02	<0.001	0.9466 *(0.6446)*	1.8 *(1.3658)*
Ammonium	Water type	2, 114	3.95	0.0221	0.0052 *(0.0136)*	0.047 *(0.0449)*

Phosphate showed a significant interaction term between water type and habitat (table 4.1), with significantly higher phosphate concentrations in pore AR samples compared to bottom water AR samples (AR bottom: AR pore $p<0.001$; figure 4.4). No other differences between water type within habitat type occurred. Pore water phosphate concentration was significantly higher on the AR than both other habitats (AR: CR $p<0.001$, AR: Sand $p<0.001$), but was not significantly different between the CR and sand flat (CR: Sand $p> 0.05$; figure 4.4). In terms of phosphate concentrations in bottom water samples, no significant difference were shown between habitat types, except between artificial reef and sand bottom water ($p = 0.0434$).

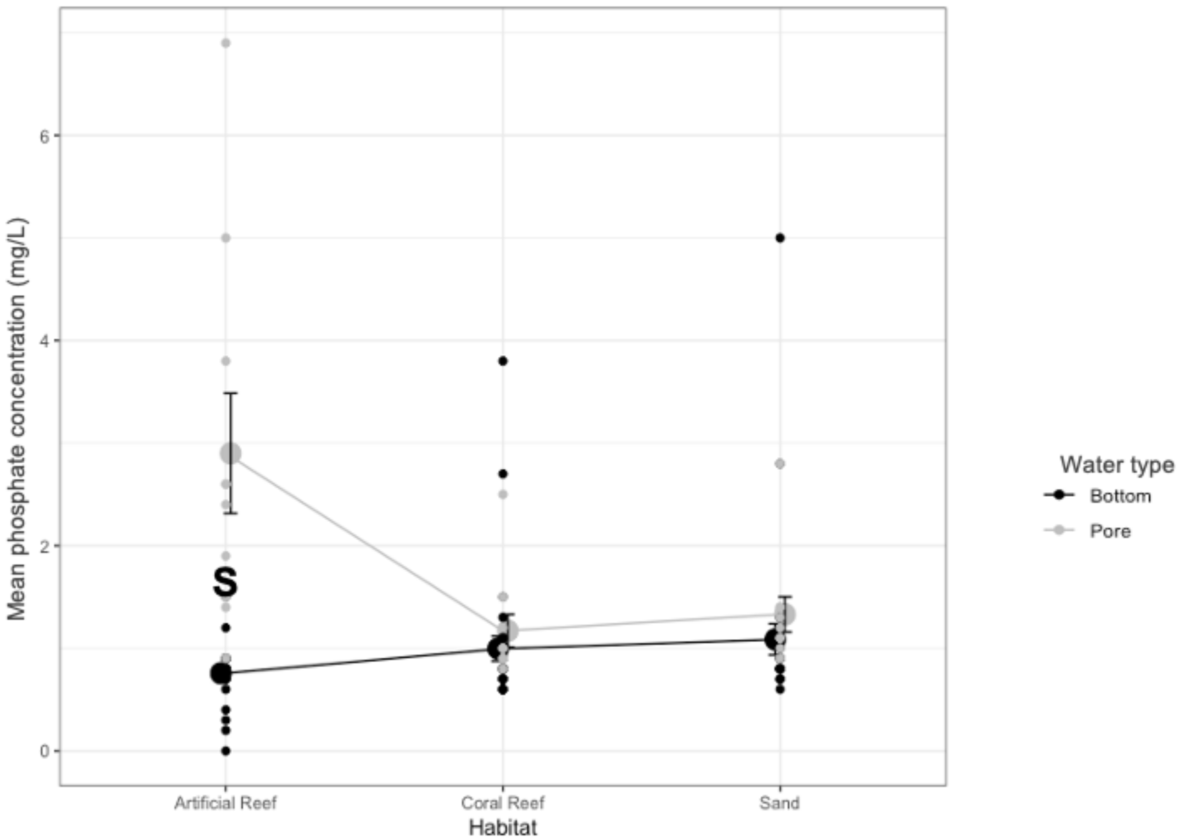

Figure 4. 4: Interaction plot (means +/-SE) of mean phosphate concentration from water samples in different habitats at the different sampling periods. 'S' indicated concentrations were significantly different ($p<0.05$) between pore water and bottom water samples.

Total organic matter in surface sediment samples and sediment traps

In terms of surface sediment sampling, one-way ANOVA showed significant differences between habitat types (table 4.2). In surface sediments, the CR was found to have significantly higher % TOM than both other habitat types (multiple comparison p<0.001). The AR was also found to have a significantly higher % TOM than the sand flat (p=0.0073). In terms of sediment traps, % TOM showed a significant interaction between habitat type and sampling period (table 4.2).

Table 4. 2: ANOVA results and mean *(+/- SD)* total organic matter content of sediment samples between habitat types.

Sampling type	Factor(s)	d.f.	F value	p value	Mean sand TOM content (%)	Mean AR TOM content (%)	Mean CR TOM content (%)
Surface sediment	Habitat types	2, 87	132.4	<0.001	0.5489 *(0.165)*	0.6617 *(0.012)*	1.3158 *(0.029)*
Sediment traps	Habitat type * sampling period	4, 25	6.94	<0.001	1.11 *(0.743)*	1.31 *(0.938)*	1.78 *(0.452)*
Comparing surface sediment and sediment traps	Habitat type * sampling period	6, 112	12.73	<0.001	n/a	n/a	n/a

In May and September, the CR was shown to have significantly higher % TOM than the other habitats, but this difference was not present in the November samples, when both sand and AR samples were not significantly different that to the CR (figure 4.5). To compare surface %TOM values with those collected from sediment traps, we included the surface samples as an additional sampling period in the sediment trap database, and ran a two-way ANOVA between habitat and sampling period again (with quasipoisson link functions). Again, a significant two way interaction term was found (table 4.2), but no differences between the values of %TOM were found for the different habitats between the May values in the sediment traps and the surface values (multiple comparisons p>0.05 in all cases), with May being the closest time period to when the surface samples were taken. Differences between habitats did occur between the surface sediments and the sediment traps at other times of year.

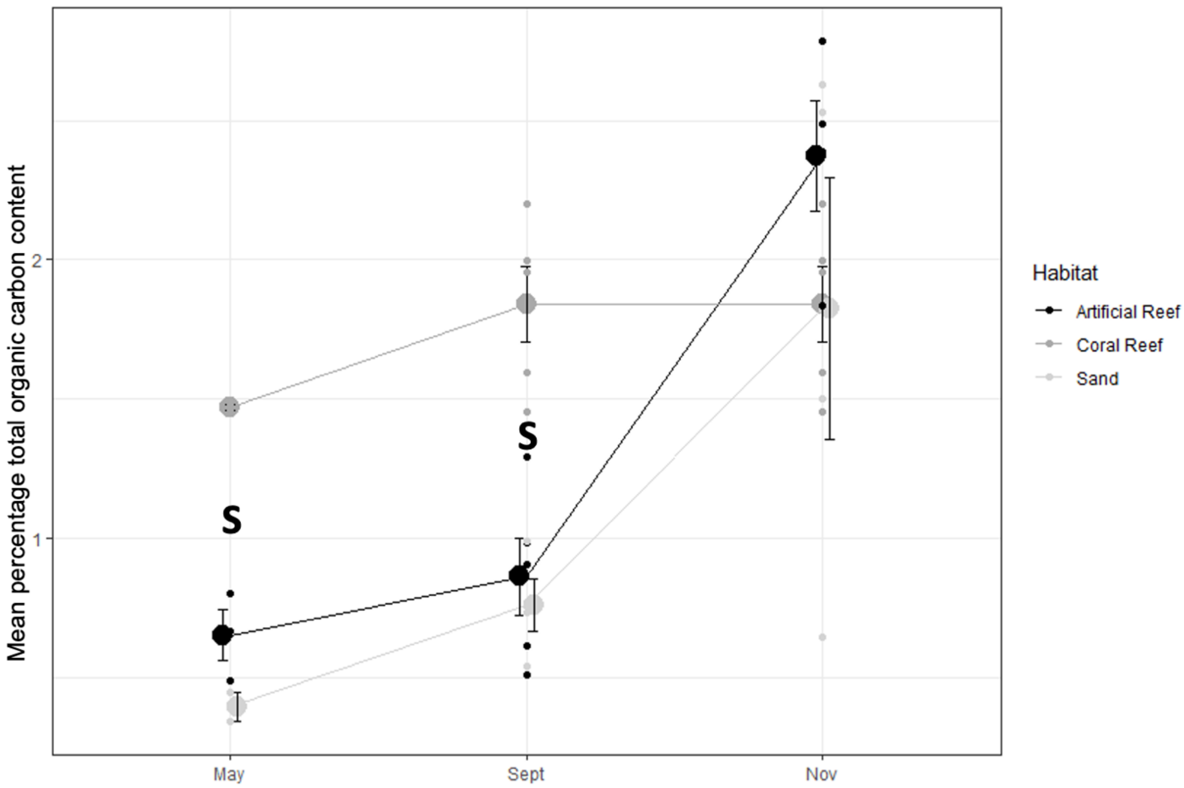

Figure 4. 5: Interaction plot (means +/-SE) of % TOM content in sediment traps in different habitats at different sampling periods (n=5 in most cases, although there were occasional missing/ broken samples). 'S' indicated coral reef % TOM was significantly higher than the other habitats at the sampling periods indicated (p<0.05). No other differences between habitats within a time period occurred.

Biological community structure

In terms of community structure, PERMANOVA showed significant differences between habitat types for mobile and benthic communities, but no significant differences for cryptobenthic communities (table 4.3). For mobile and benthic communities, pairwise comparisons showed significant differences between all habitat types (p<0.05 in all cases).

Table 4. 3: PERMANOVA of community structure between habitat types.

Community	d.f.	F value	P value
Mobile	5, 15	8.89	<0.001
Cryptobenthic	1, 4	2.22	0.10
Benthic	2, 87	65.62	<0.001

Despite the AR and CR having distinct differences in terms of mobile and benthic communities, they were shown to be much more closely related to each other than the sand flat (which was substantially different to the AR and CR; figures 4.6A and 4.6B). Mobile and benthic communities on the sand flat were greatly different to both other habitat types. This was also highlighted by table 4.4, which showed that the AR and CR had a similar MaxN of mobile and cryptobenthic communities, which were both very different to the sand flat.

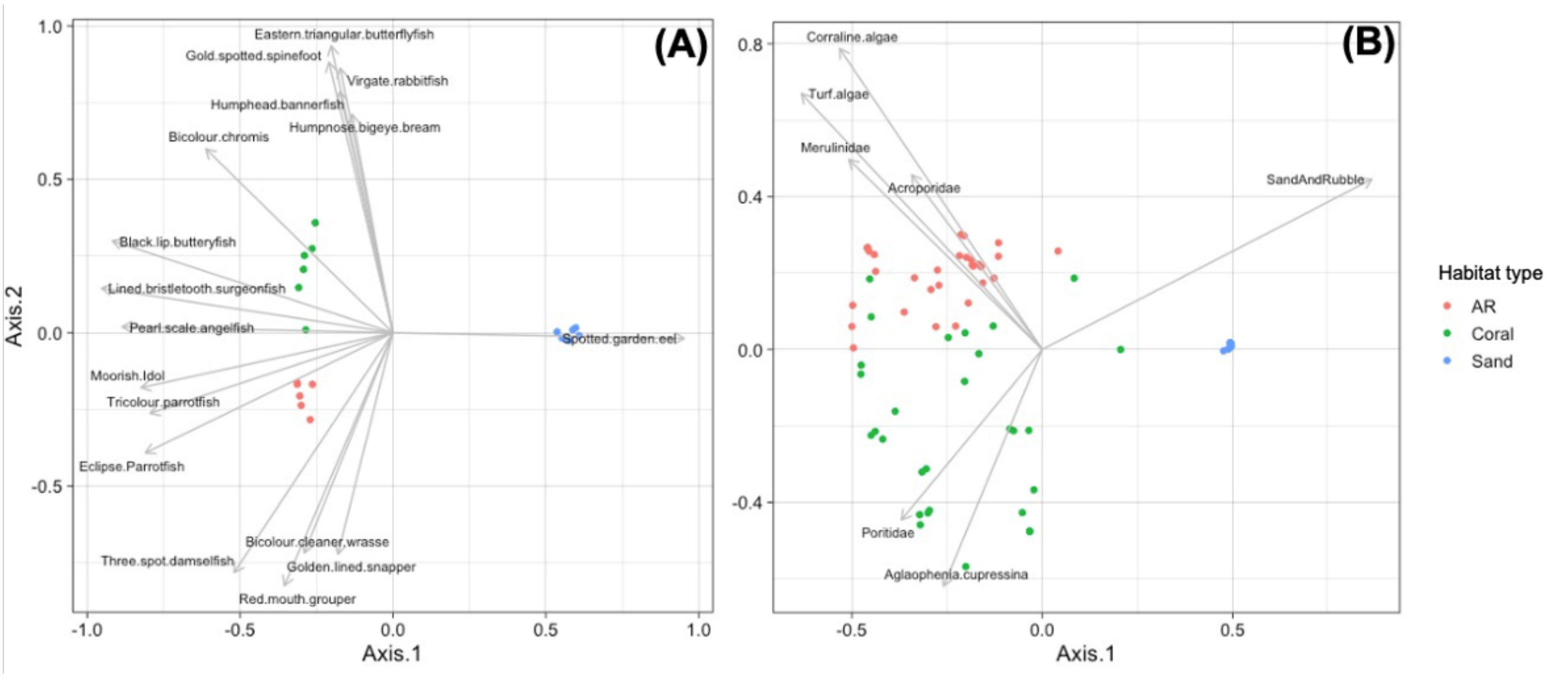

Figure 4. 6: Principal Coordinate Analysis (PCoA) plot for mobile community structure (A) and benthic community structure (B) within habitat type, with Pearson's correlation vectors (> 0.45) overlaid in black. Note: unlike 6A, scientific names were used to describe benthos (6B) as many of the genera have no known common name.

Table 4. 4: Mean *(+/- SD)* MaxN of mobile and cryptobenthic communities and percentage coral cover between habitat types.

Habitat type	Average MaxN (mobile communities)	Average MaxN (cryptobenthic communities)	Coral cover (%)
CR	42.6 *(5.7)*	13.7 *(1.5)*	44.13 *(35.2)*
AR	42.8 *(3.4)*	9.6 *(3.8)*	13.09 *(8.1)*
Sand	4.8 *(1.2)*	0	0

Discussion

Summary of results

Our key findings showed: 1) There were no significant differences in inorganic nutrients between habitat types for bottom water samples, 2) Pore water phosphate concentrations were significantly higher on the artificial reef than on both other habitats 3) Total organic matter content in sediments were significantly higher on the coral reef than both other habitat types, and 4) Sediment trap sampling period three (September – November) displayed no significant differences between habitat types in terms of total organic matter. Below, we assess how the inputs and up-take of certain nutrients may provide possible explanations for our findings.

There were no significant differences in inorganic nutrients between habitat types for bottom water samples

Research has shown how nutrients in the nearshore water column (including bottom water) are strongly influenced by currents (Fonseca and Kenworthy 1987, Lourey et al. 2006), wind (Lee et al. 1992, Vicente et al. 2008) and tides (Anwar et al. 2014, Davies et al. 2014). These environmental factors may have had a stronger influence on bottom water nutrient concentration at each of the habitat types than localised ecosystem processes, and this may explain why there were no significant difference shown for inorganic nitrates, nitrites, phosphates and ammonium concentrations from bottom water samples. In contrast, pore waters are known for storage of inorganic nutrients as they are formed by sedimentation of particles from the overlying water column, thus 'trapping' and storing compounds that were previously in the water column (Bufflap and Allen 1995, Batley and Giles 2014, Huettel et al. 2014). This explains why pore waters may be less influenced by the environmental factors discussed above, and provides a likely reason for the significant difference in inorganic nutrients concentration between pore water and bottom waters.

Pore water phosphate concentrations were significantly higher on the AR than on both other habitats.

In terms of up-take of phosphates, it is generally agreed that established coral reef communities have a very tight cycling of DINs (Steven and Atkinson 2003, Rädecker et al. 2015, Graham et al. 2018), and this may explain why the CR had significantly lower phosphate pore water concentration than the AR. Coral reefs are known to have a high phosphate uptake rates (Den Haan et al. 2016), as corals can efficiently utilise organic phosphate excreted by other organisms within their localised system (Shantz and Burkepile 2014). Furthermore, the AR had a less established benthic community (made up mostly of turf and coralline algae and pioneering acroporids, with a coral cover of ~13%) when compared to the CR (made up mostly of corals (notably massive poritids), sponges and hydroids, with a coral cover of ~44%)). Thanner et al. (2006) showed that assemblages of benthic organisms on tropical ARs may take up to 5 years before they begin to mimic natural communities. Given that the ARs in this study ranged between only 1 - 3 years, it was unsurprising that the AR had a substantially lower coral cover and less established benthic community than the CR . The lower coral cover on the AR likely explains why it absorbed less phosphate than the CR (based on Shantz and Burkepile (2014); Den Haan et al. (2016) as referenced above). Furthermore, the dominance of massive poritids on the CR may provide an additional reason why phosphate concentration was lower on the CR, especially because poritids have been shown to provide important contributions to phosphate up-take on reefs (e.g. D'Elia (1977); Atkinson et al. (1994)).

In terms of phosphate input, previous studies have highlighted that fish faecal pellets are high in micronutrients, especially phosphate (Geesey et al. 1984, Rempel et al. 2022). Groupers (specifically coral grouper and red mouth grouper) were one of the differentiating fish families for the AR, which are known to excrete large quantities of phosphate (Schiettekatte 2021)), and this, along with the reduced potential for phosphate removal from the ARs, may help to further explain the trend that pore water phosphate concentrations were significantly higher on the AR than on both other habitats. Furthermore, cryptobenthic reef fish, despite often being overlooked, have been described as a cornerstone of ecosystem functioning on coral reefs (Brandl et al. 2019) and are thought to play a key role in the cycling of reef nutrients, including phosphorus and nitrogen (Schiettekatte 2021)**.** Given that this study found no significant difference between the AR and CR in terms of cryptobenthic fish, it was not possible to make further conclusions on how their communities affected ecosystem functioning between habitat types. Further research on the role of cryptobenthic fish in reef nutrient cycling would greatly increase understanding of the links between reef biota and ecosystem functioning.

Total organic matter levels in sediments were higher on the coral reef than both other habitat types

It must first be noted that the key component of TOM in ocean sediments is TOC (Sutherland 1998), and therefore our % TOM findings were used as an approximate measure of TOC within our sediment samples. In terms of fish communities, one distinct mobile community characteristic of the CR was that it appeared to be made up of herbivorous fish, notably rabbitfish (gold spotted spinefoot and virgate rabbitfish) and surgeonfish (lined bristletooth). Communities on the other habitats did not appear to have such a strong representation of herbivorous fish, and instead were shown to have different communities, which were strongly driven by predatory fish (grouper and snapper) in the AR. Herbivorous reef fish play an important role in the carbon dynamics of marine sediments (Legendre and Le Fèvre 1995, Atwood et al. 2018) due to their specific gut bacteria, which is thought to cause increased sedimentation of OM(Montgomery and Pollak 1988, Mountfort et al. 2002, Smriga 2010). In fact, certain examples show that the faecal pellets surgeonfish are particularly high in OC (e.g. Ezzat et al. 2019), as well as research highlighting that rabbitfish faecal pellets may provide notable contributions to deposited organic matter within localised reef sediments (e.g. Peleg et al. 2020). It is possible that the higher levels of TOM recorded on the CR may be due to the differences in fish community structure, specifically the distinct communities of herbivorous fish on CR.

Despite storing less TOM than the CR, the AR was shown to store more TOM than the sand flat, with surface sediment samples having significantly higher TOM on the AR than the sand flat. Furthermore, sediment trap samples were shown to have notably higher (yet insignificant) TOM content on the AR than the sand flat. This was likely because the AR had a more complex and abundant fish community than the sand flat, which may have caused higher biogenic deposits onto the reef system and surrounding sediments (as shown by Dos Santos et al. (2005)). These findings were supported by those of Vincente et al. (2008), which linked OC deposits on ARs on with fish biomass (Vicente et al. 2008). As the ARs mobile and benthic communities start to mimic those on CRs over time (e.g. Perkol-Finkel et al. (2006), Folpp et al. (2013)), the AR may begin to store similar levels of OC, given that biological communities are one of the key drivers of ecosystem functioning (Brandl et al. 2019b).

In terms of benthic communities, it is likely that a given proportion of the TOM observed in the CR (and the AR to a lesser extent) were exudate of the habitat's benthos, notably sponges and corals. Research has shown that sponges can release vast amounts OC species by rapidly expelling their filter cells (Pawlik et al. 2016, De Goeij et al. 2017). Furthermore, literature has also shown that hard corals efficiently trap POM from the water column in their mucus, and release this carbon rich exudate to nearby sediments (Wild et al. 2004). This mucus is often considered as excess OC, because the corals have had to consume large amounts of 'low quality' food as a means of obtaining sufficient nutrients such as nitrogen and phosphorous (Bythell 1988, Pinnegar et al. 2007). Over half (56–80%) of the expelled coral mucus immediately dissolves (Moriarty et al. 1985), although much of the remaining mucus trap will increase in OC content as it traps more suspended particles, and is then thought to rapidly settle in nearby sediments (Wild et al. 2004). The transport of these materials via coral mucus sedimentation has been shown to contribute to 2-26% of the OC within sediments (Wild 2003). The results of this study showed that Poritidae and Agariciidae were two coral families which were proportionately more important on the CR. There are no known publications which directly compares mucus release rates between families, however research has associated these two families as potentially important mucus producers. For example, Domart-Coulon et al. (2006) associated Poritidae with abundant mucus production, which has been shown to stimulate the growth of vast bacterioplankton communities within nearby sediments (Silveira et al. 2017). Furthermore, Glynn et al. (2011) highlighted that Agariciidae corals are often covered by mucus-laden strings that coat the colony surfaces. The dominance of these two coral families (as well as CR's higher general coral coverage than other habitats) likely led to the CR having higher mucus release rates than the other habitats, providing another possible explanation as to why the CR samples had the highest observed TOM content.

Sediment trap sampling period three displayed no significant differences between habitat types in terms of total organic matter

Our results found that sampling period three showed no significant difference in % TOM content between any of the habitats, however sampling periods one and two did. We also highlighted that surface sediments had significantly lower % TOM content than sediment trap in sampling period three, although they displayed no significant differences to the first two sampling periods. It must be noted that Indonesia's monsoon was between the months of October to March, and in this time, higher precipitation leads to increased runoff of nutrients. It is likely that sampling three's observed differences was because it was the only sampling period within the monsoon season, and therefore the only one which would be trapping the additional organic material as a result of it. These findings are in agreement with other studies, which have also shown that nutrient concentrations on coral reefs in Indonesia are higher during the monsoon season (e.g. Nugrahadi et al. (2010) and Damar et al. (2019)). Furthermore, Wild (2003) showed that the release of OM by corals over a spawning period provides notable seasonal contributions to sedimentary OM deposition. The coral spawning season in north Bali is known to occur each November (*Yunaldi Yahya, pers. comms.*). If the coral spawning period had occurred whilst the sediment traps were still deployed, it may have provided additional contributions to OC deposition (collection within the traps), thus providing another potential reason why sampling period three had significantly higher % TOM content.

Conclusion

Overall, the AR in this study was shown to not yet be functioning at the same level as the CR, in terms of TOM storage and DIN dynamics. The difference between the AR and CR in terms of community structure, specifically less complex benthos (likely leading to less release of TOM to sediment and less up-take of phosphate), as well as different fish communities, which perhaps explained why the AR was not yet functioning as the CR. Despite this, in some cases, TOM storage and DIN dynamics, were shown to be different on the AR than the nearby sand flat (with levels on the AR being shown to be more similar to the CR), likely due to the ARs relatively more complex biological communities. Given that the ARs in this study ranged between only 1 – 3 years old, and that tropical ARs may take up to 5 years to begin to mimic natural benthic communities and 6 – 7 years to begin to mimic natural fish communities, it is encouraging that an AR may start to show similarities to the functioning of a CR over a relatively short time scale. It is expected that the functioning of ARs will show more similarities to CRs over time, as communities increase in complexity and begin to mimic those on natural reefs. Our examination of nutrient cycling and storage compared to community structure on coral and artificial reefs has given rise to a number of key hypotheses which may determine the differences found. However, considerably more research is needed to confirm the links between biological community structure and ecosystem functioning on ARs and CRs, as well as directly identifying the important species of reef flora and fauna that is associated with depositing large amounts of TOM to nearby sediments.

Statements and Declarations

Author contributions:

ZAB wrote the manuscript (with contributions from all authors) and led the fieldwork. He also conceived the original idea, alongside RS. IGNAS helped with fieldwork and carried out the lab analysis, as well as assisting with obtaining local research permits. RS supervised throughout, and helped with statistical analysis. DJF and AEH provided critical feedback and advice throughout, which helped to improve the manuscript.

Permits and local collaborations:

A research permit was obtained from Indonesia’s National Research and Innovation Agency (research permit number: 34/TU.B5.4/SIP/VII/2023) for this project. This research involved a

collaboration Universitas Pendidikan Ganesha, specifically Dr I Gusti Ngurah Agung Suryaputra (and his students), who helped with fieldwork and lab analysis.

Funding:

Zach Boakes, the first author of this paper, was supported by a studentship with Bournemouth University, UK, as well receiving the 2021 'Emerging Scientist' grant from Earthwatch Institute. All other authors declare that no specific funds, grants, or other support was received during the preparation of this manuscript.

Conflict of interest:

The authors declare no conflicts of interest.

Data availability statement:

The collected data are available on request to the corresponding author.

References

Ambrose, R. F. and Anderson, T. W., 1990. Influence of an artificial reef on the surrounding infaunal community. *Marine Biology*, 107 (1), 41–52.

Ampou, E. E, Tito, C., 2019. Begini Kondisi Kesehatan Terumbu Karang di Perairan Nusa Penida.

Anderson, M. J., 2001. A new method for non-parametric multivariate analysis of variance. *Austral ecology*, 26 (1), 32–46.

Anwar, N., Robinson, C., and Barry, D. A., 2014. Influence of tides and waves on the fate of nutrients in a nearshore aquifer: Numerical simulations. *Advances in water resources*, 73, 203–213.

Atkinson, M.J., Kotler, E. and Newton, P., 1994. Effects of water velocity on respiration, calcification, and ammonium uptake of a Porites compressa community.

Atwood, T. B., Madin, E. M. P., Harborne, A. R., Hammill, E., Luiz, O. J., Ollivier, Q. R., Roelfsema, C. M., Macreadie, P. I., and Lovelock, C. E., 2018. Predators shape sedimentary organic carbon storage in a coral reef ecosystem. *Frontiers in ecology and evolution*, 110.

Baalbaki, R., Ahmad, S. H., Kays, W., Talhouk, S. N., Saliba, N. A., and Al-Hindi, M., 2019. Citizen science in Lebanon—A case study for groundwater quality monitoring. *Royal Society Open Science*, 6 (2), 181871.

Bailey-Brock, J. H., 1989. Fouling community development on an artificial reef in Hawaiian waters. *Bulletin of Marine Science*, 44 (2), 580–591.

Baine, M., 2001. Artificial reefs: a review of their design, application, management and performance. *Ocean & coastal management*, 44 (3–4), 241–259.

Batley, G. E. and Giles, M. S., 2014. Solvent displacement of sediment interstitial waters before trace

metal analysis. *Water Research*, 13 (9), 879–886.
Baum, G., Januar, H. I., Ferse, S. C. A., and Kunzmann, A., 2015. Local and regional impacts of pollution on coral reefs along the Thousand Islands north of the megacity Jakarta, Indonesia. *PloS one*, 10 (9), e0138271.
Becker, A., Taylor, M. D., and Lowry, M. B., 2017. Monitoring of reef associated and pelagic fish communities on Australia's first purpose built offshore artificial reef. *ICES Journal of Marine Science*, 74 (1), 277–285.
Bellwood, D. R., Streit, R. P., Brandl, S. J., and Tebbett, S. B., 2019. The meaning of the term 'function'in ecology: A coral reef perspective. *Functional Ecology*, 33 (6), 948–961.
Berg, P. and McGlathery, K. J., 2001. A high-resolution pore water sampler for sandy sediments. *Limnology and Oceanography*, 46 (1), 203–210.
Bianchi, D., Carozza, D. A., Galbraith, E. D., Guiet, J., and DeVries, T., 2021. Estimating global biomass and biogeochemical cycling of marine fish with and without fishing. *Science advances*, 7 (41), eabd7554.
Boakes, Z., Hall, A., Jones, G., Prasetyo, R., Stafford, R., and Yahya, Y., 2022(a). Artificial coral reefs as a localised approach to increase fish biodiversity and abundance along the North Bali coastline. *AIMS Geosciences* [online], 8 (2), 303–325. Available from: 10.3934/geosci.2022018.
Boakes, Z., Hall, A. E., Ampou, E. E., Jones, G. C. A., Suryaputra, I. G. N. A., Mahyuni, L. P., Prasetyo, R., and Stafford, R., 2022(b). Coral reef conservation in Bali in light of international best practice, a literature review. *Journal for Nature Conservation* [online], 126190.
Brandl, S. J., Tornabene, L., Goatley, C. H. R., Casey, J. M., Morais, R. A., Côté, I. M., Baldwin, C. C., Parravicini, V., Schiettekatte, N. M. D., and Bellwood, D. R., 2019(a). Demographic dynamics of the smallest marine vertebrates fuel coral reef ecosystem functioning. *Science*, 364 (6446), 1189–1192.
Brandl, S. J., Rasher, D. B., Côté, I. M., Casey, J. M., Darling, E. S., Lefcheck, J. S., and Duffy, J. E., 2019(b). Coral reef ecosystem functioning: eight core processes and the role of biodiversity. *Frontiers in Ecology and the Environment*, 17 (8), 445–454.
Bufflap, S. E. and Allen, H. E., 1995. Sediment pore water collection methods for trace metal analysis: a review. *Water research*, 29 (1), 165–177.
Burt, J., Bartholomew, A., Bauman, A., Saif, A., and Sale, P. F., 2009. Coral recruitment and early benthic community development on several materials used in the construction of artificial reefs and breakwaters. *Journal of Experimental Marine Biology and Ecology*, 373 (1), 72–78.
Bythell, J. C., 1988. A nitrogen budget for the Caribbean elkhorn coral Acropora palmata (lamarck) from the back-reef environment of Tague Bay reef, St. Croix, US Virgin Island *(PhD Thesis)*.
Carr, M. H. and Hixon, M. A., 1997. Artificial reefs: the importance of comparisons with natural reefs. *Fisheries*, 22 (4), 28–33.
Chaves, L. T. C., Pereira, P. H. C., and Feitosa, J. L. L., 2013. Coral reef fish association with macroalgal beds on a tropical reef system in North-eastern Brazil. *Marine and Freshwater Research*, 64 (12), 1101–1111.
Chen, J.-L., Chuang, C.-T., Jan, R.-Q., Liu, L.-C., and Jan, M.-S., 2013. Recreational benefits of ecosystem services on and around artificial reefs: a case study in Penghu, Taiwan. *Ocean & coastal management*, 85, 58–64.
Claar, D. C., Szostek, L., McDevitt-Irwin, J. M., Schanze, J. J., and Baum, J. K., 2018. Global patterns and impacts of El Niño events on coral reefs: A meta-analysis. *PloS one*, 13 (2), e0190957.
Clua, E., Legendre, P., Vigliola, L., Magron, F., Kulbicki, M., Sarramegna, S., Labrosse, P., and Galzin, R., 2006. Medium scale approach (MSA) for improved assessment of coral reef fish habitat. *Journal of Experimental Marine Biology and Ecology*, 333 (2), 219–230.
Crawley M.J. 2007. The R book, Wiley: Chichester, UK.
Damar, A., Hesse, K.-J., Colijn, F., and Vitner, Y., 2019. The eutrophication states of the Indonesian sea large marine ecosystem: Jakarta Bay, 2001–2013. *Deep Sea Research Part II: Topical Studies in Oceanography*, 163, 72–86.

Davies, T. K., Mees, C. C., and Milner-Gulland, E. J., 2014. The past, present and future use of drifting fish aggregating devices (FADs) in the Indian Ocean. *Marine policy*, 45, 163–170.
D'Elia, C.F., 1977. The uptake and release of dissolved phosphorus by reef corals 1, 2. Limnology and Oceanography, 22(2), pp.301-315.
Díaz-Pérez, L., Rodríguez-Zaragoza, F. A., Ortiz, M., Cupul-Magaña, A. L., Carriquiry, J. D., Ríos-Jara, E., Rodríguez-Troncoso, A. P., and García-Rivas, M. del C., 2016. Coral reef health indices versus the biological, ecological and functional diversity of fish and coral assemblages in the Caribbean Sea. *PloS one*, 11 (8), e0161812.
Domart-Coulon, I. J., Traylor-Knowles, N., Peters, E., Elbert, D., Downs, C. A., Price, K., Stubbs, J., McLaughlin, S., Cox, E., and Aeby, G., 2006. Comprehensive characterization of skeletal tissue growth anomalies of the finger coral Porites compressa. *Coral Reefs*, 25, 531–543.
El-Naggar, H. A., 2020. Human impacts on coral reef ecosystem. *In*: *Natural Resources Management and Biological Sciences*. IntechOpen.
Ezzat, L., Lamy, T., Maher, R. L., Munsterman, K. S., Landfield, K., Schmeltzer, E. R., Gaulke, C. A., Burkepile, D. E., and Thurber, R. V., 2019. Surgeonfish feces increase microbial opportunism in reef-building corals. *Marine Ecology Progress Series*, 631, 81–97.
Fabi, G., Luccarini, F., Panfili, M., Solustri, C., and Spagnolo, A., 2002. Effects of an artificial reef on the surrounding soft-bottom community (central Adriatic Sea). *ICES Journal of Marine Science*, 59 (suppl), S343–S349.
Falcão, M., Santos, M. N., Vicente, M., and Monteiro, C. C., 2007. Biogeochemical processes and nutrient cycling within an artificial reef off Southern Portugal. *Marine Environmental Research*, 63 (5), 429–444.
Harris, L. E., 2009. Artificial reefs for ecosystem restoration and coastal erosion protection with aquaculture and recreational amenities. *Reef Journal*, 1 (1), 235–246.
Herbert, R. J. H., Collins, K., Mallinson, J., Hall, A. E., Pegg, J., Ross, K., Clarke, L., and Clements, T., 2017. Epibenthic and mobile species colonisation of a geotextile artificial surf reef on the south coast of England. *PloS one*, 12 (9), e0184100.
Heron, S. F., Eakin, C. M., Douvere, F., Anderson, K. L., Day, J. C., Geiger, E., Hoegh-Guldberg, O., Van Hooidonk, R., Hughes, T., and Marshall, P., 2017. Impacts of climate change on World Heritage coral reefs: a first global scientific assessment.
Holmström, C. and Kjelleberg, S., 1994. The effect of external biological factors on settlement of marine invertebrate and new antifouling technology. *Biofouling*, 8 (2), 147–160.
Honjo, S. and Doherty, K. W., 1988. Large aperture time-series sediment traps; design objectives, construction and application. *Deep Sea Research Part A. Oceanographic Research Papers*, 35 (1), 133–149.
Hughes, T. P., Kerry, J. T., Baird, A. H., Connolly, S. R., Dietzel, A., Eakin, C. M., Heron, S. F., Hoey, A. S., Hoogenboom, M. O., and Liu, G., 2018. Global warming transforms coral reef assemblages. *Nature*, 556 (7702), 492–496.
Folpp, H., Lowry, M., Gregson, M., and Suthers, I. M., 2013. Fish assemblages on estuarine artificial reefs: natural rocky-reef mimics or discrete assemblages? *PLoS One*, 8 (6), e63505.
Fonseca, M. S. and Kenworthy, W. J., 1987. Effects of current on photosynthesis and distribution of seagrasses. *Aquatic botany*, 27 (1), 59–78.
Froese, R. and Pauly, D., 2000. FishBase 2000: concepts, design and data sources. ICLARM, Los Ban os. *Laguna*.
Gattuso, J.-P., Frankignoulle, M., and Wollast, R., 1998. Carbon and carbonate metabolism in coastal aquatic ecosystems. *Annual Review of Ecology and Systematics*, 405–434.
Geesey, G. G., Alexander, G. V, Bray, R. N., and Miller, A. C., 1984. Fish fecal pellets are a source of minerals for inshore reef communities. *Marine ecology progress series*, 19–25.
Ghilardi, M., Salter, M. A., Parravicini, V., Ferse, S. C. A., Rixen, T., Wild, C., Birkicht, M., Perry, C. T., Berry, A., and Wilson, R. W., 2023. Temperature, species identity and morphological traits predict carbonate excretion and mineralogy in tropical reef fishes. *Nature communications*, 14

(1), 985.
Glynn, P. W., Colley, S. B., Guzman, H. M., Enochs, I. C., Cortés, J., Maté, J. L., and Feingold, J. S., 2011. Reef coral reproduction in the eastern Pacific: Costa Rica, Panamá, and the Galápagos Islands (Ecuador). VI. Agariciidae, Pavona clavus. *Marine biology*, 158, 1601–1617.
Godoy, E. A. S., Almeida, T. C. M., and Zalmon, I. R., 2002. Fish assemblages and environmental variables on an artificial reef north of Rio de Janeiro, Brazil. *ICES Journal of Marine Science*, 59 (suppl), S138–S143.
Graham, N. A. J., Jennings, S., MacNeil, M. A., Mouillot, D., and Wilson, S. K., 2015. Predicting climate-driven regime shifts versus rebound potential in coral reefs. *Nature*, 518 (7537), 94–97.
Griffiths, J. R., Kadin, M., Nascimento, F. J. A., Tamelander, T., Törnroos, A., Bonaglia, S., Bonsdorff, E., Brüchert, V., Gårdmark, A., and Järnström, M., 2017. The importance of benthic–pelagic coupling for marine ecosystem functioning in a changing world. *Global change biology*, 23 (6), 2179–2196.
De Goeij, J. M., Lesser, M. P., and Pawlik, J. R., 2017. Nutrient fluxes and ecological functions of coral reef sponges in a changing ocean. *In*: *Climate change, ocean acidification and sponges*. Springer, 373–410.
De Goeij, J. M., Van Oevelen, D., Vermeij, M. J. A., Osinga, R., Middelburg, J. J., De Goeij, A. F. P. M., and Admiraal, W., 2013. Surviving in a marine desert: the sponge loop retains resources within coral reefs. *Science*, 342 (6154), 108–110.
Graham, N. A. J., Wilson, S. K., Carr, P., Hoey, A. S., Jennings, S., and MacNeil, M. A., 2018. Seabirds enhance coral reef productivity and functioning in the absence of invasive rats. *Nature*, 559 (7713), 250–253.
Grover, R., Maguer, J.-F., Reynaud-Vaganay, S., and Ferrier-Pages, C., 2002. Uptake of ammonium by the scleractinian coral Stylophora pistillata: effect of feeding, light, and ammonium concentrations. *Limnology and oceanography*, 47 (3), 782–790.
Gust, G., Michaels, A. F., Johnson, R., Deuser, W. G., and Bowles, W., 1994. Mooring line motions and sediment trap hydromechanics: in situ intercomparison of three common deployment designs. *Deep Sea Research Part I: Oceanographic Research Papers*, 41 (5–6), 831–857.
Den Haan, J., Huisman, J., Brocke, H. J., Goehlich, H., Latijnhouwers, K. R. W., Van Heeringen, S., Honcoop, S. A. S., Bleyenberg, T. E., Schouten, S., and Cerli, C., 2016. Nitrogen and phosphorus uptake rates of different species from a coral reef community after a nutrient pulse. *Scientific reports*, 6 (1), 1–13.
Hall, A. E., Herbert, R. J. H., and Stafford, R., 2021. Temporal and spatial variation in adult and juvenile mobile fauna associated with natural and artificial coastal habitats. *Marine Biology*, 168 (2), 1–13.
Hammond, M., Bond, T., Prince, J., Hovey, R. K., and McLean, D. L., 2020. An assessment of change to fish and benthic communities following installation of an artificial reef. *Regional Studies in Marine Science*, 39, 101408.
Harrison, A. and Hall, A., 2021. *Poole Bay dive surveys 2021: Sediment traps and photo-quadrats. Bournemouth University Internal Report.*
Hatcher, A. I. and Frith, C. A., 1985. The control of nitrate and ammonium concentrations in a coral reef lagoon. *Coral Reefs*, 4, 101–110.
Higgins, E., Scheibling, R. E., Desilets, K. M., and Metaxas, A., 2019. Benthic community succession on artificial and natural coral reefs in the northern Gulf of Aqaba, Red Sea. *PloS one*, 14 (2), e0212842.
Howard, J., McLeod, E., Thomas, S., Eastwood, E., Fox, M., Wenzel, L., and Pidgeon, E., 2017. The potential to integrate blue carbon into MPA design and management. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 27, 100–115.
Howard, J., Sutton-Grier, A., Herr, D., Kleypas, J., Landis, E., Mcleod, E., Pidgeon, E., and Simpson, S., 2017. Clarifying the role of coastal and marine systems in climate mitigation. *Frontiers in Ecology and the Environment*, 15 (1), 42–50.

Hudatwi, M., Syari, I. A., Utami, E., Nugraha, M. A., Akhrianti, I., and Pamungkas, A., 2021. Diversity of Benthic Organisms on Artificial Reef Structure. *In*: *IOP Conference Series: Earth and Environmental Science*. IOP Publishing, 12033.

Huettel, M., Berg, P., and Kostka, J. E., 2014. Benthic exchange and biogeochemical cycling in permeable sediments. *Annual Review of Marine Science*, 6, 23–51.

Isbell, F., Gonzalez, A., Loreau, M., Cowles, J., Díaz, S., Hector, A., Mace, G. M., Wardle, D. A., O'Connor, M. I., and Duffy, J. E., 2017. Linking the influence and dependence of people on biodiversity across scales. *Nature*, 546 (7656), 65–72.

Israel, D., Gallo, C., and Angel, D. L., 2017. Benthic artificial reefs as a means to reduce the environmental effects of cod mariculture in Skutulsfjörður, Iceland. *Marine Biodiversity*, 47 (2), 405–411.

Jewett, S. C., Brewer, R., Chenelot, H., Clark, R., Dasher, D., Harper, S., and Hoberg, M., 2008. Scuba techniques for the Alaska Monitoring and Assessment Program (AKMAP) of the Aleutian Islands, Alaska. *In*: *Diving for Science 2008. Proceedings of the 27th American Academy of Underwater Sciences Symposium, Dauphin Island, AL: AAUS*. 71–89.

Kenyon, T. M., Doropoulos, C., Dove, S., Webb, G. E., Newman, S. P., Sim, C. W. H., Arzan, M., and Mumby, P. J., 2020. The effects of rubble mobilisation on coral fragment survival, partial mortality and growth. *Journal of Experimental Marine Biology and Ecology*, 533, 151467.

Kohler, K.E. and Gill, S.M., 2006. Coral Point Count with Excel extensions (CPCe): A Visual Basic program for the determination of coral and substrate coverage using random point count methodology. Computers & geosciences, 32(9), pp.1259-1269.

Kojansow, Jo., Sompie, D., Emor, D., and Rondonuwu, A. B., 2013. Fish settlement on reefballs artificial reef and natural coral reef at Buyat Bay and surrounding areas, North Sulawesi, Indonesia. *Galaxea, Journal of Coral Reef Studies*, 15 (Supplement), 229–237.

Komyakova, V., Chamberlain, D., Jones, G. P., and Swearer, S. E., 2019. Assessing the performance of artificial reefs as substitute habitat for temperate reef fishes: Implications for reef design and placement. *Science of the total environment*, 668, 139–152.

Lafferty, K. D., Benesh, K. C., Mahon, A. R., Jerde, C. L., and Lowe, C. G., 2018. Detecting southern California's white sharks with environmental DNA. *Frontiers in Marine Science*, 5, 355.

Larned, S. T., 1998. Nitrogen-versus phosphorus-limited growth and sources of nutrients for coral reef macroalgae. *Marine Biology*, 132 (3), 409–421.

Lee, H. Z. L., Davies, I. M., Baxter, J. M., Diele, K., and Sanderson, W. G., 2020. Missing the full story: First estimates of carbon deposition rates for the European flat oyster, Ostrea edulis. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 30 (11), 2076–2086.

Lee, T. N., Rooth, C., Williams, E., McGowan, M., Szmant, A. F., and Clarke, M. E., 1992. Influence of Florida Current, gyres and wind-driven circulation on transport of larvae and recruitment in the Florida Keys coral reefs. *Continental Shelf Research*, 12 (7–8), 971–1002.

Legendre, L. and Le Fèvre, J., 1995. Microbial food webs and the export of biogenic carbon in oceans. *Aquatic Microbial Ecology*, 9 (1), 69–77.

Leichter, J. J., Stewart, H. L., and Miller, S. L., 2003. Episodic nutrient transport to Florida coral reefs. *Limnology and Oceanography*, 48 (4), 1394–1407.

Leitão, F., 2013. Artificial reefs: from ecological processes to fishing enhancement tools. *Brazilian journal of oceanography*, 61, 77–81.

Lemoine, H. R., Paxton, A. B., Anisfeld, S. C., Rosemond, R. C., and Peterson, C. H., 2019. Selecting the optimal artificial reefs to achieve fish habitat enhancement goals. *Biological Conservation*, 238, 108200.

Lesser, M. P., 2011. Coral bleaching: causes and mechanisms. *In*: *Coral reefs: an ecosystem in transition*. Springer, 405–419.

Leujak, W. and Ormond, R. F. G., 2007. Comparative accuracy and efficiency of six coral community survey methods. *Journal of experimental marine biology and ecology*, 351 (1–2), 168–187.

Limbong, D., Kumampung, J., Rimper, J., Arai, T., and Miyazaki, N., 2003. Emissions and

environmental implications of mercury from artisanal gold mining in north Sulawesi, Indonesia. *Science of the Total Environment*, 302 (1–3), 227–236.
Lohrer, A. M., Halliday, N. J., Thrush, S. F., Hewitt, J. E., and Rodil, I. F., 2010. Ecosystem functioning in a disturbance-recovery context: contribution of macrofauna to primary production and nutrient release on intertidal sandflats. *Journal of Experimental Marine Biology and Ecology*, 390 (1), 6–13.
Lourey, M. J., Dunn, J. R., and Waring, J., 2006. A mixed-layer nutrient climatology of Leeuwin Current and Western Australian shelf waters: seasonal nutrient dynamics and biomass. *Journal of Marine Systems*, 59 (1–2), 25–51.
Mallet, D., Wantiez, L., Lemouellic, S., Vigliola, L., and Pelletier, D., 2014. Complementarity of rotating video and underwater visual census for assessing species richness, frequency and density of reef fish on coral reef slopes. *PLoS One*, 9 (1), e84344.
Mantelatto, M.C., Fleury, B.G., Menegola, C. and Creed, J.C., 2013. Cost–benefit of different methods for monitoring invasive corals on tropical rocky reefs in the southwest Atlantic. *Journal of experimental marine biology and ecology*, *449*, pp.129-134.
Mariani, S., 2003. Recruitment in invertebrates with short-lived larvae: the case of the bryozoan Disporella hispida (Fleming). *Helgoland Marine Research*, 57, 47–53.
Marinaro, J.-Y., 1995. Artificial reefs in the French Mediterranean: A critical assessment of previous experiments and a proposition in favour of a new reef-planning policy. *Biol. Mar. Mediterr.*, 2 (1), 65–76.
Montgomery, W. L and Pollak, P. E., 1988. Epulopiscium fishelsoni NG, N. Sp., a protist of uncertain taxonomic affinities from the gut of an herbivorous reef fish 1. *The Journal of protozoology*, 35 (4), 565–569.
Moriarty, D. J. W., Pollard, P. C., and Hunt, W. G., 1985. Temporal and spatial variation in bacterial production in the water column over a coral reef. *Marine Biology*, 85 (3), 285–292.
Mountfort, D. O., Campbell, J., and Clements, K. D., 2002. Hindgut fermentation in three species of marine herbivorous fish. *Applied and environmental microbiology*, 68 (3), 1374–1380.
Nelson, C. E., Wegley Kelly, L., and Haas, A. F., 2023. Microbial Interactions with Dissolved Organic Matter Are Central to Coral Reef Ecosystem Function and Resilience. *Annual Review of Marine Science*, 15, 431–460.
Nugrahadi, M. S., Yanagi, T., Tejakusuma, I. G., Adi, S., and Darmawan, R. A., 2010. Seasonal variations of nutrient budgets in Jakarta Bay, Indonesia. *Mar. Res. Indonesia*, 35 (1), 9–20.
Oren, U. and Benayahu, Y., 1997. Transplantation of juvenile corals: a new approach for enhancing colonization of artificial reefs. *Marine Biology*, 127, 499–505.
Pardo, A., 2014. A scuba diving direct sediment sampling methodology on benthic transects in glacial lakes: procedure description, safety measures, and tests results. *Environmental Science and Pollution Research*, 21 (21), 12457–12471.
Pawlik, J. R., Burkepile, D. E., and Thurber, R. V., 2016. A vicious circle? Altered carbon and nutrient cycling may explain the low resilience of Caribbean coral reefs. *BioScience*, 66 (6), 470–476.
Paxton, A. B., Shertzer, K. W., Bacheler, N. M., Kellison, G. T., Riley, K. L., and Taylor, J. C., 2020. Meta-analysis reveals artificial reefs can be effective tools for fish community enhancement but are not one-size-fits-all. *Frontiers in Marine Science*, 7, 282.
Peleg, O., Guy-Haim, T., Yeruham, E., Silverman, J., and Rilov, G., 2020. Tropicalization may invert trophic state and carbon budget of shallow temperate rocky reefs. *Journal of Ecology*, 108 (3), 844–854.
Perkol-Finkel, S. and Benayahu, Y., 2007. Differential recruitment of benthic communities on neighboring artificial and natural reefs. *Journal of Experimental Marine Biology and Ecology*, 340 (1), 25–39.
Perkol-Finkel, S., Shashar, N., Barneah, O., Ben-David-Zaslow, R., Oren, U., Reichart, T., Yacobovich, T., Yahel, G., Yahel, R., and Benayahu, Y., 2005. Fouling reefal communities on artificial reefs: does age matter? *Biofouling*, 21 (2), 127–140.

Perkol-Finkel, S., Shashar, N., and Benayahu, Y., 2006. Can artificial reefs mimic natural reef communities? The roles of structural features and age. *Marine environmental research*, 61 (2), 121–135.
Pinnegar, J. K., Polunin, N. V. C., Videler, J. J., and de Wiljes, J. J., 2007. Daily carbon, nitrogen and phosphorus budgets for the Mediterranean planktivorous damselfish Chromis chromis. *Journal of Experimental Marine Biology and Ecology*, 352 (2), 378–391.
Polunin, N.V., 1996. Trophodynamics of reef fisheries productivity. In Reef fisheries (pp. 113-135). Dordrecht: Springer Netherlands.
Precht, E. and Huettel, M., 2004. Rapid wave-driven advective pore water exchange in a permeable coastal sediment. *Journal of Sea Research*, 51 (2), 93–107.
Rädecker, N., Pogoreutz, C., Voolstra, C. R., Wiedenmann, J., and Wild, C., 2015. Nitrogen cycling in corals: the key to understanding holobiont functioning? *Trends in microbiology*, 23 (8), 490–497.
Reeds, K. A., Smith, J. A., Suthers, I. M., and Johnston, E. L., 2018. An ecological halo surrounding a large offshore artificial reef: sediments, infauna, and fish foraging. *Marine environmental research*, 141, 30–38.
Rempel, H. S., Siebert, A. K., Van Wert, J. C., Bodwin, K. N., and Ruttenberg, B. I., 2022. Feces consumption by nominally herbivorous fishes in the Caribbean: an underappreciated source of nutrients? *Coral Reefs*, 1–13.
Rizzo, W. M., 1990. Nutrient exchanges between the water column and a subtidal benthic microalgal community. *Estuaries*, 13 (3), 219–226.
Saba, G. K., Burd, A. B., Dunne, J. P., Hernández-León, S., Martin, A. H., Rose, K. A., Salisbury, J., Steinberg, D. K., Trueman, C. N., and Wilson, R. W., 2021. Toward a better understanding of fish-based contribution to ocean carbon flux. *Limnology and Oceanography*, 66 (5), 1639–1664.
dos Santos, M. N., Monteiro, C. C., and Lasserre, G., 2005. Observations and trends on the intra-annual variation of the fish assemblages on two artificial reefs in Algarve coastal waters (southern Portugal). *Scientia Marina*, 69 (3), 415–426.
Schiettekatte, N. M. D., 2021. *Fish-mediated functions on coral reefs (PhD Thesis)*.
Schmidt-Roach, S., Kunzmann, A., and Arbizu, P. M., 2008. In situ observation of coral recruitment using fluorescence census techniques. *Journal of Experimental Marine Biology and Ecology*, 367 (1), 37–40.
Schulze, A., Erdner, D. L., Grimes, C. J., Holstein, D. M., and Miglietta, M. P., 2020. Artificial reefs in the Northern Gulf of Mexico: Community ecology amid the "Ocean Sprawl". *Frontiers in Marine Science*, 7, 447.
Shantz, A. A. and Burkepile, D. E., 2014. Context-dependent effects of nutrient loading on the coral–algal mutualism. *Ecology*, 95 (7), 1995–2005.
Silbiger, N. J., Nelson, C. E., Remple, K., Sevilla, J. K., Quinlan, Z. A., Putnam, H. M., Fox, M. D., and Donahue, M. J., 2018. Nutrient pollution disrupts key ecosystem functions on coral reefs. *Proceedings of the Royal Society B*, 285 (1880), 20172718.
Silveira, C. B., Cavalcanti, G. S., Walter, J. M., Silva-Lima, A. W., Dinsdale, E. A., Bourne, D. G., Thompson, C. C., and Thompson, F. L., 2017. Microbial processes driving coral reef organic carbon flow. *FEMS microbiology reviews*, 41 (4), 575–595.
Smriga, S. P., 2010. *Ecological significance of bacteria associated with coral reef fish feces*. University of California, San Diego.
Stafford, R., Boakes, Z., Hall, A. E., and Jones, G. C. A., 2021. The role of predator removal by fishing on ocean carbon dynamics. *Anthropocene Science*, 1–7.
Steven, A. D. L. and Atkinson, M. J., 2003. Nutrient uptake by coral-reef microatolls. *Coral Reefs*, 22, 197–204.
Stone, R. B., Pratt, H. L., Parker Jr, R. O., and Davis, G. E., 1979. A comparison of fish populations on an artificial and natural reef in the Florida Keys. *Marine Fisheries Review*, 41 (9), 1–11.
Sutherland, R. A., 1998. Loss-on-ignition estimates of organic matter and relationships to organic

carbon in fluvial bed sediments. *Hydrobiologia*, 389, 153–167.
Thanner, S. E., McIntosh, T. L., and Blair, S. M., 2006. Development of benthic and fish assemblages on artificial reef materials compared to adjacent natural reef assemblages in Miami-Dade County, Florida. *Bulletin of marine Science*, 78 (1), 57–70.
Thomas, F. I. M. and Atkinson, M. J., 1997. Ammonium uptake by coral reefs: effects of water velocity and surface roughness on mass transfer. *Limnology and Oceanography*, 42 (1), 81–88.
Trap, J., Bonkowski, M., Plassard, C., Villenave, C., and Blanchart, E., 2016. Ecological importance of soil bacterivores for ecosystem functions. *Plant and Soil*, 398, 1–24.
Vicente, M., Falcão, M., Santos, M. N., Caetano, M., Serpa, D., Vale, C., and Monteiro, C., 2008. Environmental assessment of two artificial reef systems off southern Portugal (Faro and Olhão): A question of location. *Continental shelf research*, 28 (6), 839–847.
Vivier, B., Dauvin, J.-C., Navon, M., Rusig, A.-M., Mussio, I., Orvain, F., Boutouil, M., and Claquin, P., 2021. Marine artificial reefs, a meta-analysis of their design, objectives and effectiveness. *Global Ecology and Conservation*, 27, e01538.
Von Wachenfeldt, E. and Tranvik, L. J., 2008. Sedimentation in boreal lakes—the role of flocculation of allochthonous dissolved organic matter in the water column. *Ecosystems*, 11 (5), 803–814.
Watson, R. A. and Quinn Ii, T. J., 1997. Performance of transect and point count underwater visual census methods. *Ecological Modelling*, 104 (1), 103–112.
Whitmarsh, S. K., Fairweather, P. G., and Huveneers, C., 2017. What is Big BRUVver up to? Methods and uses of baited underwater video. *Reviews in Fish Biology and Fisheries*, 27 (1), 53–73.
Wild, C., 2003. Sediment-water coupling in permeable shallow water sediments with special emphasis on carbonate sands and the cycling of coral exudates in reef environments.
Wild, C., Huettel, M., Klueter, A., Kremb, S. G., Rasheed, M. Y. M., and Jørgensen, B. B., 2004. Coral mucus functions as an energy carrier and particle trap in the reef ecosystem. *Nature*, 428 (6978), 66–70.
Wild, C., Wehrmann, L. M., Mayr, C., Schöttner, S. I., Allers, E., and Lundälv, T., 2009. Microbial degradation of cold-water coral-derived organic matter: potential implication for organic C cycling in the water column above Tisler Reef. *Aquatic Biology*, 7 (1–2), 71–80.
Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A., and Smith, G. M., 2009. *Mixed effects models and extensions in ecology with R*. Springer.

Supplementary material

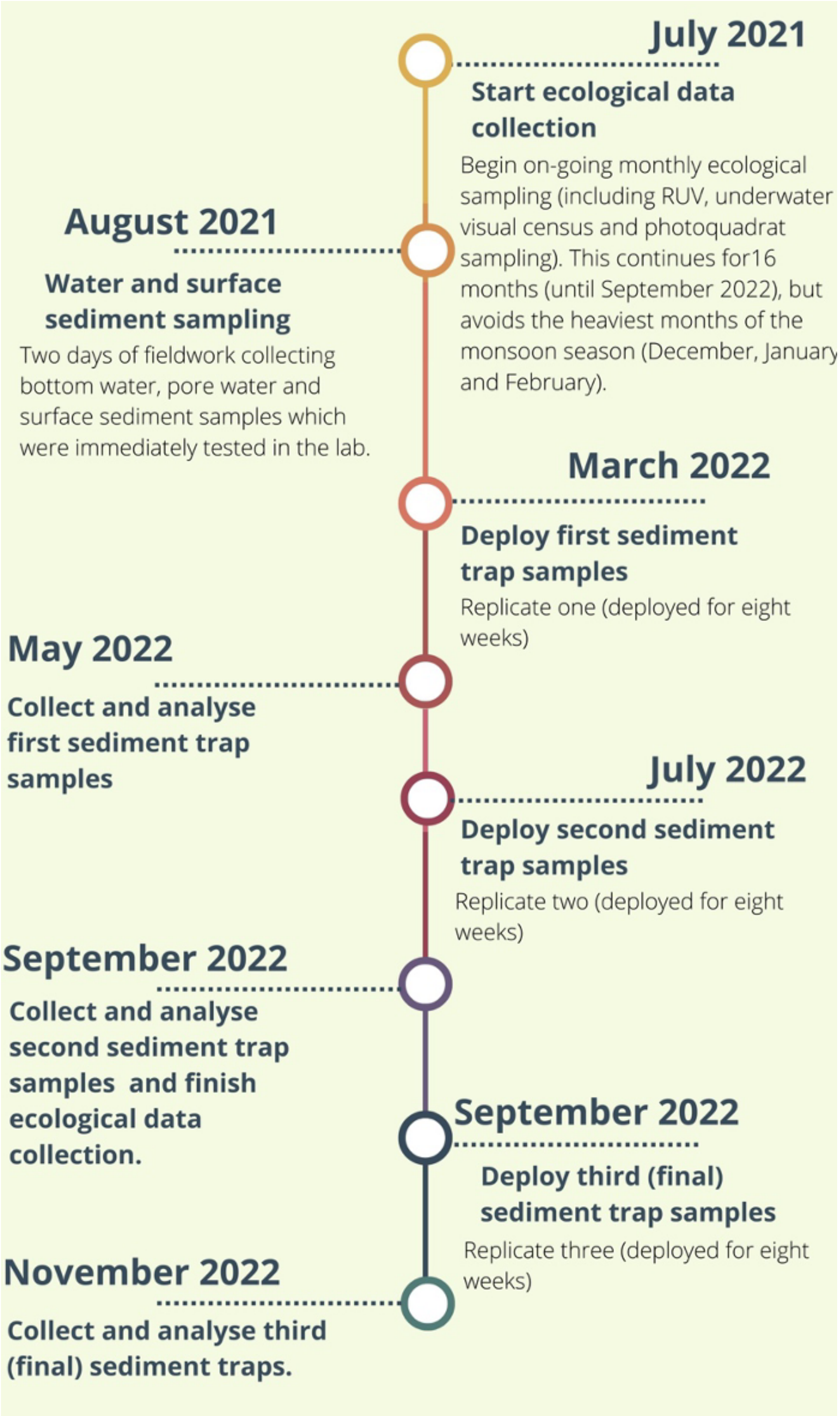

Figure S.4.1: Research timeline highlighting key fieldwork activities between July 2021 to November 2022.

Chapter 5: Is tourism helpful or harmful for coral reef health? Stakeholder assessment, actions and management measures in three reef-based tourism areas in Bali Indonesia.

Objectives: Identify how successful local stakeholders were in (1) identifying coral reef health, and (2) if relevant stakeholders were able to establish management measures which may foster a mutually beneficial relationship between the tourism economy and coral reef health

Contribution to new knowledge: Firstly, at present, the relationship between marine tourism and coral reef conservation in Bali is largely unknown, with no definitive literature specifically assessing whether tourism has helped or hindered attempts to conserve Bali's marine environment. This research aim to find out if, at the three case study sites used, reef-based tourism has been helpful or harmful for coral reef conservation in Bali. Secondly, most qualitative research has highlighted the success of conservation projects based only on hearsay from local communities (e.g. Christie (2004); Leisher et al. (2007); Pedju (2018)), with no ecological data to confirm the opinions of participants. There is limited research combining social science results with ecological data, with Stafford (2018) highlighting that there is an urgent need for multidisciplinary research in marine conservation science. Based on this, I aimed to test if local stakeholders could successfully identify reef heath, given that their actions and behaviours can be a key factor in protecting reefs on a localised scale (Boakes et al. 2023a).

How this fits in the PhD: This chapter directly addresses the part of my PhD question by assessing the socioeconomic benefits of coral reef conservation, specifically in terms of reef-based tourism (which was shown by the community to be the primary socioeconomic benefit; mapping onto objective 4).

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Abstract

Reef-based tourism's impact on coral health has had mixed outcomes, with some instances supporting conservation efforts and others posing threats and impeding protection measures. The island of Bali is Indonesia's most popular tourist destination and has been documented to host some of Indonesia's most biodiverse coral reefs on its east and north coasts. Little is known about how reef-based tourism has impacted the island's reefs and even less about how local stakeholders can identify reef health and establish necessary management measure to conserve reefs.

Our multi-disciplinary research aimed to identify how successful local stakeholders were in (1) identifying coral reef health, and (2) if relevant stakeholders were able to establish management measures which may foster a mutually beneficial relationship between the tourism economy and coral reef health. At three reef-based tourism destinations in Bali, we collected and combined ecological and qualitative data, on reef health and management. We predicted outcomes of 10 key reef management measures on coral reef health using a modified Bayesian Belief Network model. We found that (1) stakeholders could successfully identify the health of their coral reefs and (2) establish various management measures which were predicted to lead to positive outcomes for coral reef health, notably: (a) Constructing artificial reefs funded through volunteer-tourism, (b) Establishing public and private environmental regulations, and (c) Engaging stakeholders with coral reef conservation objectives. Through biological surveys we found reef Health Scores varied between locations, and those with lower reef health were found to lack management measures, whilst those with higher reef health were found to have established measures, which to a large extent, had facilitated a mutualistic relationship between tourism and coral reef health. These areas with high reef health indices are positive examples of how low-volume, well managed reef-based tourism can lead to a 'win/ win' for coral reef health and tourism economy. With a bleak projected future for global coral reef health, stakeholders in marine tourism areas should aim to utilise the

opportunity provided by tourism to conserve reefs, whilst establishing locally-supported management measures that safeguard the marine environment.

Key words: Coral reef conservation, Reef-based tourism, Bayesian Belief Network Model, Multidisciplinary research, Stakeholder actions and management measures, Indonesia

Introduction

Coral reefs provide a multitude of vital ecosystem services, including food provision, shoreline protection, and support for tourism (Costanza et al. 2014) . These ecosystem services have been estimated at a value of over US $1 trillion globally, with reef tourism alone to be worth US $35.8 billion/year (Principe et al. 2012; Woodhead et al. 2019). However, coral reefs and their associated ecosystem services are declining at rates unprecedented in human history (Andrello et al. 2021; IPCC 2021) as a consequence of human-induced threats such as climate change, overfishing, unsustainable tourism practice and pollution (Burke et al. 2012; Hughes et al. 2017).

Small-scale management measures may be established to protect, conserve and/or restore[*] the marine environment on a localised scale (Stewart et al. 2020). Despite not addressing the global issue of coral decline, such actions can to help to preserve (or restore - in cases where reefs have been previously degraded) ecosystem services, thus supporting communities who depend on their local reefs for their livelihoods (Hein et al. 2021; Hughes et al. 2023). However, the practical implementation of coral reef conservation and restoration is often problematic due to its lack of funding (Bos et al. 2015). The conservation of the marine environment is typically under-funded by governments around the world, with only 4% of global public expenditure on biodiversity estimated to be directed towards marine biodiversity (Braithwaite et al. 2022, OECD 2020). Conserving coral reefs requires a great deal of human resources and is expensive due to the equipment and time required (Bellwood et al. 2019). Although the exact figure varies depending on costs and the tools used, the annual average cost of restoring 1 hectare of coral reef habitat is estimated at USD

[*] The term ‘ecological restoration' is defined as "the action of aiding the recovery of a degraded, damaged, or destroyed ecosystem" (SER 2004), whereas ‘ecological conservation' encompasses a wider process that involves both preservation and protection (Parsons et al. 2017). We used both terms throughout, with ‘restoration’ being used in the context of ‘pro-active’ reef recovery techniques (e.g. artificial reefs deployment) and ‘conservation’ referring to both pro-active and ‘reactive’ reef recovery (e.g. marine protected area establishment).

$117,000 (Bayraktarov et al. 2019, Stewart-Sinclair et al. 2021), so it is therefore necessary to explore all potential sources to fund this work (Brathwaite et al. 2022, Howlett et al. 2022).

Marine tourism is a key and rapidly growing area of the Blue Economy (Papageorgiou 2016). More specifically, 'reef-based' tourism involves activities focused directly on coral reefs, such as SCUBA diving, snorkelling, glass-bottom boat tours and reef wildlife watching (The Nature Conservancy 2017). Research has linked marine tourism with improved economic opportunities in areas with high unemployment (Garrod et al. 2003, Shani et al. 2012), and thus has shown its potential in bringing some of the world's poorest regions out of poverty (Chok et al. 2007, Byczek 2011). Development of well managed tourism can generate new jobs, which can offer higher wages, better job satisfaction and improved career opportunities compared to previous employment (Choy 1995), thus leading to a higher perceived quality of life (Diedrich 2007). Furthermore, the additional money generated by tourism, has been shown to be a reliable source of funding for coral reef conservation (Suggett et al. 2023). There is also the potential to engage reef-based tourism operators in becoming 'stewards' of their reefs (Howlett et al. 2022). Brathwaite et al. (2022) discussed how the marine tourism should provide contributions to the conservation of localised coral reefs, especially those that directly take advantage of their associated ecosystem services (such as hoteliers that enjoy stable beaches, or snorkel/ dive operators whose incomes are derived from reef tours). Limited research currently exists which assesses the link between tourism and coral reef conservation, although a study by Diedrich et al. (2007) on marine tourism in Belize did highlight a positive correlation between tourism and conservation awareness and support, with local communities believing tourism is helping to improve coral health. Reef-based tourism has been shown to contribute to the shaping of sustainable community outcomes, and when under conditions of good governance, can promote awareness, interest and financial capital to conserve reefs (Eider et al. 2023).

While marine tourism holds potential benefits, it may also threaten coral reefs and hinder efforts to conserve them. In some cases SCUBA diving and snorkelling has been shown to lead to negative ecological consequences (Haddock-Fraser and Hampton 2012), especially in terms of the physical damage caused to corals when recreational users lack experience or respect for environmentally conscious practices (Davenport and Davenport 2006; Gladstone et al. 2013). Unregulated tourism may also cause additional pollution, damage from boat anchoring and sedimentation from coastal erosion and over-development (Diedrich 2007)). Research has also highlighted that the feeding, reproduction and resting of mobile marine organisms (e.g. fish, turtles, seahorses) may be negatively affected by regular disturbances from tourists like divers and snorkellers (Rudd and Tupper 2002;

Hayes et al. 2017; Giglio et al. 2019), but to a certain extent, it is not yet known if marine tourism can threaten these organisms on a population level (Titus et al. 2015; Hayes et al. 2017).

Other studies have highlighted that well managed marine tourism activities cause very little damage to reefs and their associated communities, especially when compared to the threats caused by pollution and overfishing (Piskurek 2001; Harriott 2004). The term 'ecotourism' describes a type of tourism which strives to be more socially and environmentally sustainable than 'standard' tourism (Orams 1995, 2002). Tourism that is managed with the aim of minimising ecological impacts and promoting restoration may be considered 'eco', however, there is no universal agreement of what constitutes 'ecotourism' (Orams 1995) and consequently providers may convey false and/or misleading information about the environmentally friendliness of the product or service they offer.

Reef-based tourism in Bali, Indonesia

Indonesia is a low-middle income country (Sujarwoto et al. 2018) which hosts coral reefs known for high biodiversity (Boakes et al. 2022b) that make up over 12 % of the world's total coral reef area (Susiloningtyas et al. 2018). Indonesia is reported to have the third-largest tourism economy in Southeast Asia (Statistik 2019, as cited by Tranter et al. 2022). Marine tourism makes up 35% of Indonesia's tourism sector, which is driven to a large extent (55%) by SCUBA diving (Tranter et al. 2022). Bali is a province of Indonesia which is located inside the 'Coral Triangle', a region in the Indo-pacific which is recognised as the global centre of marine biodiversity (Allen 2008). Bali has widespread coral reefs across its east and north coasts, which have been designated as a priority for coral conservation in Indonesia (Turak and DeVantier 2011). Its coral reefs have the second highest fish biodiversity in the Asia-pacific (Mustika and Ratha 2013), with over 805 documented reef fish species (Allen and Erdmann 2013). Bali's reefs represent a large proportion of the US$ 1991 billion annual tourism value from Indonesian coral reefs (Spalding et al. 2017) due to the popular dive sites on its north and east coasts.

Data collected in 2017 highlighted that 50% of Bali's corals are in good health, whilst 20% are declining in health and 30% are in poor condition (Marine and Fisheries Office 2017 data, as cited by Wicaksana (2020)). The degradation of Bali's reefs has been primarily associated with coral bleaching (Prasetia et al. 2017; Suparno et al. 2019; Tito et al. 2019), destructive fishing practices (Doherty et al. 2013; Frey and Berkes 2014) and marine pollution (Germanov et al. 2019, Suteja et al. 2021). Following the decline of coral reef health in Indonesia, coral reef conservation programmes have been initiated by community groups, international non-government organisations and the government (both on a local and central-government scale). A large proportion of these programmes are based in Bali (Boakes et al. 2022b), and most focus on artificial reef deployment (Wicaksana 2020, Boakes et al. 2022a), establishing marine protected areas (Pedju 2018) and/or developing ecotourism destinations which promote coral reef conservation (Trialfhianty 2017).

Given that Bali's economy is driven primarily by tourism (68% of its GDP; Antara and Sumarniasih 2017), it would be expected that this sector would provide important contributions to coral reef conservation, however there is currently limited literature in Bali (and wider Indonesia) which documents this. At present, the relationship between marine tourism and coral reef conservation in Bali is largely unknown, with no definitive literature specifically assessing whether tourism has helped or hindered attempts to conserve Bali's marine environment. Suparno et al. (2019) highlighted that SCUBA diving activities across the island have been correlated with structural damage of coral reefs and Doherty et al. (2013) attributed reef-based tourism with a substantial increase in broken and upturned corals in Bali Barat National Park dive sites. Furthermore, research has shown that scuba dive tourism at one of Bali's most well-known dive sites, 'Manta Bay', was poorly managed by dive centres, and observed that recreational users were frequently reported to have stepped on or collided with corals (Gerugan and Chia 2020). In contrast, Boakes et al. (2022b), highlighted examples of ecotourism projects contributing to marine conservation include the 'Turtle Conservation and Education Centre' (TCEC) in Serangan Island, the 'North Bali Reef Conservation' (NBRC) in Karangasem and the BioRock® reef restoration in Pemutraran, Buleleng. Ecotourism programmes in Bali, especially ones which involve ecological restoration, are more likely to be successful if they engage local stakeholders, and establish regulations which are agreed by the community (Wardana 2019; Yunitawati and Clifton 2019). A strong factor influencing the decisions of stakeholders to support reef conservation in Bali was their general understanding and knowledge of the reef system, driven by information passed down from local leaders, as well as through educational programmes and social media. Stakeholder knowledge of a reef being degraded, as well as their education on the importance of coral reefs led them to finding solutions to safeguard their coral reef (such as deploying artificial reefs or establishing protective regulations; Boakes et al. 2023a).

But how well do stakeholders actually know their health of their coral reefs and local marine ecosystem? The utilisation of stakeholders as participants in social research can provide rich information for researchers (Brown et al. 2001). Most qualitative research has highlighted the success of conservation projects based only on hearsay from local communities (e.g. Christie 2004; Leisher et al. 2007; Pedju 2018), with no ecological data to confirm the opinions of participants.

There is limited research combining social science results with ecological data, with Stafford (2018) highlighting that there is an urgent need for multidisciplinary research in marine conservation science. Based on this, it is important to test if local stakeholders can successfully identify reef heath, given that their actions and behaviours can be a key factor in protecting reefs on a localised scale (Boakes et al. 2023a). Our multi-disciplinary research aimed to identify the success of local stakeholders in identifying coral reef health, and if they were able to initiate management measures which provide a 'win/win' for both tourism economy and coral reef health.

Methodology

This study utilised a three-stage methodological framework which included:

1) **Collection of ecological data:** At three reef-based tourism areas in Bali. This data was quantified using the Reef Health Index (Following Díaz-Pérez et al. (2016)), which compared four key parameters of reef health and gave each area a 'Reef Health Score' used for comparison between sites.

2) **Collection of qualitative data:** Focus group discussions and semi-structured one-to-one interviews were conducted at the same three reef-based tourism areas. This qualitative data allowed participants opinions of the health of their reefs to be directly compared to the results of the above (1). It also allowed us to identify key management scenarios which local stakeholders implement in reef-based tourism areas.

3) **Quantification of data through a Bayesian Belief Network (BBN) model:** Following Stafford et al. (2020), a predictive BBN was created. This network predicted the outcomes of the previously identified reef management scenarios on coral reef health as well as a suite of social and economic outcomes.

This study was based in three reef-based tourism areas in Bali: Tianyar Village, Kalibukbuk Village and the Nusa Penida Marine Protected Area (MPA; figure 5.1). Each area hosted at least one coral conservation site, although conservation approaches and funding sources differed between sites.

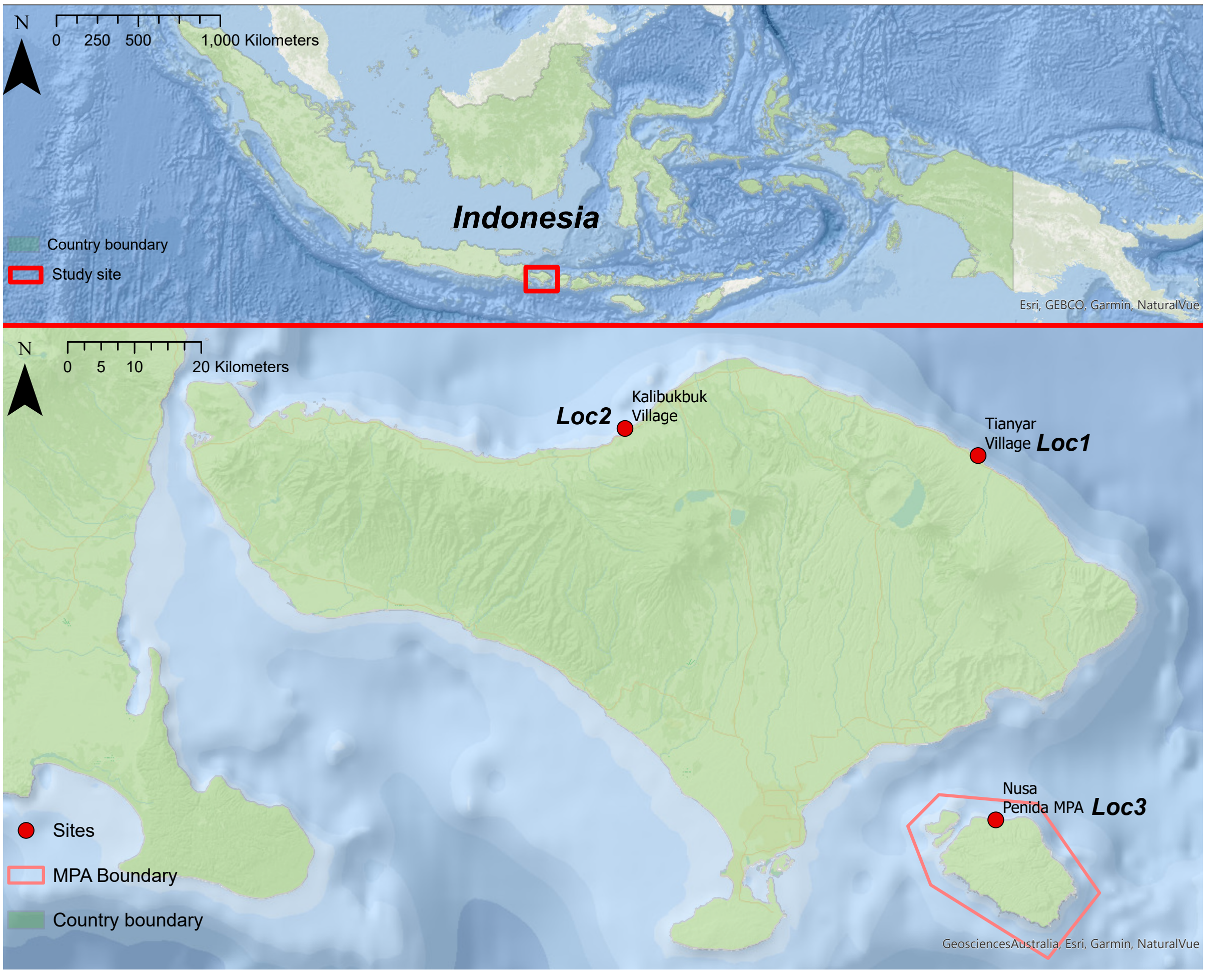

Figure 5. 1: The island of Bali within Indonesia and the location of the three case study sites within Bali. This included location one (-8.191289, 115.498080) in Tianyar Village, Karangasem ('Loc1'), location two (-8.155207, 115.023220) in Kalibukbuk Village, Buleleng ('Loc2') and location three (-8.675900, 115.521856) in the Nusa Penida MPA ('Loc3'). Note: coordinates provided are those of the reef site studied within the location).

Location 1: Tianyar Village in Karangasem District (north-east Bali) had an estimated population of 10,000 (*information derived from personal communication with village leaders).* Primary job occupations within the village were dominated by fishing and the selling of fish, with only a small percentage from tourism. The village received a relatively low number of tourists compared to others in Bali, and its small tourism industry was driven mostly by the volunteer programmes 'North Bali Reef Conservation'. Volunteers at this programme work with a local fisher group 'Yowana Bhakti Segara' to build and deploy ARs, with the aim of restoring a coral reef which was previously degraded, as a result of coral harvesting and destructive fishing practices. At the time of data collection, the volunteer programme had built approximately 15,000 artificial reef units, which were thought to make up one of the largest artificial reefs in Indonesia.

Location 2: Kalibukbuk Village in Buleleng District (north Bali) had an estimated population of 6,000 (information derived from personal communication with village leaders). Despite being a relatively small village, Kalibukbuk hosted Lovina Beach, a popular tourist destination which was well known for marine tourism activities including dolphin watching trips and snorkelling experiences. Primary job occupations in Desa Kalibukbuk were associated with the tourism industry, and included hotel and restaurant staff, snorkelling guides, boat drivers and taxi drivers. The village hosted a small community artificial reef deployment project which had received government funding as part of the 'Indonesian Coral Reef Garden' programme in 2021-2022.

Location 3: Nusa Penida, Lembongan and Ceningan (the Nusa Penida MPA) in Klungkung District (south-east Bali) had a combined population of approximately 50,000 (*information derived from personal communication with the local government)* across all three of the small-medium size islands. They were a short 30 minute boat journey from mainland Bali's east coast. They hosted some of Indonesia's most well-known dive sites, known for their diverse fish communities and abundant megafaunal populations (Allen and Erdmann 2008; Ruchimat et al. 2013; Prasetyo et al. 2019). The majority of the islands' economy was driven by its thriving tourism industry (*information derived from personal communication with the local government),* which has been criticised due to its consequential impact on the marine environment (Sudipa et al. 2020; Mustika et al. 2021). As a potential solution, multiple marine restoration programmes have been established across the islands, and a 20,000+ hectare marine protected area (MPA) was officially established in 2010 (decree no. 12/2010; Ruchimat et al. 2013). The topics of 'conservation of marine biodiversity' and 'reef-based/ marine tourism' have both been well researched on the islands, with multiple publications on each topic, however there is no known research which has assessed the link between the two.

Stage 1: Ecological data collection

There are a lack of studies in marine social science which also incorporate ecological data, often leading to a potential mismatch between the opinions of respondents and accurate biological data (Stafford, 2018). Based on this, data on ecological communities in each location was collected to compare biological reef health to respondents opinions on the current state of the marine environment. Following Díaz-Pérez et al. (2016), reef health in each location was determined using the Reef Health Index (RHI), an international, quantifiable framework of measurable indicators which was developed by ‘Healthy Reefs For Healthy People’ (www.healthyreefs.org/cms/). The RHI used two benthic indicators (coral cover and fleshy macroalgal cover) which were measured using benthic photo-quadrats, and two fish indicators (herbivorous and commercial fish biomass) measured using Remote Underwater Video (RUV). In each of the three locations, permission was obtained from the local community to conduct the survey in all three locations. Each of these communities were asked to recommend one area which was popular locally for marine tourism (mostly in terms recreational dive and snorkel activities), and based on these recommendations, a sampling site in each location was chosen. Sampling for both fish biomass and benthic cover was conducted between the months of June – September, which was in the middle of Bali’s dry season and thus known for consistently calm underwater conditions and water visibility.

Remote Underwater Video Sampling

Remote Underwater Video (RUV) is a cost-effective, safe and non-destructive method (Folpp et al. 2013; King et al. 2018), which was used to obtain estimates on the biomass of herbivorous and commercial fish (following Díaz-Pérez et al. 2016). In each sampling site (one per location, three in total), a GoPro Hero 7 HD 1080p underwater camera, fixed to a weighted unit, was placed on a sand-bottom patch which ranged between 4-6m depth. The camera was deployed 2m away from the coral substrata (example highlighted in figure 5.2) and recorded for 25 minutes at a time (following Boakes et al. 2022a). Following the methods of Hall et al. (2021), recordings were taken only taken on days with small/no waves, little/no wind and an average water visibility of 15m. Following the methods of Boakes et al. (2022a), 20 minute recording were taken from the same site 3 times (N = 3) on varying tidal conditions at different times of day (between 8am-3pm). As a relative measure of abundance, the maximum number of individuals seen in any frame (herein MaxN; following Whitmarsh et al., 2017) during each 20 minute video was recorded, which was later be used to calculate biomass of herbivorous and commercial fish for a RHI score. Videos were then analysed following the methods of Boakes et al. (2022a).

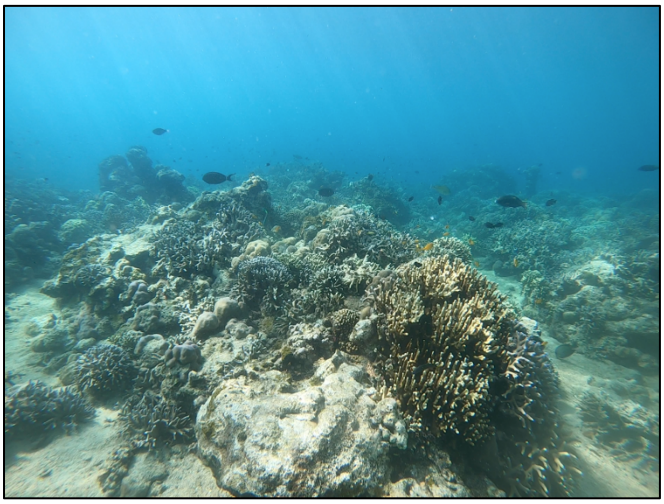

Figure 5. 2: Screenshot taken from a remote underwater video recording in Kalibukbuk (location two).

Photo-quadrat sampling

Photo-quadrat sampling, a method frequently used to examine benthic communities on coral reefs (Leujak and Ormond 2007), was used to obtain estimates on coral cover and fleshy macroalgal cover. Following Clua et al. (2006) and Chaves et al. (2013), 40cm^2 quadrats were positioned randomly along fixed 50m line transects across each site (the same sites used for the RUV samples). Photos were taken at approximately the same depth between all locations, which ranged between 4-6m. SCUBA divers using a GoPro Hero 7 camera took 30 photo-quadrats at each location. Following the methods of Leujak and Ormond (2007), photos were taken 2m away from the coral to ensure that the whole quadrat fit into the frame (figure 5.3). Benthic species from the photographs were identified to at least family level (following Schmidt-Roach et al. 2008). Approximate coral and algal percentage cover was calculated using Coral Point Count software, following the methods of Boakes et al. (2023b).

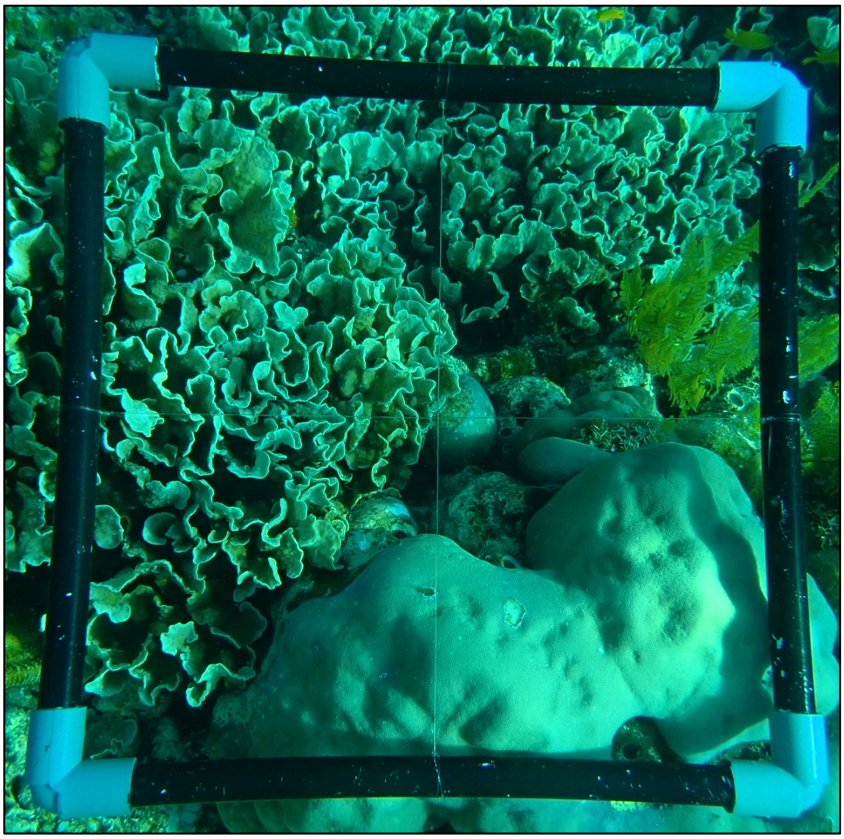

Figure 5. 3: Example of a randomly positioned photo-quadrat sample which was taken in Tianyar.

Calculating Reef Health Index (RHI) score

Unlike location one and two, location three (the Nusa Penida MPA) was a far larger case study that incorporated several reef-based tourism sites across two islands. Due to the timing constraints of this study, it was only possible to collect data in one field site per location. It was decided that data from one study site would not be representative of the many reef sites in the Nusa Penida MPA, so we also utilised the recent coral reef monitoring data (on coral cover, algal cover, as well biomass of herbivorous and commercial fish) on 14 reef sites across the Nusa Penida MPA of Kasman et al. (2021). Alongside the ecological data we collected for this study (at the Desa Ped Dive Site (-8.675900, 115.521856)), Nusa Penida, averages were taken on the data across the 14 sites, so that data was more representative of reef health across the whole location. To ensure that the data used was consistent, we compared the data we collected in the Nusa Penida MPA at the Desa Ped Dive Site, to that of Kasman et al. (2021) at the same site. This comparison showed similar results in terms of herbivorous and commercial fish biomass, as well as algal and coral cover. This supported our justification for deciding to combine both data sets for our study on the Nusa Penida MPA.

Four indicators were used to calculate RHI, including (i) average herbivorous fish biomass ($g/100m^2$) , (ii) average commercial fish biomass ($g/100m^2$), (iii) average percentage coral cover, and (iv) average percentage fleshy macroalgal cover (Díaz-Pérez et al. 2016).

From the RUV data, average MaxN was calculated across all three recordings. Then, to estimate biomass of herbivorous fish, the total average MaxN for surgeonfish (Acanthuridae), parrotfish (Scarinidae) and rabbitfish (Siganidae) was calculated (the three primary herbivorous reef fish families in Bali, described by Kasman et al (2021)). To estimate biomass of commercial fish, the total average MaxN for surgeonfish (Acanthuridae), parrotfish (Scarinidae), rabbitfish (Siganidae), fusiliers (Caesionidae), trevally (Carangidae), spadefish (Ephippidae), sweetlips (Haemulidae), chubs (Khyposidae), wrasse (Labridae), snapper (Lutjanidae), grouper (Serranidae) and barracuda (Sphyraenidae) was calculated (the 12 primary commercial reef fish families in Bali, as described by Kasman et al (2021)). Then, through utilising information available on Fishbase.se (Froese and Pauly 2000), the average weight of each fish family was obtained (presented in table S.5.2 in the supplementary material) which then allowed biomass to be calculated following Li et al. (2020)). Average biomass per RUV recording (of both herbivorous and commercial fish) was calculated by adding up each biomass value of the listed fish families. It was estimated that the field of view for each RUV recording was $50m^2$, however the RHI required biomass to be calculated as $g/100m^2$, so each biomass value was doubled.

From the photo-quadrat data, average percentage benthic cover per site was calculated by:

Total percentage cover (for each benthic family or group) / 30 (photo-quadrats per location).

These sum of these average percentage cover values (e.g average percentage cover of all corals) allowed average percentage coral cover and fleshy macroalgal cover to be calculated.

Once all biomass and percentage cover values were calculated, values were converted to an ordinal scale with values of 1 (“critical”) to 5 (“very good”), producing five grades of health (following Díaz-Pérez et al. 2016), based on the score values agreed by ‘Healthy Reefs For Healthy People’ (table S.5.3).

Stage 2: Qualitative data collection

Focus groups and one-to-one interviews

Qualitative data in the form of semi-structured interviews and multi-stakeholder focus groups discussions (FGDs) were collected with relevant stakeholders in each the three locations over a three-month period. Ethical approval for the study was obtained from Bournemouth University (ethics reference 37431). Following Wager et al. (2013), to reduce bias in the interviews as much as possible, respondents were reassured that their names and responses would remain anonymous within any research outputs. Each interviewee / focus group was given a code (e.g. interviewee 1 = I1 / focus group 1 = FG1), which was used instead of their name to ensure anonymity.

Focus group discussions were conducted in the interests of obtaining a greater range of opinions as participants in focus groups may agree and disagree any given point or opinion raised (Greenbaum 1998). Additionally, focus group discussions allowed researchers to assess if individual opinions were different in a group situation (Kellmereit 2015), and thus examine if responses were different between one to one interviews and focus groups. Following similar methods to Legare et al. (2020) and Boakes et al. (2023a) multi-stakeholder FGD were held in each of the three locations, with on average 10 attendees (which were mostly the leaders of community groups, and thus had a good knowledge of community opinions and issues). Examples of attendees included the head of the village, leaders of community groups (e.g. religion, environment, marine affairs, local law enforcement, fishing, transport, tourism, diving, education, women's association) and officials from the local government. Participants were chosen based on their positions and were approached specifically by the head of the village (of each location) to attend the FGD. All FGD discussions lasted between 2-3 hours and started with an introduction to the research, and then were asked to discuss a variety of points as a group (e.g. "tourism can be harmful to coral reef conservation", "tourism can support coral reef conservation" and "supporting the development of marine tourism"). All FGD discussions were conducted in Bahasa Indonesia, the national language of Indonesia. Key points throughout the discussion were noted by researchers who were fluent in Indonesian, and then full transcriptions and translations to English were made manually after listening back to the recordings.

Alongside the FGDs, semi-structured one-to-one interviews were conducted with individuals including dive centre managers, snorkel and dive guides, marine biologists, fishers, fish sellers, local students, international students, international tourists and international (coral conservation) volunteers. The locations of interviews depended on the participants, for instance, some participants (e.g. village leaders) requested that the interviews were held at their personal offices, whereas others (e.g. fishers) had no preference. In the instances where respondents expressed no preference, interviews were held in a private meeting room. International tourists were approached in public, in tourist destinations such as beaches, and in front of restaurants and dive resorts. Individuals were approached based on factors such as age and gender, with the aim of making the sample as diverse as possible (following Gidman et al. 2007). A full explanation of how the interviews were structured was described in full in Boakes et al. (2023a).

Sample size

Maximum variation sampling (following Patton 2014) ensured a diverse range of perspectives, with representation from major stakeholder groups across various demographics. Heads of community groups served as representatives, guaranteeing that qualitative findings reflected primary views of relevant stakeholders in each location. Initial contact, either by phone or in person, followed Kruglov and Davidson (1953), resulting in the participation of all approached individuals, except for two.

This study had 56 participants in total, which included local Indonesia citizens, ex-pats and tourists (table S.5.1). Certain studies have suggested a minimum sample size for qualitative research, including Marshall et al. (2013), who highlighted studies should contain between 15 to 30 interviews. However, instead of pre-determining a sample size, interviews can be concluded when a 'degree of saturation' (when additional interviews rarely offer new insights and information about the given topic has been reached; Mason 2010; Moura et al. 2021). For the FGDs, the sample size was limited those attending the discussion, which in all three locations was on average 10 attendees. For the one to one interviews with tourists, interviews were concluded based on when it was felt 'a degree of saturation' was reached. Once it was agreed this point was reached (and all responses from participants were similar), two more tourists were interviewed. If the answers of these additional two participants were the same or almost the same, it was decided a 'degree of saturation' had been reached, and thus it was no longer necessary to continue the interviews.

Table S.5.1 highlighted the sociodemographic characteristics of respondents. The respondents were from a wide range of countries and age groups, with considerably more males (68%) than females (32%). This was because it was more difficult to find Indonesian females willing to be interviewed. Despite having less female participants overall, it was still possible to acquire female Indonesian respondents from a diverse range of occupational and social groups (e.g. fish sellers, educational workers, students and tourism workers). In contrast, it was far easier to find female tourists/ ex-pats willing to be interviewed, so it was ensured that over half of interviewees in this group were females.

Utilisation of the qualiative data

The qualiative data was discussed in light of the quantitative data and is included in the discussion of this study. The qualitative data served three primary purposes:

1) To help the researchers understand stakeholders perceptions of reef health, which was compared to RHI scores to assess if they were able to successfully identify reef health.
2) To identify the key management measures and scenarios reported by stakeholders for the conservation of coral reefs. Researchers chose ten key reef-based management measures and scenarios which were identified by participants to have occured within their areas. Table S.5.4 (supplementary material) highlighted example supporting quotes from participants who identified these key occurances and justfied the researchers' choice of selected actions and scenarios, which were then tested for their impact on coral reef health through the Bayesian Belief Network model.
3) To inform interaction values between nodes for the model, where key quotes were presented to highlight how stakeholder opinions helped to explain a given relationship between nodes (described in full below).

Stage 3: Quantifying our results using a Bayesian Belief Network model

We quantified the findings of our qualitative and ecological data collection using a Bayesian belief network (described in full in Stafford et al. 2020), allowing us to predict the outcomes of 10 key scenarios for reef-based tourism areas for coral reef health. This network was made up of a series of 'nodes' which were connected by weighted 'edges' (circles and connecting lines, respectively; figure 5.4).

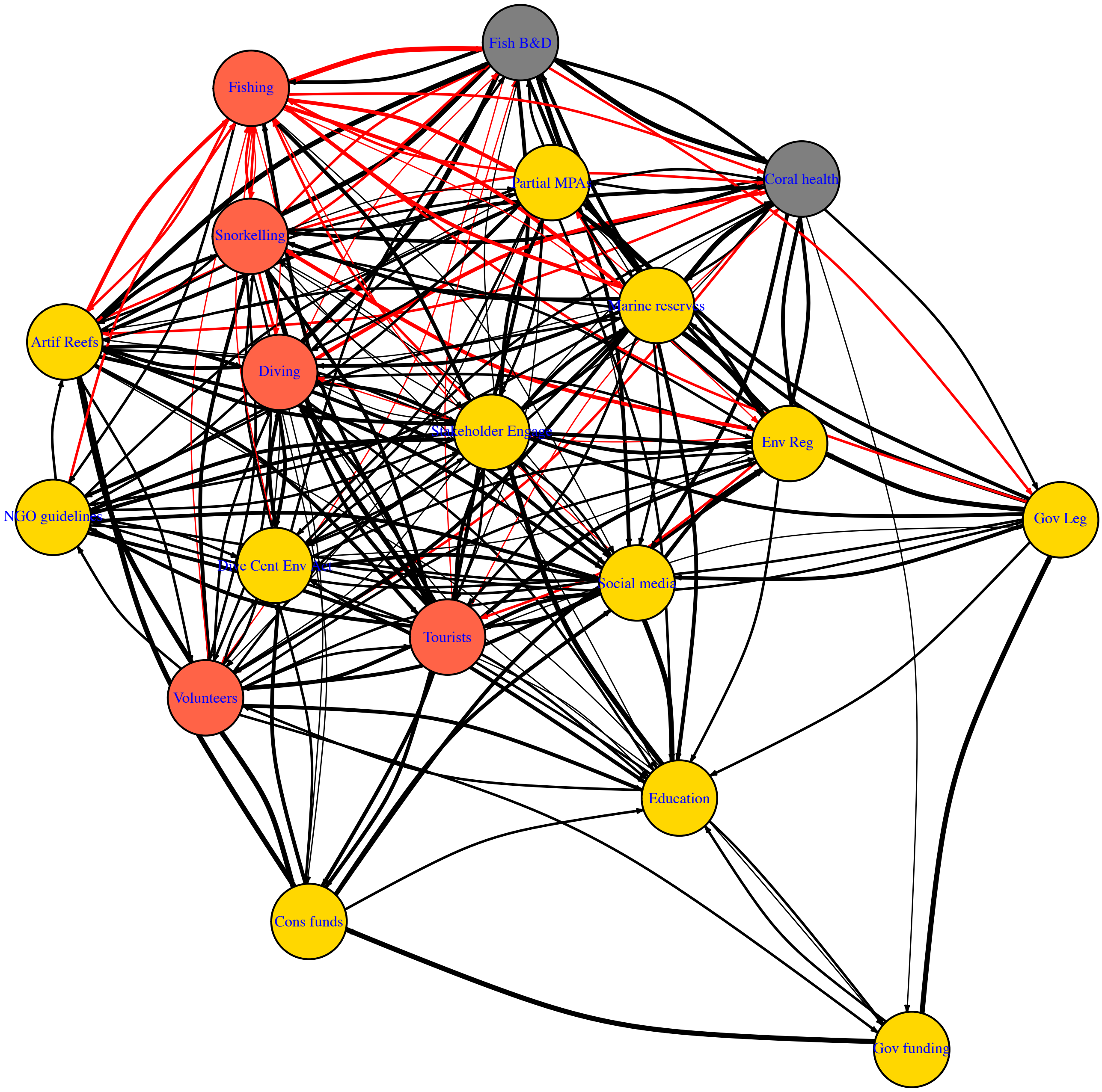

Figure 5. 4: Visualisation of the Bayesian belief network model. Circles represent nodes (broadly, grey= biological indicators, yellow= management measures, orange= ecosystem services). Red arrows represent negative interactions between nodes and black arrows positive interactions. Line thickness indicates the strength of interaction (slight, moderate and strong).

A full list of the nodes, as well as a working version of the model can be found in the supplementary material (figure S.5.1). The weights of each edge was based on changes likely to occur to receiving nodes (known as 'child' nodes) given a change in the 'parent' or originating node. These positive interactions (if the parent node increases, it is most likely that the child node will also increase) or negative interactions (if parent node increases, the child node will most likely decrease), the values give to the edges were determined from combinations of previously published literature, results of the qualitative study and expert opinion. The edge values contributing most to overall variation in fish and coral health values were determined and validated through a sensitivity analysis approach, which progress through the network, with child nodes becoming parent nodes for subsequent interactions. Certain nodes were given 'prior' values, ranging from -4 to 4, where -4 represented a strong prior decrease in the value of the node and 4 represented a strong prior increase, values closer to zero represent weaker interactions, with zero containing no prior information about whether the node should increase or decrease. The priors changed depending on the given scenarios investigated, which were highlighted as plausible actions stakeholders may take to protect coral reefs in reef-based tourism areas. To help quantify uncertainty in the network, each scenario was run 10,000 times. The first run uses the exact model as provided (and forms the circular point in the results figures), the remaining runs involve randomly selecting 10% of interactions in each run and adjusting each of them by a randomly determined amount of up to ± 0.8. 95% confidence intervals of the output of each parameter are calculated by removing the highest and lowest 2.5 % of values. The ten scenarios were given in table 5.1, and the key literature and qualitative findings which were used to justify the choice of scenarios was given in table S.5.4 in the supplementary material. The R code and priors files were also included in the supplementary material (figure S.5.3).

Table 5. 1: Scenarios implemented in the Bayesian belief network. Note: If specific changes to model priors were not highlighted in this table, it can be assumed that they remained unchanged (value stayed at 0) under each scenario.

Scenario number	Scenario	Specific changes to model priors
1	Constructing artificial reefs through volunteer-tourism	Tourists = 2 Artificial reefs = 4
2	Unregulated reef-based tourism (no specific regulations on boating, anchoring, diving etc)	Tourists = 3 Env regulations = -3
3	Reef-based tourism organisations (non-government) establishing environmental standards (such as no anchoring rules, cutting boat engine rules over dive sites etc)	Env regulations = 4 NGO guidelines = 3 Dive centre env activity = 3
4	Using social media as a tool to increase conservation awareness (e.g. Instagram conservation pages and local WhatsApp awareness groups)	Social media campaigns =4 Education = 3

5	Government established local environmental regulations	Government legislation = 4
6	Local tourism organisations contributing to marine restoration/ protection through running coral outplanting/ transplantation programmes and beach cleaning events	Dive centre env activity = 4
7	Government support for coral reef conservation (such as providing funding for artificial reef construction)	Conservation funds = 3 Artificial reefs = 3 Government funding = 4
8	Holding socialisation events to engage stakeholders with coral reef conservation and sustainable practices, and educate them on the mutual benefits of coral conservation / sustainable reef-based tourism	Education = 3 Stakeholder engagement = 4
9	Establishing marine protected areas	Partial MPAs = 4 Full marine reserves = 4
10	Pandemics causing reef-based tourism sites to temporarily close (no visiting tourists)	Diving = -3 Snorkelling = -3 Tourists = -4

Interactions between nodes, which were used to create the model, were based upon various factors including ecological data (reef health), qualitative findings (e.g. stakeholders opinions on effective management measures to conserve coral reefs) and relevant literature. Figure S.5.2 in the supplementary material highlighted the key literature and qualitative findings (in the form of participant quotes) which were used to inform interaction values between nodes for the model. For example, in determining the interaction value between the nodes 'Tourists' and 'Conservation funds', the qualitative findings revealed that tourism had provided essential funding for coral conservation (e.g. T17 highlighted "*Tourists give jobs to the locals which allows them to do conservation work*"), which was also supported by relevant literature highlighting similar findings (e.g. Boley and Green (2016) and Brathwaite et al. (2022)). This information allowed us to identify that there was a strong positive interaction between 'Tourists' and 'Conservation funds', and therefore decided that this particular interaction would be given an interaction value of 3 (highlighted in Figure S.5.2 on row 7).

Results

Reef health index

Table 5. 2: RHI scores according to limit values for each indicator, according to the grading system provided by Healthy Reefs for Healthy People Initiative (shown in table S.5.3 – supplementary material).

Location	Coral cover	Fleshy algae cover	Herbivorous fish biomass (g 100m-2)	Commercial fish biomass (g 100m-2)	Reef Health Index score (0-5)
1 – Tianyar Village	**43%** *(very good)*	**9.3%** *(fair)*	**3516** *(very good)*	**1814** *(very good)*	**4.5** *(good – very good)*
2 – Kalibukbuk Village	**34%** *(good)*	**8.9%** *(fair)*	**2109** *(fair)*	**1161** *(fair)*	**3.25** *(fair – good)*
3 – The Nusa Islands	**43%** *(very good)*	**6.3 %** *(fair)*	**4550** *(very good)*	**1456** *(good)*	**4.25** *(good – very good)*

The scores given by table 5.2 highlighted that Tianyar (location one) had the highest RHI score, which was closely followed by the Nusa Penida MPA (location three). Both had RHI scores which were between 'good' – 'very good'. Kalibukbuk (location two), had the lowest RHI score, which was towards the lower end of 'fair' –'good'. The interviewees opinions on the current state of their localised marine environment were generally supported by the RHI scores. For example, participants from Tianyar and the Nusa Penida MPA mostly agreed their coral reefs were in good - very good condition (e.g. "*Both the natural and artificial reef look amazing*" (T17) and "*The reefs here are very healthy, just like like an aquarium*" (N10), consectutively), as supported by RHI scores (table 5.1). In contrast, participants from Kalibukbuk generally commented that the reefs were in poor – fair condition (e.g. "*The reef here is not in great condition"* (K1). Our qualitative results in discussed further in light of our quantitative results in the discussion.

Figure 5.5 showed the key outcomes to 'Coral health' and 'Fish biomass and diversity' from the ten scenarios*. Generally, it found that the node 'Unregulated reef-based tourism' (scenario 2) would negatively affect 'Coral health' and 'Fish biomass and diversity' and a situation causing reef-based

* Through the Results and Discussion, we discuss 'coral reef health' in regards to 'Coral health' and 'Fish biomass and diversity'. The term 'coral reef health' refers to the overall health of the ecosystem, with the two key components being its associated coral and fish communities.

tourism sites to temporarily close (scenario 10) would positively affect them. Our model showed that the node 'Fish biomass and diversity' was most positively affected by 'Constructing artificial reefs funded through volunteer-tourism' (scenario 1), as well as 'Government established local environmental regulations' (scenario 5) and 'Establishing marine protected areas' (scenario 9). It also found that node 'Coral health' was most positively affected by 'Reef-based tourism organisations (non-government) establishing environmental standards' (such as no anchoring rules, cutting boat engine rules over dive sites etc; scenario 3), as well as the scenario 'Establishing marine protected areas' (scenario 9).

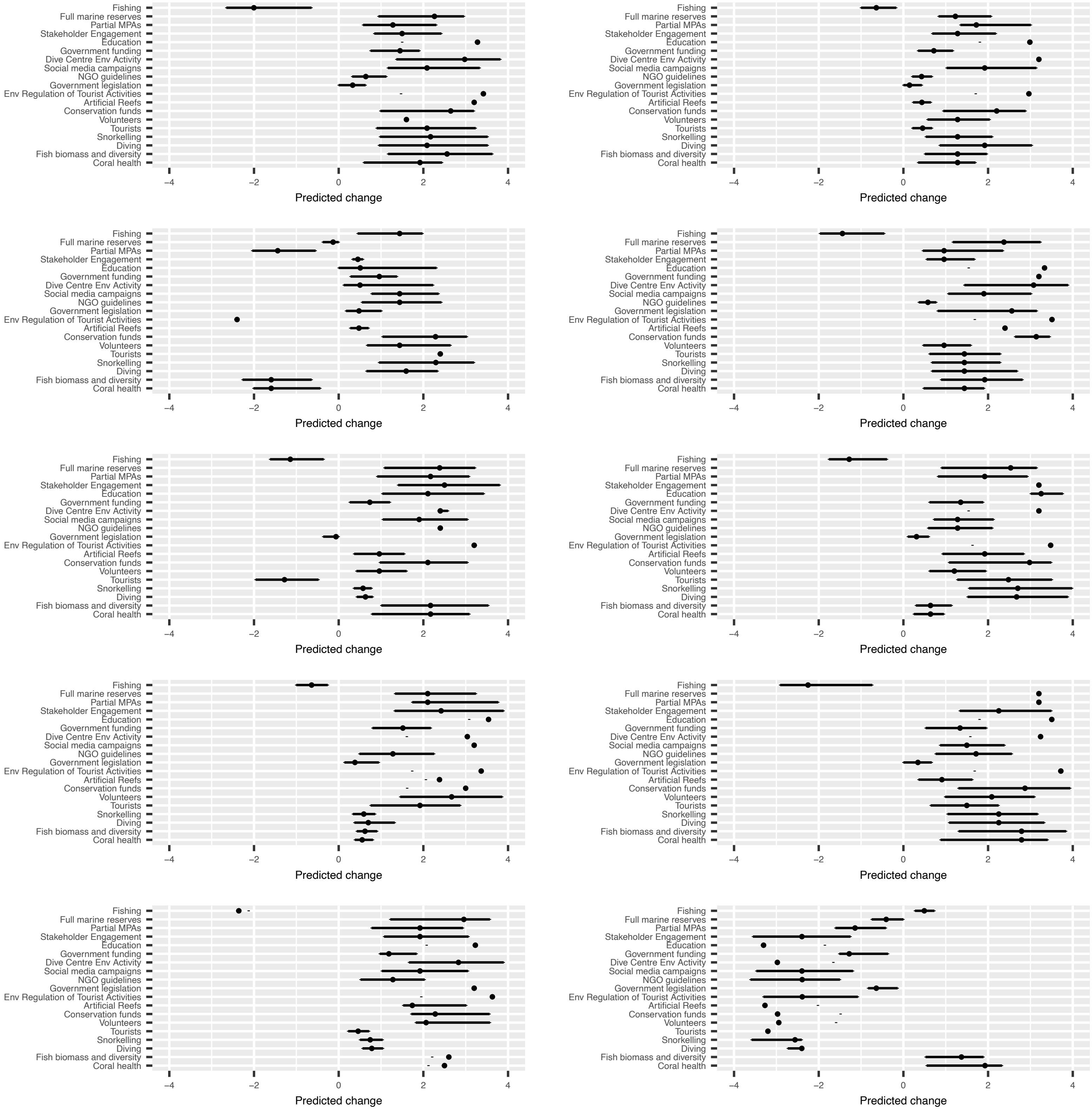

Figure 5. 5: Calculated mean (+/- 95 % Confidence intervals from n = 10,000 bootstrap replicates) probability of increase in each category. Values > 0 mean likely increases, those < 0 mean likely decreases. Details of model inputs are given in table S.5.1 (supplementary material) and presented here by scenario number.

Discussion

The insufficient integration of ecological and social science data, as noted by Stafford (2018), has led to a limited understanding of whether stakeholders' opinions on system health align with accurate biological data. By comparing RHI scores to stakeholders perceptions of the health of their reefs, our results showed that interviewees opinions of coral reef health were generally supported by RHI scores, suggesting that stakeholders were able to identify the health of their coral reefs well. Subsequently, our model predicted the outcomes of 10 key actions/ management measures in terms of outcomes for coral reef health. It predicted that the scenario 'Unregulated reef-based tourism' would negatively affect coral reef health, whereas 'Constructing artificial reefs funded through volunteer-tourism', 'Government established local environmental regulations', 'Establishing marine protected areas' and 'Reef-based tourism organisations (non-government) establishing environmental standards' would positively affect it. This section will now assess localised coral reef conservation, in terms of the key helpful and harmful stakeholder actions, as identified by our qualitative findings and quantified by our Bayesian belief network model.

Helpful management measures for coral reef conservation

The model predicted that there were several stakeholder actions / management measures which can be utilised in marine tourism areas to help conserve coral reefs. The node 'Fish biomass and diversity' was most positively affected by the scenario 'Constructing artificial reefs funded through volunteer-tourism', as well as 'Government established local environmental regulations' and 'Establishing marine protected areas'. It also predicted that node 'Coral health' was most positively affected by the scenario 'Reef-based tourism organisations (non-government) establishing environmental standards' (such as no anchoring rules, cutting boat engine rules over dive sites etc), as well as 'Establishing marine protected areas'. This section will discuss some of the key management measures which our model predicted would lead to positive outcomes for coral reef health.

Constructing artificial reefs funded through volunteer-tourism

It was predicted that 'Fish biomass and diversity' would substantially increase as a result of the scenario 'Constructing artificial reefs funded through volunteer-tourism'. This was supported by literature highlighting that the deployment of artificial reefs onto degraded reefs can substantially enhance fish biomass and diversity, because they provide previously unavailable substrata used for shelter, egg-laying and hunting (Folpp et al. 2020; Paxton et al. 2020; Boakes et al.2022a). The model also predicted that this scenario would lead to increases in coral health, but to a slightly lesser extent. The corals can successfully recruit and grow on artificial reefs, although this process is slower than fish colonisation (Burt et al. 2009; Boakes et al. 2023b). The restoration of coral reefs through artificial reef deployment may be partially or fully funded by the 'Volunteer-Tourism Model', whereby the payment of a volunteer (to be involved in a given project) is used to fund a programmes activities (Kitney et al. 2018). Participant T1 commented "*Now our reef restoration programme is sustainably funded through volunteer payments – 100% of it*". Literature has also highlighted the benefits of the volunteer-tourism model, in providing essential funding for social or environmental projects (Campbell and Smith 2006; Grimm and Needham 2012), and has "*allowed our programme [in Tianyar] to build one of the largest artificial reefs in Indonesia, over 15,000 reef units*" (T10). Tianyar was shown to have the highest RHI score (4.5) out of all of the locations in our study. Given that two of the four indices used to calculate this score were related to fish biomass, it is likely that the deployment of artificial reefs (and therefore increased fish biomass) contributed to its high RHI score, providing an example of how volunteer-tourism can lead to positive outcomes for coral reef health in reef-based tourism areas (as per the findings of Diedrich (2007); Boley and Green (2016); Eider et al. (2023)).

Based on the above paragraph, it must be noted that to calculate RHI values, the methods for this paper followed those of Díaz-Pérez et al. (2016), which estimated average biomass per RUV recording (of both herbivorous and commercial fish) by adding up each biomass value of the listed fish families. The limitation of this method is that fish species within families can vary greatly in length. For example, an adult Bump head wrasse can be over 1m long, whereas an adult Canary wrasse can be less than 10cm long, and this method would have taken an average of the two (amongst others) to calculate average Wrasse family fish length. As a consequence, the values used to calculate fish biomass are likely to have some inaccuracies, and therefore the RHI values should be considered an estimate quantification of reef health, rather than precise reef health values.

Establishing public and private environmental regulations

The model predicted substantial improvements to coral reef health (fish and coral) as a result of establishing environmental regulations. Participants in all locations discussed coastal rules and regulations which could be or have been established in their village to protect the marine environment, with examples including fines for those who dispose of their household waste illegally, banning anchoring on coral reefs, establishing marine protected areas and setting up tourism quotas which limit the number of dive/ snorkel boats allowed to visit popular marine tourism sites. Merging our ecological and qualitative results revealed that the research locations with the strictest coastal regulations had the highest RHI scores (Tianyar and Nusa Penida MPA), whilst Kalibukbuk, with the least strict coastal regulations, had the lowest RHI score. These findings suggest that the establishment of these regulations directly support coral reef conservation, and in their absence, coral reef health will likely suffer. This was also predicted by our model, which found that coral and fish biomass and diversity would be strongly increased by Government established local environmental regulations and 'Reef-based tourism organisations (non-government) establishing environmental standards'.

Location 1 (Tianyar) reportedly had several locally enforced coastal regulations, including a "*locally established and enforced no-take (no fishing) zone*" (T12), "*strict rules and punishments for those who wrongly dispose of their plastic waste*" (T11) and "*banning unsustainable fishing practices*" (T2). However participant T1 mentioned that "*all environmental legislations here were set-up by the local community and do not have official government support or backing*". Research suggests that MPAs that are developed and enforced by local people will likely lead to greater ecological success (Lundquist and Granek 2005; Suadi 2009; Glaser et al. 2010). In contrast, location 3 (Nusa Penida MPA) hosted Bali's largest officially recognised MPA, which was generally agreed by participants to be successful, for example: "*The government established the MPA here several years ago and it has been very effective – we can't do certain activities there to protect fish and corals*" (N6). The MPA has also received a substantial amount of research attention, with several publications highlighting its success in terms of improving ecosystem health (Pedju 2018; Yunitawati and Clifton 2019) and overall management (Weeks et al. 2014). It is again likely that the establishment of the coastal regulations in location 1 and 3 contributed to their observed high RHI scores.

In contrast, location 2, Kalibukbuk had a notably lower RHI score. This may, in part, be explained by the lack of coastal regulations, combined with mass reef-based tourism. K1 reported that "*60-70% of corals here have been destroyed*", which may be explained, in part, by K2's explanation that "*Many regulations here are still missing - e.g. unsustainable fishing is still allowed on the reef, and there is no limit on the amount of divers or snorkellers".* Furthermore, K6 explained that "*Most people working in tourism don't understand the marine environment - they just think about "today", not about the future. Their practice is unsustainable, and there are no regulations to stop them*".

Furthermore, when interviewed, tourists generally agreed that the tourism they had experienced was harmful for coral reef health, for example K11 said "*It was clear that the guides don't really care about protecting the marine environment. The tour was unsustainable and there were so many tourists. That's probably why the marine life here is unhealthy"*.

Engaging stakeholders with coral reef conservation objectives

It was predicted that engaging stakeholders with coral reef conservation would lead to positive outcomes for coral reef health. In Nusa Penida, it was highlighted (through participant responses and relevant literature) that additional work was needed to engage stakeholders to find management solutions to the management challenges experienced by marine park faces. Participant N3 commented "*We should make a limit on the number of boats allowed, like in the tourism area in Komodo [marine park in east Indonesia]. There are way too many [tourism] boats here right* now*, it's not sustainable".* Participant N9 explained that "*The MPA zonation is confusing and unclear. As the head of the village, I don't even know the MPA zonation because there is a lack of communication between the government [who establish the MPA] and the community",* which was also discussed in Ruchimat et al. (2013). Many participants stressed that "*Education and socialization is very important for sustainability*" (N2) and that "*We need the government to bring stakeholders together at a socialisation event, to agree on issues related to the MPA*" (N6). Similar suggestions were made by participants in Kalibukbuk, for example K7 commented that "*We need to hold socialisation to agree on protective legislations amongst all stakeholders".* Existing literature has also highlighted the importance of communication between stakeholders within reef based tourism areas and MPAs, in terms of planning and implementing sustainable marine tourism (Wongthong and Harvey 2014; Tranter et al. 2022). An important tool to engage stakeholders in coral restoration projects is by making them aware of the ecosystem services that coral reefs provide, as well as those that can be captured by a coral restoration project (Boakes et al. 2023a). It is generally agreed that communities are more likely to engage in and/or support a project if they recognise its value (Bennett and Dearden 2014; Kusumawati and Huang 2015; Grúňová et al. 2017; Abdurrahim et al. 2022), especially if they obtain direct socio-economic benefits it, such as new jobs in the tourism sector or higher fishing yields (Boakes et al. 2023a). This was also confirmed by T10, who commented that "*The community are now aware of the link between engaging in coral conservation and improving future prospects, which has been obtained through regular meetings and socialisation*". Communities are diverse and perceptions vary between groups, suggesting that implementing uniform management measures may be incomplete or ineffective. Awareness campaigns and capacity-building efforts must be tailored to reduce misperceptions about the state of local resources and to address the specific needs and challenges faced by different groups(Hamza et al. 2023).

Unregulated tourism is harmful for coral reefs

Mass tourism refers to the movement of a large number of organised tourists to a given area (Naumov and Green 2016). Unregulated mass coastal tourism over much of southeast Asia has rapidly expanded over recent decades (Fabinyi 2010; King 2018). Within Bali itself, mass reef-based tourism has been associated with physical damage to corals, through dive tourists stepping on with corals (Suparno et al. 2019) or dive boats directly anchoring on top of reefs (Doherty et al. 2013). Our model further substantiated this finding, indicating that unregulated reef-based tourism, defined by unrestricted tourist numbers and activities, would detrimentally impact coral health. It also predicted a temporary pause in tourism through the scenario 'Pandemics causing reef-based tourism sites to temporarily close' would positively affect coral health, likely because the threats to reefs associated with tourism would not be present in this scenario. This prediction was also supported by N2, a tourism worker in Nusa Penida, who mentioned that "*After 2 years of covid the corals were looking so healthy because there was no tourism. But since covid stopped and tourists have started coming back, the corals are getting broken again*". It should also be noted that the volume of tourists that each location receives likely influences its outcomes for coral reef health. Chong (2020) highlighted that mass tourism in Bali had led to environmental and societal consequences because local infrastructure cannot cope with such a large number of tourists. However, low-volume, well-managed tourism is generally more sustainable and leads to fewer environmental consequences (Beyer et al. 2005; Nyaupane 2007).

A win/win situation for ecology and economics

Budowski et al. (1976) highlighted that tourism and conservation aims may conflict "without due planning", however a 'win-win' situation for economics and ecology may be achieved by low-volume, well-managed tourism (Burgin and Hardiman 2010; Duffy 2015; Tong 2022). The establishment of effective management measures within a reef-tourism area can facilitate a symbiotic relationship between tourism and conservation in the wide sense, which offers substantial environmental and economic benefits to a marine tourism area (Budowski 1976; Pascal et al. 2021; Brathwaite et al. 2022). This 'mutualistic' relationship in the context of reef tourism, would foster greater financial investment in coral reef conservation, and as coral reef health improves, more tourists are attracted to visit these areas (Diedrich 2007).

Whilst this concept initially appears to be a faultless solution, in reality, the lack of effective management measures in most reef-tourism areas in Bali (and perhaps globally) has meant that that a well-balanced, mutually beneficial situation is seldom achieved. However, some of the coral reef areas in this study (especially those with the highest RHI scores) did appear to be obtaining a substantial level of benefit from reef-based tourism. In these area(s), tourism had undoubtably provided the awareness and capital needed to conduct reef conservation work – findings also supported by Diedrich (2007); Brathwaite et al. (2022); Eider et al. (2023). In Tianyar, participant T1 explained that "*Our restoration programme is entirely funded by volunteer-tourism, and this funding pays for the salaries for many people within the village*". This funding not only provided direct capital for restoration work (and employment for local people associated with it), but was also to a large extent, responsible for generating the subsequent community interest and engagement, which was found to be key to the restoration programme's success (Boakes et al. 2023a)).

Conclusion

Our research firstly aimed to investigate the extent to which stakeholders in reef-based tourism areas in Bali understood the health of their local coral reefs, and by comparing RHI scores our qualitative data, we found that stakeholders were able to identify the health of their coral reefs well. We then aimed to assess if relevant stakeholders could create management measures which fostered a mutually beneficial relationship between tourism economy and coral reef health. Overall, our research found that the extent to which tourism is helpful or harmful to coral reef health depends upon the level management measures in place, and whether stakeholder actions balance tourism economy with coral reef conservation. We found that the locations with the highest RHI scores (notably Tianyar) had established the most/ strictest reef management measures, and those with the lowest RHI scores (notably Kalibukbuk) had limited measures in place. Tianyar served as a positive example of how low-volume (maximum 50 tourists), reef-based tourism may be able to provide mutual benefits for ecology and economy, in large part due to the establishment of management measures and stakeholder actions which allow economic activities to continue, whilst also directly supporting reef conservation. With projected future declines in global coral reef health, stakeholders in marine tourism areas should aim to utilise the opportunity provided by tourism to conserve reefs, whilst establishing locally-supported management measures that safeguard the marine environment.

References

Abdurrahim, A. Y., Adhuri, D. S., Ross, H. and Phelan, A., 2022. Community champions of ecosystem services: The role of local agency in protecting Indonesian coral reefs. *Frontiers in Ecology and Evolution*, 10, 868218.

Allen, G. R., 2008. Conservation hotspots of biodiversity and endemism for Indo-Pacific coral reef fishes. *Aquatic conservation marine and freshwater ecosystems*.

Allen, G. R. and Erdman, M. V, 2008. Reef fish of Nusa Penida, Indonesia. *Final Report to Conservation International*, 22.

Allen, G. R. and Erdmann, M. V, 2013. Reef Fishes of Bali, Indonesia. *In*: *Bali Marine Rapid Assessment Program 2011*. BioOne, 15.

Andrello, M., Darling, E. S., Wenger, A., Suárez-Castro, A. F., Gelfand, S. and Ahmadia, G. N., 2021. A global map of human pressures on tropical coral reefs. *Conservation Letters*, e12858.

Antara, M. and Sumarniasih, M. S., 2017. Role of tourism in economy of Bali and Indonesia. *Journal of Tourism and Hospitality Management*, 5 (2), 34–44.

Bayraktarov, E., Stewart-Sinclair, P. J., Brisbane, S., Boström-Einarsson, L., Saunders, M. I., Lovelock, C. E., Possingham, H. P., Mumby, P. J. and Wilson, K. A., 2019. Motivations, success, and cost of coral reef restoration. *Restoration Ecology*, 27 (5), 981–991.

Bellwood, D. R., Pratchett, M. S., Morrison, T. H., Gurney, G. G., Hughes, T. P., Álvarez-Romero, J. G., Day, J. C., Grantham, R., Grech, A. and Hoey, A. S., 2019. Coral reef conservation in the Anthropocene: confronting spatial mismatches and prioritizing functions. *Biological conservation*, 236, 604–615.

Beyer, D., Anda, M., Elber, B., Revell, G. and Spring, F., 2005. Best practice model for low-impact nature-based sustainable tourism facilities in remote areas.

Boakes, Z., Hall, A. E., Ampou, E. E., Jones, G. C. A., Suryaputra, I. G. N. A., Mahyuni, L. P., Prasetyo, R. and Stafford, R., 2022. Coral reef conservation in Bali in light of international best practice, a literature review. *Journal for Nature Conservation*, 126190.

Boakes, Z., Hall, A., Jones, G., Prasetyo, R., Stafford, R. and Yahya, Y., 2022. Artificial coral reefs as a localised approach to increase fish biodiversity and abundance along the North Bali coastline. *AIMS Geosciences*, 8 (2), 303–325.

Boakes, Z., Mahyuni, L. P., Hall, A. E., Cvitanovic, M. and Stafford, R., 2023. Can Coral Reef Restoration Programmes Facilitate Changes in Environmental Attitudes? A Case Study on a Rural Fisher Community in North Bali, Indonesia. *Human Ecology*, 1–15.

Boakes, Z., Suryaputra, I. G. N. A., Hall, A. E., Franklin, D. J. and Stafford, R., 2023. Nutrient dynamics, carbon storage and community composition on artificial and natural reefs in Bali, Indonesia. *Marine Biology*, 170 (10), 130.

Boley, B. B. and Green, G. T., 2016. Ecotourism and natural resource conservation: the 'potential'for a sustainable symbiotic relationship. *Journal of Ecotourism*, 15 (1), 36–50.

Bos, M., Pressey, R. L. and Stoeckl, N., 2015. Marine conservation finance: The need for and scope of an emerging field. *Ocean & Coastal Management*, 114, 116–128.

Brathwaite, A., Pascal, N. and Clua, E., 2022. Private Capital to Improve Nature-Based Solutions for Coastal Protection: Time for a Boost. *In*: *Oceans*. MDPI, 60–71.

Brown, K., Adger, W. N., Tompkins, E., Bacon, P., Shim, D. and Young, K., 2001. Trade-off analysis for marine protected area management. *Ecological economics*, 37 (3), 417–434.

Budowski, G., 1976. Tourism and environmental conservation: conflict, coexistence, or symbiosis? *Environmental conservation*, 3 (1), 27–31.

Burgin, S. and Hardiman, N., 2010. Eco-accreditation: win-win for the environment and small business? *International Journal of Business Studies: A Publication of the Faculty of Business Administration, Edith Cowan University*, 18 (1), 23–38.

Burke, L., Reytar, K., Spalding, M. and Perry, A., 2012. *Reefs at risk revisited in the Coral Triangle*.

Burt, J., Bartholomew, A., Bauman, A., Saif, A. and Sale, P. F., 2009. Coral recruitment and early benthic community development on several materials used in the construction of artificial reefs and breakwaters. *Journal of Experimental Marine Biology and Ecology*, 373 (1), 72–78.

Byczek, C., 2011. Blessings for All? Community-Based Ecotourism in Bali Between Global, National, and Local Interests–A Case Study. *Austrian Journal of South-East Asian Studies*, 4 (1), 81–106.

Campbell, L. M. and Smith, C., 2006. What makes them pay? Values of volunteer tourists working for sea turtle conservation. *Environmental management*, 38, 84–98.

Chaves, L. T. C., Pereira, P. H. C. and Feitosa, J. L. L., 2013. Coral reef fish association with macroalgal beds on a tropical reef system in North-eastern Brazil. *Marine and Freshwater Research*, 64 (12), 1101–1111.

Chok, S., Macbeth, J. and Warren, C., 2007. Tourism as a tool for poverty alleviation: A critical analysis of 'pro-poor tourism'and implications for sustainability. *Current issues in Tourism*, 10 (2–3), 144–165.

Choy, D. J. L., 1995. The quality of tourism employment. *Tourism management*, 16 (2), 129–137.

Christie, P., 2004. Marine protected areas as biological successes and social failures in Southeast Asia. *In*: *American fisheries society symposium*. Bethesda, MD.

Clua, E., Legendre, P., Vigliola, L., Magron, F., Kulbicki, M., Sarramegna, S., Labrosse, P. and Galzin, R., 2006. Medium scale approach (MSA) for improved assessment of coral reef fish habitat. *Journal of Experimental Marine Biology and Ecology*, 333 (2), 219–230.

Costanza, R., De Groot, R., Sutton, P., Van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S. and Turner, R. K., 2014. Changes in the global value of ecosystem services. *Global environmental change*, 26, 152–158.

Davenport, J. and Davenport, J. L., 2006. The impact of tourism and personal leisure transport on coastal environments: a review. *Estuarine, coastal and shelf science*, 67 (1–2), 280–292.

Díaz-Pérez, L., Rodríguez-Zaragoza, F. A., Ortiz, M., Cupul-Magaña, A. L., Carriquiry, J. D., Ríos-Jara, E., Rodríguez-Troncoso, A. P. and García-Rivas, M. del C., 2016. Coral reef health indices versus the biological, ecological and functional diversity of fish and coral assemblages in the Caribbean Sea. *PloS one*, 11 (8), e0161812.

Diedrich, A., 2007. The impacts of tourism on coral reef conservation awareness and support in coastal communities in Belize. *Coral Reefs*, 26 (4), 985–996.

Doherty, O., Milner, C., Dustan, P., Campbell, S., Pardede, S., Kartawijaya, T. and Alling, A., 2013. Report on Menjangan Island's coral reef: A Bali Barat National Park marine protected area. *Atoll Research Bulletin*, 19 (599), 1–18.

Duffy, R., 2015. Nature-based tourism and neoliberalism: Concealing contradictions. *Tourism Geographies*, 17 (4), 529–543.

Eider, D., Partelow, S., Albrecht, S., Adrianto, L. and Kluger, L. C., 2023. SCUBA tourism and coral reefs: a social-ecological network analysis of governance challenges in Indonesia. *Current Issues in Tourism*, 26 (7), 1031–1050.

Fabinyi, M., 2010. The intensification of fishing and the rise of tourism: competing coastal livelihoods in the Calamianes Islands, Philippines. *Human Ecology*, 38 (3), 415–427.

Folpp, H., Lowry, M., Gregson, M. and Suthers, I. M., 2013. Fish assemblages on estuarine artificial reefs: natural rocky-reef mimics or discrete assemblages? *PLoS One*, 8 (6), e63505.

Folpp, H. R., Schilling, H. T., Clark, G. F., Lowry, M. B., Maslen, B., Gregson, M. and Suthers, I. M., 2020. Artificial reefs increase fish abundance in habitat-limited estuaries. *Journal of Applied Ecology*, 57 (9), 1752–1761.

Frey, J. and Berkes, F., 2014. Can partnerships and community-based conservation reverse the decline of coral reef social-ecological systems? *International Journal of the Commons*, 8 (1).

Froese, R. and Pauly, D., 2000. FishBase 2000: concepts, design and data sources. ICLARM, Los Ban os. *Laguna*.

Garrod, B., Wilson, J. C. and Bruce, D. M., 2003. Defining marine ecotourism: a Delphi study. *Marine Ecotourism: Issues and Experiences. Channel View, Clevedon, Australia*, 17–36.

Germanov, E. S., Marshall, A. D., Hendrawan, I. G., Admiraal, R., Rohner, C. A., Argeswara, J., Wulandari, R., Himawan, M. R. and Loneragan, N. R., 2019. Microplastics on the menu: plastics pollute indonesian manta ray and whale shark feeding grounds. *Frontiers in Marine Science*, 6, 679.

Gidman, W. K., Hassell, K., Day, J. and Payne, K., 2007. The impact of increasing workloads and role expansion on female community pharmacists in the United Kingdom. *Research in Social and Administrative Pharmacy*, 3 (3), 285–302.

Giglio, V. J., Ternes, M. L. F., Kassuga, A. D. and Ferreira, C. E. L., 2019. Scuba diving and sedentary fish watching: effects of photographer approach on seahorse behavior. *Journal of Ecotourism*, 18 (2), 142–151.

Gladstone, W., Curley, B. and Shokri, M. R., 2013. Environmental impacts of tourism in the Gulf and the Red Sea. *Marine pollution bulletin*, 72 (2), 375–388.

Glaser, M., Baitoningsih, W., Ferse, S. C. A., Neil, M. and Deswandi, R., 2010. Whose sustainability? Top–down participation and emergent rules in marine protected area management in Indonesia. *Marine Policy*, 34 (6), 1215–1225.

Greenbaum, T. L., 1998. *The handbook for focus group research*. Sage.

Grimm, K. E. and Needham, M. D., 2012. Moving beyond the "I" in motivation: Attributes and perceptions of conservation volunteer tourists. *Journal of Travel Research*, 51 (4), 488–501.

Haddock-Fraser, J. and Hampton, M. P., 2012. Multistakeholder values on the sustainability of dive tourism: case studies of Sipadan and Perhentian Islands, Malaysia. *Tourism Analysis*, 17 (1), 27–41.

Hall, A. E., Herbert, R. J. H. and Stafford, R., 2021. Temporal and spatial variation in adult and juvenile mobile fauna associated with natural and artificial coastal habitats. *Marine Biology*, 168 (2), 1–13.

Hamza, A. J., Esteves, L. S., Cvitanović, M. and Kairo, J. G., 2023. Sustainable natural resource management must recognise community diversity. *International Journal of Sustainable Development & World Ecology*, 1–18.

Harriott, V. J., 2004. Marine tourism impacts on the Great Barrier Reef. *Tourism in Marine Environments*, 1 (1), 29–40.

Hayes, C. T., Baumbach, D. S., Juma, D. and Dunbar, S. G., 2017. Impacts of recreational diving on hawksbill sea turtle (Eretmochelys imbricata) behaviour in a marine protected area. *Journal of Sustainable Tourism*, 25 (1), 79–95.

Hein, M. Y., Vardi, T., Shaver, E. C., Pioch, S., Boström-Einarsson, L., Ahmed, M., Grimsditch, G. and McLeod, I. M., 2021. Perspectives on the use of coral reef restoration as a strategy to support and improve reef ecosystem services. *Frontiers in Marine Science*, 299.

Howlett, L., Camp, E. F., Edmondson, J., Edmondson, J., Agius, T., Hosp, R., Coulthard, P., Edmondson, S. and Suggett, D. J., 2022. Adoption of coral propagation and out-planting via the tourism industry to advance site stewardship on the northern Great Barrier Reef. *Ocean & Coastal Management*, 225, 106199.

Hughes, T. P., Baird, A. H., Morrison, T. H. and Torda, G., 2023. Principles for coral reef restoration in the anthropocene. *One Earth*.

Hughes, T. P., Kerry, J. T., Álvarez-Noriega, M., Álvarez-Romero, J. G., Anderson, K. D., Baird, A. H., Babcock, R. C., Beger, M., Bellwood, D. R. and Berkelmans, R., 2017. Global warming and recurrent mass bleaching of corals. *Nature*, 543 (7645), 373–377.

IPCC, 2021. *The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*.

Kasman, E., Purwanto, E. N. and Wirya, S., 2021. *Baseline Data Identification of the Coral Reef Ecosystems (Reef Health Monitoring) in Nusa Penida*.

Kellmereit, B., 2015. Focus groups. *International Journal of Sales, Retailing & Marketing*, 4 (9), 42–52.

King, A. J., George, A., Buckle, D. J., Novak, P. A. and Fulton, C. J., 2018. Efficacy of remote underwater video cameras for monitoring tropical wetland fishes. *Hydrobiologia*, 807 (1), 145–164.

King, V. T., 2018. Tourism and culture in Malaysia 1. *In*: *Tourism in South-East Asia*. Routledge, 99–116.

Kitney, S., Stanway, A. R. and Ryan, M. M., 2018. Volunteer tourism motivations of the Marine Conservation Cambodia project. *Current Issues in Tourism*, 21 (10), 1091–1096.

Kruglov, L. P. and Davidson, H. H., 1953. The willingness to be interviewed: a selective factor in sampling. *The Journal of Social Psychology*, 38 (1), 39–47.

Legare, C. H., Akhauri, S., Chaudhuri, I., Hashmi, F. A., Johnson, T., Little, E. E., Lunkenheimer, H. G., Mandelbaum, A., Mandlik, H. and Mondal, S., 2020. Perinatal risk and the cultural ecology of health in Bihar, India. *Philosophical Transactions of the Royal Society B*, 375 (1805), 20190433.

Leisher, C., Nature Conservancy, C., Van Beukering, P. and Scherl, L. M., 2007. Nature's investment bank: how marine protected areas contributed to poverty reduction.

Leujak, W. and Ormond, R. F. G., 2007. Comparative accuracy and efficiency of six coral community survey methods. *Journal of experimental marine biology and ecology*, 351 (1–2), 168–187.

Li, D., Hao, Y. and Duan, Y., 2020. Nonintrusive methods for biomass estimation in aquaculture with emphasis on fish: a review. *Reviews in Aquaculture*, 12 (3), 1390–1411.

Lundquist, C. J. and Granek, E. F., 2005. Strategies for successful marine conservation: integrating socioeconomic, political, and scientific factors. *Conservation Biology*, 19 (6), 1771–1778.

Marshall, B., Cardon, P., Poddar, A. and Fontenot, R., 2013. Does sample size matter in qualitative research?: A review of qualitative interviews in IS research. *Journal of computer information systems*, 54 (1), 11–22.

Mason, M., 2010. Sample size and saturation in PhD studies using qualitative interviews. *In*: *Forum qualitative Sozialforschung/Forum: qualitative social research*.

Moura, C. O. de, Silva, Í. R., Silva, T. P. da, Santos, K. A., Crespo, M. da C. A. and Silva, M. M. da, 2021. Methodological path to reach the degree of saturation in qualitative research: grounded theory. *Revista Brasileira de Enfermagem*, 75.

Mustika, P. L. K., Williams, R., Kadarisman, H. P., Purba, A. O., Maharta, I. P. R. F., Rahmadani, D., Faiqoh, E. and Dewantama, I. M. I., 2021. A rapid assessment of the marine megafauna biodiversity around South Bali, Indonesia. *Frontiers in Marine Science*, 8, 606998.

Mustika, P. L. and Ratha, I., 2013. Towards the Bali MPA Network. *In*: *Bali Marine Rapid Assessment Program 2011*. BioOne, 131.

Naumov, N. and Green, D., 2016. Mass tourism. *Encyclopedia of Tourism. Springer, Cham.*

Nyaupane, G. P., 2007. Ecotourism versus nature-based tourism: Do tourists really know the difference? *Anatolia*, 18 (1), 161–165.

OECD, A., 2020. Comprehensive Overview of Global Biodiversity Finance. *Organization for Economic Cooperation and Development*.

Orams, M., 2002. *Marine tourism: development, impacts and management*. Routledge.

Orams, M. B., 1995. Towards a more desirable form of ecotourism. *Tourism management*, 16 (1), 3–8.

Papageorgiou, M., 2016. Coastal and marine tourism: A challenging factor in Marine Spatial Planning. *Ocean & coastal management*, 129, 44–48.

Parsons, E. C. M., MacPherson, R. and Villagomez, A., 2017. Marine "conservation": you keep using that word but I don't think it means what you think it means. *Frontiers in Marine Science*, 4, 299.

Pascal, N., Brathwaite, A., Bladon, A., Claudet, J. and Clua, E., 2021. Impact investment in marine conservation. *Ecosystem Services*, 48, 101248.

Patton, M. Q., 2014. *Qualitative research & evaluation methods: Integrating theory and practice*. Sage publications.

Paxton, A. B., Shertzer, K. W., Bacheler, N. M., Kellison, G. T., Riley, K. L. and Taylor, J. C., 2020. Meta-analysis reveals artificial reefs can be effective tools for fish community enhancement but are not one-size-fits-all. *Frontiers in Marine Science*, 7, 282.

Pedju, F. M. K., 2018a. Stakeholders' perceptions of the impacts of tourism on the social and ecological resilience of marine protected areas in Bali, Indonesia.

Pedju, F. M. K., 2018b. Stakeholders' perceptions of the impacts of tourism on the social and ecological resilience of marine protected areas in Bali, Indonesia.

Piskurek, N., 2001. Dive tourism in coral reefs-impacts and conditions for sustainability: a case study from Desa Pemuteran(Bali/ Indonesia). *ACP-EU Fisheries Research Report*, 10, 57–65.

Prasetia, D., Supriharyono, M., Anggoro, S. and Sya'Rani, L., 2017. Coral Bleaching on Lembongan Island, Nusa Penida, Bali. *In*: . Atlantis Press.

Prasetyo, A. P., Simpfendorfer, C., Sherman, S. and Moore, S., 2019. The study of shark and ray abundance in Nusa Penida Aquatic Conservation Area.

Principe, P., Bradley, P., Yee, S., Fisher, W., Johnson, E., Allen, P. and Campbell, D., 2012. Quantifying coral reef ecosystem services. US Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC.

Ruchimat, T., Basuki, R. and Welly, M., 2013. Nusa Penida Marine Protected Area (Mpa) Bali-Indonesia: Why Need to be Protected? *Transylvanian Review of Systematical and Ecological Research*, 15 (1), 193–202.

Rudd, M. A. and Tupper, M. H., 2002. The impact of Nassau grouper size and abundance on scuba diver site selection and MPA economics. *Coastal Management*, 30 (2), 133–151.

Schmidt-Roach, S., Kunzmann, A. and Arbizu, P. M., 2008. In situ observation of coral recruitment using fluorescence census techniques. *Journal of Experimental Marine Biology and Ecology*, 367 (1), 37–40.

SER, 2004. Society for ecological restoration international science & policy working group.

Shani, A., Polak, O. and Shashar, N., 2012. Artificial reefs and mass marine ecotourism. *Tourism Geographies*, 14 (3), 361–382.

Spalding, M., Burke, L., Wood, S. A., Ashpole, J., Hutchison, J. and Zu Ermgassen, P., 2017. Mapping the global value and distribution of coral reef tourism. *Marine Policy*, 82, 104–113.

Stafford, R., 2018. Lack of evidence that governance structures provide real ecological benefits in marine protected areas. *Ocean & coastal management*, 152, 57–61.

Stafford, R., Croker, A. R., Rivers, E. M., Cantarello, E., Costelloe, B., Ginige, T., Sokolnicki, J., Kang, K., Jones, P. J. S. and McKinley, E., 2020. Evaluating optimal solutions to environmental breakdown. *Environmental Science & Policy*, 112, 340–347.

Statistik, B. P., 2019. Profil Perempuan Indonesia 2019. *Kementeri. Pemberdaya. Peremp. dan Perlindungan Anak*.

Stewart, B. D., Howarth, L. M., Wood, H., Whiteside, K., Carney, W., Crimmins, É., O'Leary, B. C., Hawkins, J. P. and Roberts, C. M., 2020. Marine conservation begins at home: how a local community and protection of a small bay sent waves of change around the UK and beyond. *Frontiers in Marine Science*, 7, 76.

Stewart-Sinclair, P. J., Klein, C. J., Bateman, I. J. and Lovelock, C. E., 2021. Spatial cost–benefit analysis of blue restoration and factors driving net benefits globally. *Conservation Biology*, 35 (6), 1850–1860.

Suadi, N. M., 2009. Sharing the commons: resolving the tragedy of the commons through collaborative Management of Coastal Commons at Kedonganan Village. *Bali. J Kyosei Stud*, 3 (1), 263–285.

Sudipa, N., Mahendra, M. S., Adnyana, W. S. and Pujaastawa, I. B., 2020. Tourism impact on the environment in Nusa Penida tourism area. *Journal of Environmental Management & Tourism*, 11 (1 (41)), 113–124.

Suggett, D. J., Edwards, M., Cotton, D., Hein, M. and Camp, E. F., 2023. An integrative framework for sustainable coral reef restoration. *One Earth*.

Sujarwoto, S., Tampubolon, G. and Pierewan, A. C., 2018. Individual and contextual factors of happiness and life satisfaction in a low middle income country. *Applied Research in Quality of Life*, 13 (4), 927–945.

Suparno, N., Currier, K., Milner, C., Alling, A. and Dustan, P., 2019. Ecological changes in the coral reef communities of Indonesia's Bali Barat National Park, 2011–2016. *Atoll Res Bull*, 620, 1–35.

Susiloningtyas, D., Handayani, T. and Amalia, A. N., 2018. The impact of coral reefs destruction and climate change in Nusa Dua and Nusa Penida, Bali, Indonesia. *In*: . IOP Publishing, 12054.

Suteja, Y., Atmadipoera, A. S., Riani, E., Nurjaya, I. W., Nugroho, D. and Cordova, M. R., 2021. Spatial and temporal distribution of microplastic in surface water of tropical estuary: Case study in Benoa Bay, Bali, Indonesia. *Marine pollution bulletin*, 163, 111979.

The Nature Conservancy, 2017. *How Tourism Can Be Good for Coral Reefs* [online]. Available from: https://www.nature.org/en-us/what-we-do/our-insights/perspectives/how-tourism-can-be-good-for-coral-reefs/ [Accessed 2 Nov 2023].

Tito, C. K., Ampou, E. E. and Wibawa, T. A., 2019. Stressor-Response of Reef-Building Corals to Climate Change in the Menjangan Island, West Bali National Park, Indonesia. *In*: . IOP Publishing, 12011.

Titus, B. M., Daly, M. and Exton, D. A., 2015. Do reef fish habituate to diver presence? Evidence from two reef sites with contrasting historical levels of SCUBA intensity in the Bay Islands, Honduras. *PloS one*, 10 (3), e0119645.

Tong, H., 2022. Sustainable development and win-win cooperation in Asian marine tourism: Conference report. *Marine Policy*, 146, 105304.

Tranter, S. N., Ahmadia, G. N., Andradi-Brown, D. A., Muenzel, D., Agung, F., Ford, A. K., Habibi, A., Handayani, C. N., Iqbal, M. and Krueck, N. C., 2022. The inclusion of fisheries and tourism in marine protected areas to support conservation in Indonesia. *Marine Policy*, 146, 105301.

Trialfhianty, T. I., 2017. The role of the community in supporting coral reef restoration in Pemuteran, Bali, Indonesia. *Journal of Coastal Conservation*, 21 (6), 873–882.

Turak, E. and DeVantier, L., 2011. Chapter 5: Biodiversity and Conservation Priorities of Reef-building Corals in Bali, Indonesia. *Bali Marine Rapid Assessment Program*, 78–130.

Wager, E. and Williams, P., 2013. “Hardly worth the effort”? Medical journals’ policies and their editors’ and publishers’ views on trial registration and publication bias: quantitative and qualitative study. *Bmj*, 347.

Wardana, A., 2019. *Contemporary Bali: Contested Space and Governance*. Palgrave Macmillan.

Weeks, R., Aliño, P. M., Atkinson, S., Beldia, P., Binson, A., Campos, W. L., Djohani, R., Green, A. L., Hamilton, R. and Horigue, V., 2014. Developing marine protected area networks in the Coral Triangle: good practices for expanding the Coral Triangle Marine Protected Area System. *Coastal Management*, 42 (2), 183–205.

Whitmarsh, S. K., Fairweather, P. G. and Huveneers, C., 2017. What is Big BRUVver up to? Methods and uses of baited underwater video. *Reviews in Fish Biology and Fisheries*, 27 (1), 53–73.

Wicaksana, I. B. A., 2020. 2020 Bali’s Coral Conservation: from pandemic challenge to Government coral garden project. *Bali Tourism Journal*, 4 (2), 35–39.

Wongthong, P. and Harvey, N., 2014. Integrated coastal management and sustainable tourism: A case study of the reef-based SCUBA dive industry from Thailand. *Ocean & coastal management*, 95, 138–146.

Woodhead, A. J., Hicks, C. C., Norström, A. V, Williams, G. J. and Graham, N. A. J., 2019. Coral reef ecosystem services in the Anthropocene. *Functional Ecology*, 33 (6), 1023–1034.

Yunitawati, D. and Clifton, J., 2019. Governance in the early stages of marine protected area development: A case study of Nusa Penida District Marine Conservation Area, Indonesia. *Marine Policy*, 103653.

Supplementary material

Figure S.5. 1: A working version of the model, which is based on a Microsoft Excel template: https://doi.org/10.18746/bmth.data.00000337

Figure S.5. 2: The key literature and qualitative findings (in the form of participant quotes) which were used to inform interaction values between nodes for the model: https://doi.org/10.18746/bmth.data.00000339

Figure S.5. 3: R code and priors files used to create the model: https://doi.org/10.18746/bmth.data.00000340

Table S.5. 1: Summary table of respondents' sociodemographic characteristics

Gender		**Country of origin**	
Female	32%	Indonesia	63%
Male	68%	UK	7%
Age Group		Holland	6%
16-24	22%	Belgium	2%
25-34	22%	France	4%
35-44	31%	Germany	10%
45-54	15%	Spain	3%
55-64	6%	Canada	2%
65 or over	4%	Columbia	3%

Table S.5. 2: Average weight (as per Fishbase.se (Froese and Pauly 2000)) of the 12 fish families used for RHI calculations.

Fish family	Average weight (g)
Surgeonfish (*Acanthuridae*)	473
Parrotfish (*Scarinidae*)	1800
Rabbitfish (*Siganidae*)	400
Fusilier (*Caesionidae*)	1600
Trevally (*Carangidae*)	6600
Spadefish (*Ephippidae*)	2000
Sweetlips (*Haemulidae*)	4400
Chub (*Khyposidae*)	3600
Wrasse (*Labridae*)	400
Snapper (*Lutjanidae*)	1600
Grouper (*Serranidae*)	4000
Barracuda (*Sphyraenidae)*	4000

Table S.5. 3: RHI health grades according to limit values for each indicator, according to Healthy Reefs For Healthy People Initiative and used by Díaz-Pérez et al. (2016).

RHI Category (Indicators)	Very Good (5)	Good (4)	Fair (3)	Poor (2)	Critical (1)
Coral cover %	≥40	20 – 39.9	10 – 19.9	5 – 9.9	<5
Fleshy algae cover %	0 – 0.9	1 - 5	5.1 - 12	12.1 - 25	>25
Herbivorous fish biomass (g $100m^{-2}$)	≥3,480	2880 – 3,479	1920 - 2879	961 - 1919	<960
Commercial fish biomass (g $100m^{-2}$)	≥1,680	1260 – 1,679	840 - 1259	421 - 839	<420

Table S.5. 4: Supporting quotes from participants used to justify the researchers' choice of the ten key actions and scenarios for coral reef conservation.

Scenario/ stakeholder action	Example supporting quote
1 - Constructing artificial reefs through volunteer-tourism	"*We deploy artificial reef structures to restore degraded reefs here*" (T1) "*International volunteers have allowed the programme to build one of the largest artificial reefs to exist in Indonesia*" (T10)
2 - Unregulated reef-based tourism (no specific regulations on boating, anchoring, diving etc)	"*Many regulations are still missing – for example [there is still] unsustainable fishing on the reef and no limit on the amount of divers or snorkellers*" (K2)
3 - Reef-based tourism organisations (non-government) establishing environmental standards (such as no anchoring rules, cutting boat engine rules over dive sites etc)	"*We have set standards for safety (guest to guide ratios), as well as no anchoring rules, cutting boat engine rules etc*" (N8)
4 - Using social media as a tool to increase conservation awareness (e.g. Instagram conservation pages and local WhatsApp awareness groups)	"*But as we started to build artificial reefs and document their positive results through social media, the international community became interested in our work and then we started getting a regular flow of support*" (T7)
5 - Government established local environmental regulations	"*For the dive centres here in Lembongan, we have very strict regulations on what we are allowed and not allowed to do. These are set by the government, because we are diving in a marine protected area*" (N2)
6 - Local tourism organisations contributing to marine restoration/ protection through running coral outplanting/ transplantation programmes and beach cleaning events	"*As part of our environmental protection programme, alongside running beach cleans and transplanting corals, we also run education workshops with relevant stakeholders about sustainable diving practice*" (N4)
7 - Government support for coral reef conservation (such as providing funding for artificial reef construction)	"*The local government of Tianyar Village provided some funding to buy diving equipment and build artificial reef structures*" (T1)
8 - Holding socialisation events to engage stakeholders with coral reef conservation and sustainable practices, and educate them on the mutual benefits of coral conservation / sustainable reef-based tourism	"*We need to hold socialisation to agree on protective legislations amongst all stakeholders*" (K6) "*Socialisation is very important for sustainability – people aren't really aware of the MPA zone border – this must be done*" (N2)

9 - Establishing marine protected areas	"*Now we have a locally established small MPA here*" T10 "*The government established the MPA here several years ago and it has been very effective"* (N8)
10 - Pandemics causing reef-based tourism sites to temporarily close (no visiting tourists)	"*The support and donations from foreign volunteers has decreased because of covid-19*" (T10) "*Throughout covid there were very few divers, so the corals which had been broken from tourism started to recover and grow again – but now, since the end of covid – I've seen them getting broken again*" (N1)

Chapter 6: Can coral reef conservation programmes generate changes in environmental attitudes? A case study on a rural fisher community in north Bali, Indonesia

Objectives: Investigate how a coral reef conservation programme has generated changes in the environmental attitudes of a local fisher community in Bali.

Contribution to new knowledge: This type of qualitative research on coastal communities is particularly rare in Indonesia, and insights may be useful for governing bodies and marine management authorities. It also provides a voice for communities that are generally under-represented within international research, and based on their opinions, makes a unique set of recommendations to engage local communities in coral reef conservation.

How this fits in the PhD: This researched assessed how the local community perceived the benefits of coral reef conservation, and if/how this had caused changes in the overall communities support towards the conservation projects and its objectives (mapping onto objective 4).Understanding the communities attitudes towards a given project is essential, as a project is unlikely to succeed without local support (referenced within the paper). Therefore, this paper provides an important link between the community and the benefits associated with the programme.

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Boakes, Z., Mahyuni, L. P., Hall, A. E., Cvitanovic, M. and Stafford, R., 2023. Can Coral Reef Restoration Programmes Facilitate Changes in Environmental Attitudes? A Case Study on a Rural Fisher Community in North Bali, Indonesia. Human Ecology, 51, 891-905. https://doi.org/10.1007/s10745-023-00452-7

Abstract

There is currently limited research assessing the ecological potential of coral restoration programmes of habitat enhancement and restoration of benthic and mobile populations for influencing the attitudes (and subsequent behaviours) of the communities where they are based. Our qualitative study investigated the impact of a coral reef restoration programmes on local environmental attitudes in a rural fishing community in north Bali, Indonesia. We conducted semi-structured interviews with individuals and multi-stakeholder focus groups (n=31) in Tianyar Village, where the NGO 'North Bali Reef Conservation' ('Yowana Bhakti Segara') was based. Our results highlight several factors that influenced environmental behaviours, including perceived value of coral reefs (e.g.changes in fishing yield), drivers of support for coral reef restoration (e.g., local leaders' influence) and barriers to coral reef restoration support (e.g., lack of investment). Overall, our data indicate that the restoration programme has influenced positive environmental attitudes within the community through improvements in waste management, increased support for restoration work, and the establishment of new environmental regulations. Based on our results, we make five recommendations: (1) continuing environmental education within the community, (2) strengthening regulations and improving enforcement, (3) increasing financial and logistical support for waste management and ecotourism, (4) continuing the construction and deployment of artificial reefs, ensuring 'best practice' recommendations are followed, and (5) utilising the influence of local leaders to create positive environmental behaviours.

Key words: Coral reef restoration, environmental attitudes, pro-environmental behaviours (PEB), qualitative research, fisher communities, Tianyar Village, north Bali, Indonesia

Introduction

Coral reefs are critically important to tropical coastlines, providing ecosystem services such as food provision, shoreline protection, biogeochemical cycling, and tourism (Principe et al. 2012; Woodhead et al. 2019) estimated at over US $1 trillion globally (Costanza et al. 2014). However, the health of coral reefs is declining globally at unprecedented rates (Andrello et al. 2021; IPCC 2021) resulting in losses in associated biodiversity, abundance, and reef structural complexity (Pandolfi et al. 2011; Hughes et al. 2018). Climate change induced coral bleaching is identified as the main reason for coral reef degradation worldwide (Cornwall et al. 2021; IPCC 2021). Other localised issues such as destructive/over-exploitative fishing techniques (Bacalso and Wolff 2014; Andrello et al. 2021), nutrient enrichment (Lapointe et al. 2019; Andrello et al. 2021), and pollution (Clukey et al. 2018) pose additional threats to these ecosystems. Heron et al. (2017) estimated that climate-related losses of reef ecosystem services will total approximately US $500 billion by 2100, with the greatest of these impacts experienced by people who rely upon reef services for day-to-day subsistence.

The bleaching of corals is one of the greatest threats to corals reefs worldwide (Hughes et al. 2017; Sully et al. 2019) indicating the urgent need for an immediate, large-scale reduction in greenhouse gas emissions (Ben-Romdhane et al. 2020). However, small-scale restoration tools may be utilised to capture some of the benefits of ecosystem services from healthy coral reefs to support local communities that depend on them (Hein et al. 2021; Hughes et al. 2023). These include the construction of artificial reefs (Boakes et al., 2022a), establishing propagated coral out-planting projects (Howlett et al. 2022), waste management and environmental education within the community (Sigit et al. 2019), and establishment and enforcement of marine protected areas (MPA) (Pedju 2018; Zhao et al. 2020).

Ecological 'restoration' has been defined as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER 2004). However, ecological 'conservation' describes a broader process that includes preservation and protection (Parsons et al. 2017). Our study focuses primarily on tools that actively aim to aid the recovery of previously degraded coral reefs. Consequently, we use the term 'restoration' throughout. Research indicates that programmes to restore coral reefs increase their overall sustainability and success when they involve local communities (Bennett and Dearden 2014; Kusumawati and Huang 2015; Grúňová et al. 2017). Support for marine environment restoration programmes relies heavily on local people's perceptions of personal benefit (Gurney et al. 2014; Bennett 2016), overall well-being (Diedrich et al. 2017), and /or financial gain (Berkes 2010).[1] A programme with community support will experience greater engagement from local people (ibid.) that is expected to lead to a general improvement in the community's overall support for protecting the environment (e.g., Rokicka 2002; Liu et al. 2010), although there is currently limited research among coastal communities.

Education programmes on environmental protection also potentially increase immediate and long-term community support for restoration (dos Santos et al. 2005; Leisher et al. 2012). However, multiple studies have highlighted that an increase in knowledge alone is insufficient for

[1] For example, a fisher increasing yield due to higher fish biomass as a result of 'the spill-over effect' from a marine protected area (MPA) (Di Lorenzo et al. 2020; Lenihan et al. 2021), or an increase in tourism related jobs (Mangubhai et al. 2020).

substantial changes to a community's support for coral reef restoration (Brown et al. 2017; Grúňová et al. 2017; Trialfhianty 2017). Several other factors in engaging local communities in marine restoration programmes: (1) Inclusion of local people in restoration decision-making processes (Lundquist and Granek 2005) (e.g., compliance with MPA regulations has been shown to be higher when local fishers are involved in their creation (Glaser et al. 2010)); (2) establishing regulations that are clearly understood by local people (Suparno et al. 2019) and ensuring that they are effectively enforced by a respected authority (Doherty et al. 2013); and (3) influence from local leaders has been shown to 'bridge the gap' between local people and marine restoration objectives, and also to promote positive environmental attitudes[2] within the community (Trialfhianty 2017).

Environmental psychologists have employed various models to gain a deeper understanding of what shapes EAs and thus motivates subsequent PEBs (see Schwartz 1977; Dunlap and Liere 1978; Ajzen 1985; Stern et al. 1999). We identified three constructs emerging from these theories: i) 'Attitude' reflects an individual's or community's perception of engaging in a particular behaviour (e.g., perceived personal financial gain); ii) 'Subjective Norm' refers to the belief that other individuals or groups will approve or disapprove of a given behaviour (e.g., influence from local leaders (Ajzen 1985)); iii) 'Perceived Behavioural Control' reflects the perceived difficulty of enacting a given behaviour, and is based on relevant factors that may facilitate or impede it (e.g., lack of investment). It is expected that individuals with differing levels of these three theoretical constructs will systematically differ in their EAs and PEBs (Aral and López-Sintas 2023).

Our aim was to investigate if and how a coral reef restoration programme in Tianyar Village, north Bali, had facilitated changes in environmental attitudes and behaviours within the local community through qualitative research. More specifically, we were interested in understanding the attitudes, subjective norms, and perceived behavioural control that influenced a community's support for their local coral reef restoration programme (Bennett and Dearden 2014; Kusumawati and Huang 2015; Grúňová et al. 2017). Research on EAs of coastal communities is limited and

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[2] Environmental attitudes (EAs) are important in the field of environmental conservation and restoration because they often determine behaviours (of an individual or a community) that can impact environmental quality (Milfont 2007; Gifford and Sussman 2012). Individuals' attitudes are formed from their experiences, social factors, and observational learning (Cherry 2018), and their internal and stable responses to objects, ideas, or people are reflected in their EAs. These can be specific to behaviours, such as perceived behavioural importance, or based on value orientations, such as ecocentrism (Naiman et al. 2023). EAs are strong positive predictors of pro-environmental behaviours (PEB) (Ertz and Sarigöllü 2019), which can be defined as all possible actions aimed at reducing threats to and/or safeguarding the environment (Steg and Vlek 2009). Hofman et al. (2020) created a list of 34 PEBs that can be undertaken by individuals to protect the marine environment, including reducing /refusing plastics, following good diving / snorkelling etiquette, and volunteering time to support environmental causes. The social landscape of a community strongly influences individuals' EAs (Mainzer and Luloff 2017), which in some cases, can lead to poorly informed behaviours that are damaging to the environment (Moran 2016).

inconclusive in Bali and wider Indonesia. However, it has been shown that environmental issues Indonesia could be greatly reduced through PEBs (see Ulhasanah and Goto 2018; Adam et al. 2021). Our research is especially pertinent because it engages with the opinions of generally under-represented communities in a low-middle income nation to gather data for a unique set of recommendations for improving coral reef restoration in other parts of the world.

Study Site

We conducted our qualitative research in Bali, an island province of Indonesia during July and August 2021 (figure 6.1). Indonesia is a low-middle income country (Sujarwoto et al. 2018) and contains 12.5 % of the world's total coral reef area (Susiloningtyas et al. 2018). Bali has the second highest documented reef fish species richness in the Asia-Pacific. Data from 2017 indicated that 50% of Bali's corals are in good health, while 20% are declining, and 30% are poor (Marine and Fisheries Office 2017 data, as cited in Wicaksana 2020). The primary reasons for the decline are associated with climate change (Prasetia et al. 2017; Suparno et al. 2019; Tito et al. 2019), destructive fishing practices (Doherty et al. 2013' Frey and Berkes 2014) and marine pollution (Germanov et al. 201;, Suteja et al. 2021).

Methods

Study Site

Following a decline in coral health in recent decades, the pro-active restoration of Indonesia's reefs has been initiated by community groups, international NGOs, and the government (both on a local and central scale). The most notable restoration tools employed include the establishment of three MPAs (Pedju 2018), deployment of artificial reefs as habitat enhancement tools on many degraded reefs (Wicaksana 2020; Boakes et al. 2022a), and the development of ecotourism destinations that promote coral reef restoration (Trialfhianty 2017). Sustainability-related programmes, specifically in Bali, have been shown to obtain greater support from the wider community when the initial ideas are discussed at community meetings (*desa adat*; table 6.3) (Trialfhianty 2017; Wardana 2019; Yunitawati and Clifton 2019).

We collected qualitative data in Tianyar Village in Bali's Karangasem regency, where primary occupations are fishing and selling fish (De Brauwer et al. 2017). The 3km coastline includes a natural coral reef considered healthy (>40% coral cover, ≥3,480g/100m^2 herbivorous fish biomass and ≥1,680g/100m^2 commercial fish biomass (Díaz-Pérez et al. 2016) as well as an empty, degraded area (<5% coral cover, <960g/100m^2 herbivorous fish biomass and ≥420g/100m^2 commercial fish biomass (Díaz-Pérez et al. 2016)), where reefs were destroyed by unsustainable fishing techniques and boat anchoring (personal communications). Tianyar village attracts relatively few tourists, especially in comparison to the mass tourism areas in the south of the island.

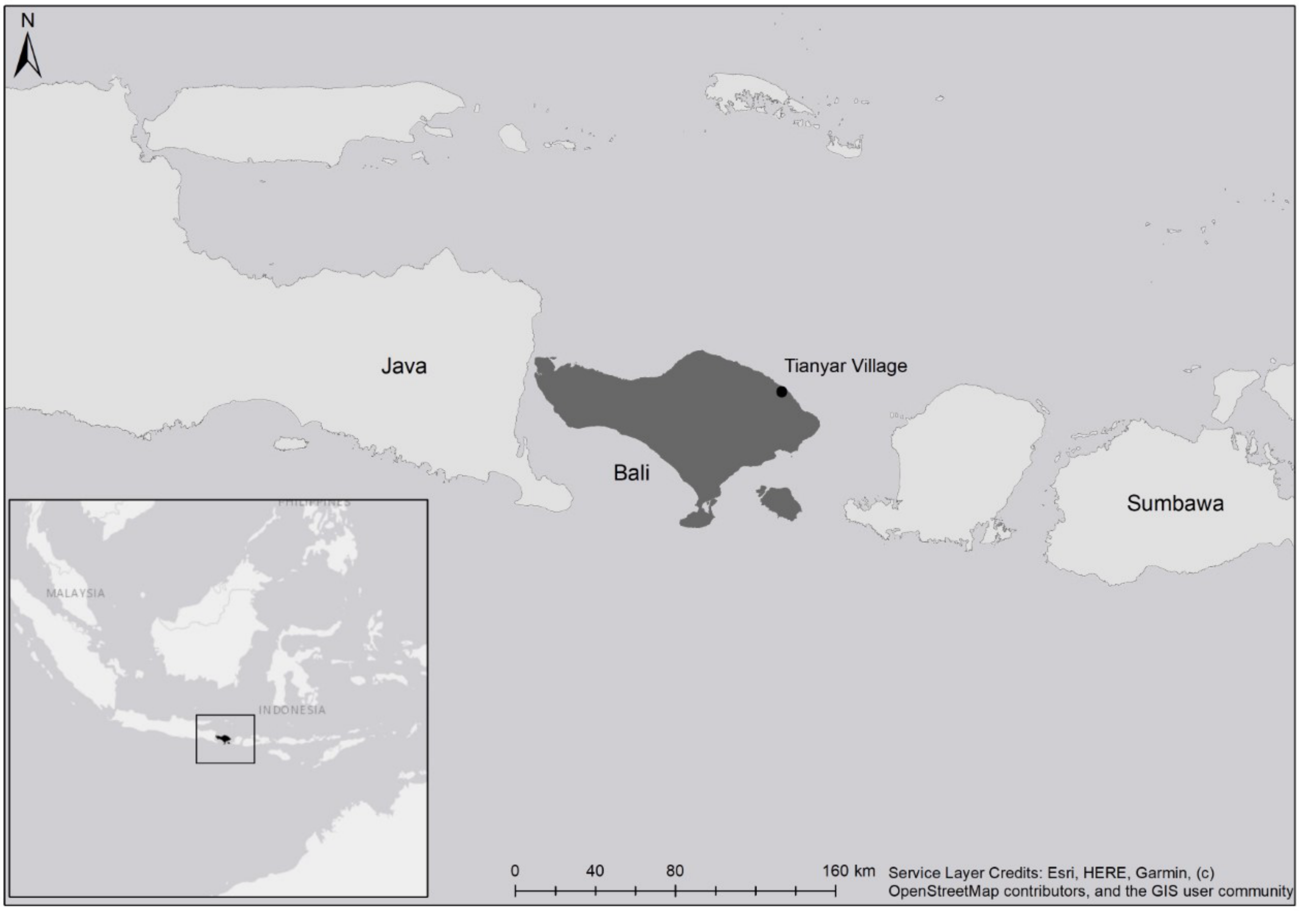

Figure 6. 1: Location of Tianyar Village and Bali within Indonesia *(created using ArcGIS OpenStreetMap powered by Esri).*

The coral reef restoration non-government organisation (NGO) 'North Bali Reef Conservation' (locally known as 'Yowana Bhakti Segara') was based in the village at the time of data collection. Established in 2017, NGO was well-known for its community coral reef restoration efforts, notably the deployment of approximately 15,000 artificial reef (AR) structures (1m x 0.5m) in areas of previously destroyed reef. Its work is funded by ongoing international donations and occasional government grants, and (at the time of data collection) was the only organisation of its kind in the local area. Its ARs are located inside a no-take-zone MPA established and regulated by the local community, which was familiar with foreign-assisted coral restoration projects and was involved in the establishment of environmental targets, as well as providing scientific and logistical support (personal communication). In an earlier study (Boakes et al. 2022b) we described the work of this community in successfully restoring an area of reef in North Bali to its earlier level of marine biodiversity similar to a nearby healthy natural reef. Based on this previous research, we were able to assess how the EAs of the community in Tianyar Village have changed as a result of the restoration programme. Social research is rarely undertaken in the region, especially in area of EAs in coastal communities, allowing new insights for local governing bodies and marine management authorities (see also e.g., Bennett and Dearden 2014; Kusumawati and Huang 2015; Grúňová et al. 2017).

Interviews and Focus Group Discussions

Between July-August 2021 we conducted both semi-structured interviews and multi-stakeholder focus group discussions with 31 participants. Following Gelcich et al. (2009), we conducted 11 in-depth, semi-structured interviews with key informants of groups from a cross section of the community, including community leaders (from the local government, educational institutions, businesses and religious groups), fishers, fishmongers, tourism workers, and school students. Additionally, following Legare et al. (2020), we conducted two multi-stakeholder fisher focus group discussions each of 10 participants, which allowed us to assess if individual opinions differed in a group rather than face-to-fact context (Kellmereit 2015).

Participants for both interviews and focus groups were selected purposefully aided by a village leader familiar with the community, based on our perceptions of how a participant might enhance our understanding of how coral reef restoration activities had influenced environmental attitudes in the community (Kuper et al. 2008; Creswell and Creswell 2017). Participants were selected based on the nature of their employment (e.g., fishers), perspective (whether for and against coral reef restoration – as advised by members of the community), and/or social diversity (e.g., age, gender, and educational background). Chosen participants were initially approached (either via a phone call or in-person) and asked if they were willing to participate in our study (Kruglov and Davidson 1953). Out of the 32 people we approached, one declined. Participants were given written information about our research goals and methods and Bournemouth University's ethical review process (reference number: 37431), and asked to indicate their consent to participate in the project. Following Wager et al. (2013), to reduce bias in the interviews, We further explained that their names and responses would remain anonymous and be allocated a code (e.g., interviewee 1 = I1 / focus group 1 = FG1) to ensure anonymity. The interviews and focus groups were conducted in a variety of locations according to the preference and availability of the interviewee. In the instances where participants expressed no preference, interviews were conducted in a private meeting room. The interviews were conducted in a mixture of Indonesian and Balinese language by a local researcher fluent in both languages.

The Theory of Planne Behaviour (TPB) was used as a framework to categorise key topics emerging from the interview responses (Kumar-Chaudary et al. 2017; Steg and de Groot(2018). Based on the literature (and our understanding of the topic and case study at the time), our framework identified three main factors that affected attitudes and behaviours towards coral restoration, which we used to design our interview and focus group questions (table 6.1).

Table 6. 1: Factors affecting attitudes towards coral restoration and associated interview questions.

Key factor	Associated interview questions	Relevant literature
Perceived value of coral reefs	- *Do you think there a link between coral reef conservation and ecotourism development? And if so, please explain.* - *Has you experienced increases in fishing yield as a result of the coral reef conservation program? And [if yes or no], why do you think this is?*	Schwartz 1992, 2012 Choi and Lee 2012 Woo and Kim 2019 Kim et al. 2020 Rizzi et al. 2020
Drivers of support for coral reef restoration	- *Who are the people leading coral reef conservation here? What have these people done to engage the community in the project?* - *What makes people want to support coral reef conservation here?* - *How do you find out information and/or news about coral reef conservation?*	Hungerford and Volk 1990 Diedrich 2007 Berkes 2010 Bennett and Dearden 2014 McLeod and Palmer 2015 Bakari et al. 2017 Grúňová et al. 2017 Trialfhianty 2017 Rizzi et al. 2020
Barriers to coral reef restoration support	- *What is needed to develop coral reef conservation community support here?*	Steg and Vlek 2009 Doherty et al. 2013

	- *What are the reasons people here may choose to not support coral reef conservation?*	Kostić and Petrović 2013 Nordfjærn et al. 2014 Mahyuni 2016 Suparno et al. 2019

Interviews and focus groups were recorded on a SONY ICD -UX533F recorder, and initiated with general questions or conversation topics such as “Tell me about your typical day,” and then proceeded to topics related to food, work, or the marine environment (Grimm and Needham 2012; Patton et al. 2014) before addressing more specific topics (table 6.1). Not all questions were asked in every interview, but chosen according to the interviewee’s background (e.g., only fishers or fish sellers were asked about fishing yield).[3] Focus group discussions lasted between 1-2 hours, and all interviews lasted between 30 minutes to 2 hours.

Sample size and characteristics

Justification of sample size often depends on the research topic, quality of data, cultural factors, and interviewees’ responses (Morse 2000; Marshall et al. 2013; Patton 2014). Mason (2010) reported that, from 560 studies, the mean sample size was 31, and Marshall et al. (2013) suggested that single case studies should include 15 to 30 interviews. However, qualitative researchers generally agree that rather than pre-determining a sample size, it is more useful to finish at a given degree of saturation (Moura et al. 2021). ‘Saturation’ refers to the point at which additional interviews no longer offer new insights and information about the given topic (Charmaz 2006; Dworkin 2012). Following Moura et al. (2021), we decided that when we reached saturation we would interview two more participants. We reached this point after 31 interviews (11 one-one interviews, and two focus groups each with 10 people). Due to the similarity of responses between focus groups and individuals, we also decided that findings in the results section would be presented jointly.

[3] We provide a glossary (table 6.3) for multiple important key words used by participants (in Indonesian and Balinese) that have no direct translation to English.

Table 6. 2: Respondents' sociodemographic characteristics, highlighting the percentage of respondents within a given group.

Gender		**Occupation**	
Female	**27%**	Government and policy	**18%**
Male	**73%**	Education	**9%**
Prefer not to say	**0%**	Tourism	**18%**
Age Group		Fishing / selling fish	**36%**
16-24	**9%**	Non-government organisation	**9%**
25-34	**18%**	Student	**9%**
35-44	**36%**	**Highest Level of Education**	
45-54	**27%**	Elementary school	**36%**
55-64	**9%**	Middle school	**27%**
65 or over	**0%**	High school	**18%**
		Undergraduate degree	**9%**
Tianyar Village, Bali	**100%**	Postgraduate degree	**9%**

Additionally, finding female participants willing to be interviewed proved problematic (table 6.2). However, we were able to recruit female respondents from a wide range of occupational and social groups (e.g., fish sellers, education workers, students, tourism workers). Interviews with female respondents were discontinued after it was clear that we had recorded a wide variety of their opinions from a broad range of groups.

Qualitative analysis

Interviews were translated into English and then transcribed by ZB and LMP (authors fluent in both languages). This was done manually due to the lack of speech recognition software in Indonesian and Balinese. We then followed four stages: (1) coding, (2) assigning themes, (3) structuring and (4) comparing answers between interviews and focus groups.

Stage 1 followed the thematic coding analysis guidelines of Braun and Clarke (2006), which involved the generation of numerous category codes, without limiting the number of codes (Charmaz 2006). For stage two, we listed key emerging ideas (McKinley and Ballinger 2018; Saldaña 2021) from words or phrases interviewees used frequently (Nyumba et al. 2018). Each interview question (table 6.1) directly corresponded with one of the key emerging themes. Stage 3 involved identifying reoccurring themes with connecting and/or opposing views (Charmaz 2006). Finally, in stage four we compared the responses of the individual participants with those of the focus groups.

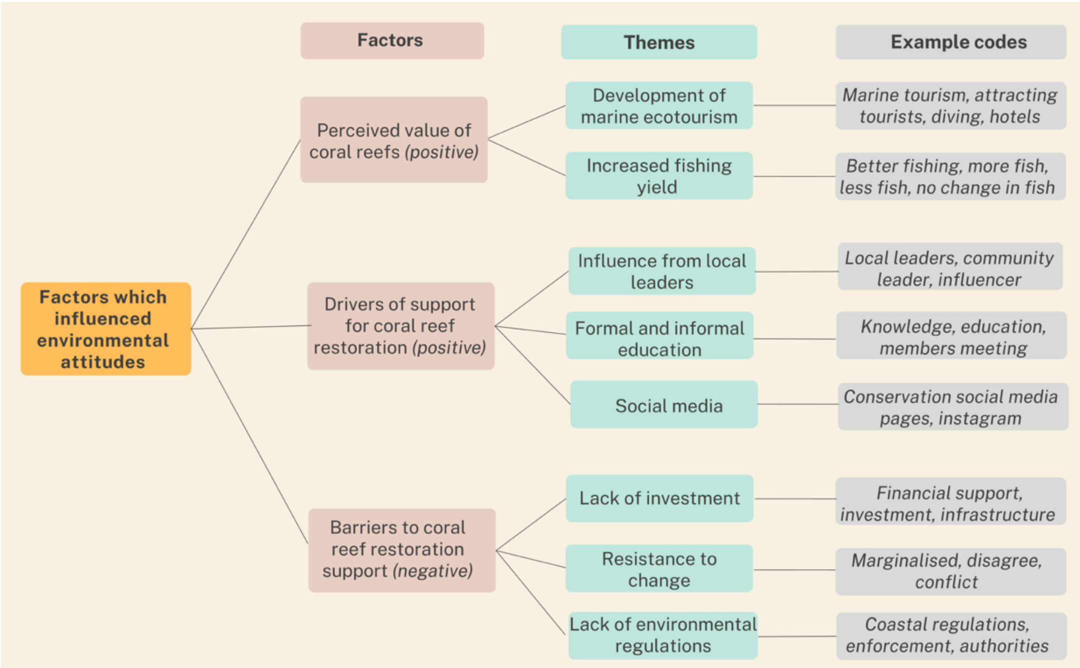

Figure 6. 2: Connecting diagram highlighting the factors which were shown to influence the communities environmental attitudes towards coral reef restoration, as well as key themes and examples of their indicative codes.

Results and Discussion

Table 6. 3: Glossary of key words used by participants within interviews.

Key words	Meaning
Desa Adat	Customary village (semi-autonomous village governance system that is responsible for organising religious ceremonies and socio-cultural activities). Desa Adat has the authority to produce its own rules based on a members agreement.
Desa Dinas	Administrative village (village governance system that is responsible for managing government-related administrative matters)
Banjar	A small unit of a community group that share responsibilities to perform religious ceremonies and socio-cultural activities. Desa Adat consists of several Banjars.
Awig-awig	Laws produced by Desa Adat
Pecalang	Desa Adat security force
Yayasan	Non-profit foundation

Perceived value of coral reefs

The community's perceptions of the value of coral reefs emerged as a key positive factor influencing EAs towards coral reef restoration and were divided between two main themes (1) development of marine ecotourism, and 2) increased fishing yield (figure 6.2). The interviewees described how Tianyar's coral reefs had earlier been exposed to coral mining (manual removal of patches of reef used a valuable construction material) that resulted in serious coral degradation. Since these harmful activities were stopped, an improvement of EAs in the village has led to the (reported) recovery of the marine environment:

> "Comparing the situation now to 5 years ago, we have observed a substantial improvement in coral health and overall cleanliness of the environment" (FG1).

Interviewees described the coral reef restoration work that has taken place:

> "Before the conservation programme started, there was only bare sand, but now after we have deployed around 8000 artificial reef structures, a large number of fish populations have come back again. The corals now grow by themselves, so natural recruitment is happening, without our intervention to transplant the corals" (I1).

Several participants commented that this work has had direct benefits on the marine environment and stated that after 4 to 5 years of restoration, the coral reef is now in good condition. Two main reasons emerged for community support of these restoration efforts: concern for the environment and perceived economic prospects associated with coral reefs, mostly ecotourism development (cf. Rizzi et al. (2020) who showed that perceived financial self-benefit is an important factor in driving PEBs). Generally, our participants indicated enthusiastic support for the development of marine ecotourism due to its potential to provide new, relatively well-paid jobs and improve livelihoods:

> "Tianyar has high potential to be developed as a marine tourism destination, because we have very beautiful natural coral reefs and many variations of fish species" (I1).

> "I am hoping to become a dive guide or instructor so that I can teach new guests to dive" (I5).

It has been widely acknowledged that eco-tourism developments can provide economic opportunities to areas with high unemployment (Garrod et al. 2003; Shani et al. 2012). Alongside generating socio-economic benefits, eco-tourism can also help to protect (and often pro-actively restore) local environments (Mangubhai et al. 2020).

Our respondents indicated that the community's reef restoration efforts in Tianyar were largely driven by motivations to improve their livelihoods, for example: "the conservation programme has improved our incomes and quality of life, and that is one primary reason we choose to support it" (FG1). Berkes' (2010) study on a fisher community in Les Village, Bali, found that fishers chose to engage in their local coral restoration project because they felt they would benefit financially from doing so (notably through increasing their fishing yields). Additionally, our personal communication with the local community in Nusa Penida (Boakes et al. 2022a) showed that coral restoration had led to the generation of tourism, which created new, higher-paid jobs for local people. These improvements in livelihoods generated local support to continue work to protect local coral reefs (cf. Romañach et al. (2018) regarding mangrove reforestation in India). Concerns were raised several times during our interviews that many people had joined restoration efforts purely for economic reasons, without a genuine desire to protect the environment. For example:

> "I don't think the local people genuinely care or are aware about environmental protection. Even though they join conservation groups, I have observed that their actions are not representative of environmental awareness at all. For instance, they keep throwing away waste while they are sitting on the beach" (I1).

Stem et al. (2003) and Boakes et al. (2022a) note that it can be problematic if support for marine restoration is driven purely by financial gain because it will consequently diminish should profits decrease.

One interviewee (I2) described the local community as “50% of people as fisher, 30% as farmer, 20% as trader,” reflecting the continued centrality of fishing for the local community and their perceptions of the value of coral reefs. Research has shown that localised fisheries yield can increase as a result of artificial reefs (ARs) through increased production of commercial species (Santos and Monteiro 1997; Ramos et al. 2019) as well as establishment MPAs as a result of the ‘spill-over effect’ (Di Lorenzo et al. 2020; Lenihan et al. 2021). Some interviewees discussed the benefits of the restoration work to fish populations:

> “[because of the MPA and ARs] fish have come back to the area, particularly fish that are commonly consumed by local people, such as snapper fish. There are [now] so many of them and this is benefitting local fishers” (I1).

Despite this, most fishers did not report experiencing an increase in fishing yield, likely because the species they target are often caught far from the AR / MPA, and are not species generally found on the artificial reef. It is also important to note that ‘no change’ in fishing yield may be positive, as MPA establishment can sometimes lead to initial reductions in yield, especially if it prohibits fishing in a previously productive fishing area (Goñi et al. 2011).

Drivers of support for coral reef restoration

Drivers of support for coral reef restoration fell into three main themes (1) influence from local leaders, (2) formal and informal education, and (3) social media (figure 6.2). Bakari et al. (2017) showed that successful changes in EAs are often created by local leaders who influence attitudes among their constituents, a finding reflected in our interview data:

> “[local leader name] has an important role in influencing local people too, for example through beach cleaning, or turtle hatchling protection” (I4).
>
> “Local leaders encourage people that are throwing away litter on the beach to change their habits. The leaders can influence people's behaviours” (FG1).

Local leaders who have positive EAs and have influenced others within the community were described as having: “connected foreign volunteers and scientists with members of the fisher conservation group and the general community” (I1) (see also Schwartz 1992, 2012; McLeod et al. 2009; Frey and Berkes 2014; Trialfhianty 2017).

Other reasons for local support of restoration efforts include the view that the sea is a source of food and livelihood:

> "As indigenous people in Tianyar, the sea is important as a food source for us and for many of my family's livelihoods. This for us is an important reason to protect the marine environment" (I7).

Another respondent cited Hindu religious beliefs and traditional scriptures as a reason to support restoration efforts:

> "In Sundari Bungkah Lontar [traditional scripture] it is mentioned that the function of the sea on earth is like vital arteries in our body. So if the sea is unhealthy it will make us unhealthy as well" (I2).

Our interviews also revealed limited environmental knowledge among community members. For example:

> "Local people lack knowledge and understanding about the importance of conserving the environment" (I3)
>
> "Information about coral conservation and the environment has been spread traditionally through members meetings" (FG1),

which was described as "an ineffective way of communicating environmental issues and coral reef conservation" (FG2). Fishers have a relatively limited understanding of the ecological and socio-economic benefits of coral reef restoration since there is very limited formal education on environmental awareness in Bali and wider Indonesia (Parker and Prabawa-Sear 2019). One respondent I1 discussed the link between high-level formal education and environmental care: "I think there is a link between the level of formal education and the level of environmental awareness. Those who have studied [at university] in cities have seen good waste management systems, with waste bins and plastic sorting points. When they come back to the village they won't throw away their waste anymore, but will find a bin instead" (1I) (see also Littledyke 2008**;** Strieder-Philippssen et al. 2017).

However, interviewees explained that informal teaching sessions with children in the village appear to be an effective method of environmental education:

> "Before the educational programme [Yayasan Widya Sari] started, most young people had no idea about the marine life on their beach. They now know how beautiful their local

marine life is because they have been able to go snorkelling with international volunteers and see it” (I5).

These sessions were provided mostly by international volunteers, many of whom visit specificially to teach students about the environment:. “The environmental activities were first initiated by international volunteers here and I believe they have played an important and positive role in changing local people’s attitudes” (I5). Throughout our study we found local perceptions on the presence of foreigners were generally very positive (however, see also Cohen 1982; Fabinyi 2010; Boakes et al. 2022a; among others).

The importance of informal education as a powerful tool for changing EAs has been widely noted (Steg and De Groot 2018, Varela-Candamio et al. 2018; Parker and Prabawa-Sear 2019). Our interviews indicated that the youngest generation in Tianyar are the most aware about the environment (see also Williams et al. 2011). Based on the Environmental Citizenship Model (Hungerford and Volk 1990), our study community in this case study can be described as at an earlier stage of educational involvement (basic sensitivity to and knowledge of the environments). The community’s level of involvement environmental initiatives would largely depend upon their education and awareness of the environment. Further increasing environmental education within the village would encourage more of the community towards ‘ownership’ and ‘empowerment’ variables, which would further improve EAs, generate PEBs and lead to greater support for coral reef restoration

Barriers to coral reef restoration support

Barriers to coral reef restoration support proved to be a key negative factor influencing EAs towards coral reef restoration and comprised three main themes: (1) lack of investment, 2) resistance to change, and 3) lack of environmental regulations (figure 6.2). Lack of investment was consistently cited as the main factor hindering the development of ecotourism, especially in terms of waste management:[4]

> “…we want to improve our waste management, but we haven’t received the financial support to do it” (I6).

[4] The lack of waste management on the island, as highlighted by the interviews, caused Bali to declare a state of ‘Garbage Emergency’ in 2017 (Garcia et al. 2019).

"There is still limited help from the government, and no trucks taking the plastic away" (I5).

"…most waste gets thrown away or goes into holes that are dug into the ground, or is burnt" (I7).

Substantial financial investments are often needed to create long term changes to EAs (Lavelle et al. 2015).[5] Regions in north Bali receive relatively few tourists compared to mass tourism centres in the south and are consequently not a likely priority for investment (Kostić et al. 2013; Khamdevi and Bot 2018).

Resistance to change, the second barrier to reef restoration efforts, can be a contributing factor that predicts behavioural intention and thus behavioural outcomes (Nordfjærn et al. 2014). Some conflicts and issues related to ecotourism were reported by fishers:

"…there are some people who argue with us when no fishing zones are established. They insist that the sea belongs to everyone, and they can fish anywhere they want" (FG2).

"So far, we haven't experienced issues, but we hope we can still do our job as fishers [with the development of ecotourism]. If we are marginalised, I think we will fight. It is very important to make conservation, tourism, and fishing zones" (FG1).

"…. potential issues with stakeholders, for example "boat owners complaining about divers or snorkellers in the areas where they went to fish" (I4).

It appeared that this resistance to change was driven by concern for their livelihoods and culture, which they fear might impacted by the proposed development. Marine restoration and eco-tourism projects are far more likely to succeed when stakeholder opinions are listened to and their concerns addressed. In the case of development of eco-tourism in Tianyar, it is necessary to accommodate the concerns of stakeholders within planning processes (Waayers et al. 2012; Pedju 2018). Responses showed that most resistance to change (especially in terms of waste management) came from the older generation. Intervention strategies may improve EAs, including provision of education and consideration of stakeholder feedback[6], as well as setting community goals with signed pledges (Steg and De Groot 2018).

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[55] In contrast, Terrier and Marfaing (2015) showed that although large environmental initiatives sometimes require substantial financing, other small behavioural changes (such as reducing single use plastics) are much less onerous.

[6] This is particularly relevant in terms of the community's lack of environmental knowledge and waste management problem. However, this is a particularly sensitive topic, and if someone's waste management practices are criticised: "…

Participants generally agreed that local environmental regulations, as well as enforcement practices to support them, are insufficient and hindered positive changes to EAs:

"We need more rules from the local government for environmental protection" I4.

"… authorities need to be involved with strictly enforcing environmental regulations. Otherwise local people will not follow them" (I8).

This latter view is also reflected in the West Bali Marine Park MPA, where poorly enforced regulations led to user non-compliance (Doherty et al. 2013) , suggesting that enforcement of regulations by authorities are necessary for success. Additionally, one informant noted that:

"There is a lack of communication of environmental rules and this needs to be made clearer. For example, sometimes fishers aren't certain if they're allowed to fish in an area or not - this is a common theme across Bali's marine protected areas" (I4). [7]

Some respondents commented that plans were being developed to create regulations:

"… we are planning to create village rules to protect the environment, particularly the marine environment. But it may take quite a long time to do this" (I2).

FG2 noted the importance of the Desa Adat and the Pecalang (table 6.3):

"… the pecalang play an important role in Desa Adat law enforcement. These rules are often more effective than the government laws that are enforced by police because people are more fearful of Pecalang."

Recommendations

The following recommendations are based on the responses and opinions of interviewees.

Continuing community environmental education

There should be continued support for the work of the 'Yayasan Widya Sari' (the local learning centre) and similar initiatives to raise young people's awareness about environmental issues and marine restoration (Varela-Candamio et al. 2018; Blythe et al. 2021). Engaging with 'The Environmental Citizenship model' (Hungerford and Volk 1990) would further increase environmental

they will be offended, and this will trigger conflict between us. Local leaders need to approach them and talk to them personally" (FG2).

[7] See also Suparno et al. (2019).

education within the village encouraging more community identification with 'ownership' and 'empowerment' that would lead to greater support for coral reef restoration. As our results indicate that Tianyar fishers have a relatively limited understanding of the economic benefits of coral reef restoration, specifically with regards to increased fishing yield, we recommend further resources be allocated to increase their awareness of these benefits, which would likely generate more support for the restoration programme (Leisher et al. 2012).

Strengthening regulations and improving enforcement

In consultation with stakeholders the local government should strengthen environmental regulations in the village, specifically with regards to waste management, including imposition of fines for disposing of waste on the beach and other public areas, as well as creating clear zones for marine users. These regulations need to be strictly enforced by the relevant authorities to ensure compliance. We also recommend that alongside the establishment of official government regulations, the Desa Adat (and their associated Pecalang security force) establish and enforce locally specific laws to protect the environment.

Increasing support for eco-tourism

Village leaders' ability to approach local government and access various state-owned enterprises and social responsibility government grants is crucial to develop ecotourism within the village (Bhuiyan et al. 2011). However local universities may also assist village leaders in creating ecotourism initiatives, as well as expand potential sources of grants. As with all our recommendations, we emphasize the crucial importance of engaging local stakeholders in the development of ecotourism initiatives.

Continue constructing artificial reefs

ARs have been used across Bali as a habitat enhancement tool to successfully restore marine biodiversity and abundance (Syam et al. 2017, Puspasari et al. 2020) while at the same time enhancing positive local attitudes towards their environment. In terms of achieving restoration objectives, it is important that programmes follow guidelines of 'best practice' (Boakes et al. 2022b).

Utilising the influence of local leaders

The influence of local leaders in shaping EAs within the local community is widely recognised (McLeod and Palmer 2015; Trialfhianty 2017; Steg and De Groot 2018), including the use of intervention strategies to increase pro-environmental behaviours through the widespread provision education and feedback, as well as setting community goals with signed pledges. This is particularly relevant in terms of Tianyar's communities lack environmental knowledge and waste management problem.

Many of our recommendations for our study in one village in Bali are also relevant for marine and terrestrial restoration in the global north. Embedding the generation of positive EAs within the wider community, based on clear personal gains, social norms and overcoming barrier to change is very different to many initiatives used in global north countries, where the overwhelming response to plastic pollution has been based on an approach encouraging use of reusable products, such as water bottles or coffee cups (Stafford and Jones 2019a). However, the hope that undertaking one PEB will lead to 'spill-over' into more beneficial behaviours is disputed by current evidence (Maki et al. 2019; Stafford and Jones 2019b). A holistic, community-based approach such as we recommend here or as adopted by the Coast4C projects in the Philippines (Blanco 2021), may provide the necessary conditions to facilitate positive EAs in coastal communities, but such approaches may require development, or redevelopment, of closer knit communities than currently exist in many countries (e.g., Monbiot 2017).

Conclusion

Our results highlight several factors that influence local attitudes towards the restoration of coral reefs, including perceived value of coral reefs (such as perceived changes in fishing yield), drivers of support for coral reef restoration (such as influence from local leaders), and barriers to support for coral reef restoration (such as lack of investment), and suggest that the restoration programme had influenced EAs within the community, which potentially have led to an increase in PEBs (notably, increased support for the coral reef restoration programme and its objectives). These behavioural changes are mostly driven by the perceived economic prospects that the community associate with restoration programme. This qualitative research adds new knowledge to the existing scientific literature on the topic of EAs and coral reef restoration programmes, however this case study is limited to one fishing village in north Bali. It is recommended that qualitative research continues to be conducted in Indonesia (and other low-middle income nations) to further investigate the link between ecological restoration and EAs.

Statements and Declarations

Conflict of interest

Author (ZAB) was involvement with the initial start-up and management of NGO. He personally knew some of the participants, which may be considered as potential competing interests. Participants responses were anonymised throughout. After fieldwork finished, research results were presented to local stakeholders to maintain objectivity. ZAB declares no financial conflicts of interest.
All other researchers had connections with the NGO and declare no other conflicts of interest.

Research ethics

We further confirm that any aspect of the work covered has involved human participants was approved by the Bournemouth University Ethics Committee (reference number: 37431). Written consent to publish potentially identifying information, such as details or the case and photographs, was obtained from the patient(s) or their legal guardian(s).

Data availability statement

To remain GDPR compliant on the holding of social and economic data the aggregated data is available on request to the corresponding author.

Permits

A research permit was obtained from Indonesia’s Ministry of Research (BRIN).
Research permit number: 15/SIP.EXT/IV/FR/5/2023

Funding

Zach Boakes, the first author, was supported by a studentship with Bournemouth University, UK, as well receiving the 2021 ‘Emerging Scientist’ grant from Earthwatch Institute. Luh Putu Mahyuni was supported by Universitas Pendidikan Nasional through a research grant scheme number: 136/I-5/UND/III/2021.

References

Adam, D. H., Siregar, Z. M. E., and Supriadi, Y. N., 2021. Environmental Concern and Environmental Knowledge, Attitude toward Pro-Environmental Behavior as Predictors of Pro-Environmental Behavior: Evidence from Textile Industry in Indonesia. *Calitatea*, 22 (182), 138–144.

Ajzen, I., 1985. From intentions to actions: A theory of planned behavior. *In*: *Action control*. Springer, 11–39.

Andrello, M., Darling, E. S., Wenger, A., Suárez-Castro, A. F., Gelfand, S., and Ahmadia, G. N., 2021. A global map of human pressures on tropical coral reefs. *Conservation Letters*, e12858.

Aral, Ö. H. and López-Sintas, J., 2023. Environmental behavior patterns across clusters of European Union countries: Uncovering heterogeneity in the attitude-behavior-context relationship. *Journal of Cleaner Production*, 135936.

Bacalso, R. T. M. and Wolff, M., 2014. Trophic flow structure of the Danajon ecosystem (Central Philippines) and impacts of illegal and destructive fishing practices. *Journal of Marine Systems*, 139, 103–118.

Bakari, H., Hunjra, A. I., and Niazi, G. S. K., 2017. How does authentic leadership influence planned organizational change? The role of employees' perceptions: Integration of theory of planned behavior and Lewin's three step model. *Journal of Change Management*, 17 (2), 155–187.

Ben-Romdhane, H., Jabado, R. W., Grandcourt, E. M., Perry, R. J. O., Al Blooshi, A. Y., Marpu, P. R., Ouarda, T. B. M. J., and Ghedira, H., 2020. Coral reefs of Abu Dhabi, United Arab Emirates: Analysis of management approaches in light of international best practices and a changing climate. *Frontiers in Marine Science*, 7, 541.

Bennett, N. J., 2016. Using perceptions as evidence to improve conservation and environmental management. *Conservation Biology*, 30 (3), 582–592.

Bennett, N. J. and Dearden, P., 2014. Why local people do not support conservation: Community perceptions of marine protected area livelihood impacts, governance and management in Thailand. *Marine policy*, 44, 107–116.

Berkes, F., 2010. Linkages and multilevel systems for matching governance and ecology: lessons from roving bandits. *Bulletin of Marine Science*, 86 (2), 235–250.

Bhuiyan, M. A. H., Siwar, C., Ismail, S. M., and Islam, R., 2011. The role of government for ecotourism development: Focusing on east coast economic region. *Journal of social sciences*, 7 (4), 557.

Blanco, A., 2021. *Making seaweed farming even better for the ocean – watch our brand new video to see how we reduce plastic waste. Coast4C: Cronulla AU.* [online]. Available from: https://coast4c.com/how-do-we-reduce-plastic-pollution-from-seaweed-farming-watch-out-brand-new-video/.

Blythe, J., Baird, J., Bennett, N., Dale, G., Nash, K. L., Pickering, G., and Wabnitz, C. C. C., 2021. Fostering ocean empathy through future scenarios. *People and Nature*.

Boakes, Z., Hall, A. E., Ampou, E. E., Jones, G. C. A., Suryaputra, I. G. N. A., Mahyuni, L. P., Prasetyo, R., and Stafford, R., 2022(a). Coral reef conservation in Bali in light of international best practice, a literature review. *Journal for Nature Conservation* [online], 126190. Available from: https://doi.org/10.1016/j.jnc.2022.126190.

Boakes, Z., Hall, A., Jones, G., Prasetyo, R., Stafford, R., and Yahya, Y., 2022(b). Artificial coral reefs as a localised approach to increase fish biodiversity and abundance along the North Bali coastline. *AIMS Geosciences* [online], 8 (2), 303–325. Available from: 10.3934/geosci.2022018.

Braun, V. and Clarke, V., 2006. Using thematic analysis in psychology. *Qualitative research in psychology*, 3 (2), 77–101.

De Brauwer, M., Harvey, E. S., McIlwain, J. L., Hobbs, J.-P. A., Jompa, J., and Burton, M., 2017. The economic contribution of the muck dive industry to tourism in Southeast Asia. *Marine Policy*, 83, 92–99.

Brown, K., Eernstman, N., Huke, A. R., and Reding, N., 2017. The drama of resilience: learning, doing,

and sharing for sustainability. *Ecology and Society*, 22 (2).
Byczek, C., 2011. Blessings for All? Community-Based Ecotourism in Bali Between Global, National, and Local Interests–A Case Study. *Austrian Journal of South-East Asian Studies*, 4 (1), 81–106.
Charmaz, K., 2006. *Constructing grounded theory: A practical guide through qualitative analysis*. sage.
Cherry, K., 2018. Attitudes and behavior in psychology. *Very Well Mind*.
Choi, W.-S. and Lee, S.-B., 2012. The effect of servicescape of an eco-friendly restaurant on customer perceived value, attitude and behavior intention. *Culinary science and hospitality research*, 18 (5), 45–62.
Chong, K. L., 2020. The side effects of mass tourism: the voices of Bali islanders. *Asia Pacific Journal of Tourism Research*, 25 (2), 157–169.
Clukey, K. E., Lepczyk, C. A., Balazs, G. H., Work, T. M., Li, Q. X., Bachman, M. J., and Lynch, J. M., 2018. Persistent organic pollutants in fat of three species of Pacific pelagic longline caught sea turtles: Accumulation in relation to ingested plastic marine debris. *Science of the Total Environment*, 610, 402–411.
Cohen, E., 1982. Marginal paradises: Bungalow tourism on the islands of Southern Thailand. *Annals of Tourism Research*, 9 (2), 189–228.
Cornwall, C. E., Comeau, S., Kornder, N. A., Perry, C. T., van Hooidonk, R., DeCarlo, T. M., Pratchett, M. S., Anderson, K. D., Browne, N., and Carpenter, R., 2021. Global declines in coral reef calcium carbonate production under ocean acidification and warming. *Proceedings of the National Academy of Sciences*, 118 (21).
Costanza, R., De Groot, R., Sutton, P., Van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S., and Turner, R. K., 2014. Changes in the global value of ecosystem services. *Global environmental change*, 26, 152–158.
Creswell, J. W. and Creswell, J. D., 2017. *Research design: Qualitative, quantitative, and mixed methods approaches*. Sage publications.
Díaz-Pérez, L., Rodríguez-Zaragoza, F. A., Ortiz, M., Cupul-Magaña, A. L., Carriquiry, J. D., Ríos-Jara, E., Rodríguez-Troncoso, A. P., and García-Rivas, M. del C., 2016. Coral reef health indices versus the biological, ecological and functional diversity of fish and coral assemblages in the Caribbean Sea. *PloS one*, 11 (8), e0161812.
Diedrich, A., 2007. The impacts of tourism on coral reef conservation awareness and support in coastal communities in Belize. *Coral Reefs*, 26 (4), 985–996.
Diedrich, A., Stoeckl, N., Gurney, G. G., Esparon, M., and Pollnac, R., 2017. Social capital as a key determinant of perceived benefits of community-based marine protected areas. *Conservation Biology*, 31 (2), 311–321.
Doherty, O., Milner, C., Dustan, P., Campbell, S., Pardede, S., Kartawijaya, T., and Alling, A., 2013. Report on Menjangan Island's coral reef: A Bali Barat National Park marine protected area. *Atoll Research Bulletin*, 19 (599), 1–18.
Dunlap, R. E. and Van Liere, K. D., 1978. The "new environmental paradigm". *The journal of environmental education*, 9 (4), 10–19.
Dworkin, S. L., 2012. Sample size policy for qualitative studies using in-depth interviews.
Ertz, M. and Sarigöllü, E., 2019. The behavior-attitude relationship and satisfaction in proenvironmental behavior. *Environment and Behavior*, 51 (9–10), 1106–1132.
Fabinyi, M., 2010. The intensification of fishing and the rise of tourism: competing coastal livelihoods in the Calamianes Islands, Philippines. *Human Ecology*, 38 (3), 415–427.
Frey, J. and Berkes, F., 2014. Can partnerships and community-based conservation reverse the decline of coral reef social-ecological systems? *International Journal of the Commons* [online], 8 (1). Available from: http://10.0.71.176/ijc.408.
Garcia, B., Fang, M. M., and Lin, J., 2019. Marine plastic pollution in Asia: All hands on deck! *Chinese Journal of Environmental Law*, 3 (1), 11–46.
Garrod, B., Wilson, J. C., and Bruce, D. M., 2003. Defining marine ecotourism: a Delphi study. *Marine*

Ecotourism: Issues and Experiences. Channel View, Clevedon, Australia, 17–36.
Gelcich, S., Godoy, N., and Castilla, J. C., 2009. Artisanal fishers' perceptions regarding coastal co-management policies in Chile and their potentials to scale-up marine biodiversity conservation. *Ocean & Coastal Management*, 52 (8), 424–432.
Germanov, E. S., Marshall, A. D., Hendrawan, I. G., Admiraal, R., Rohner, C. A., Argeswara, J., Wulandari, R., Himawan, M. R., and Loneragan, N. R., 2019. Microplastics on the menu: plastics pollute indonesian manta ray and whale shark feeding grounds. *Frontiers in Marine Science*, 6, 679.
Gifford, R. and Sussman, R., 2012. Environmental attitudes.
Glaser, M., Baitoningsih, W., Ferse, S. C. A., Neil, M., and Deswandi, R., 2010. Whose sustainability? Top–down participation and emergent rules in marine protected area management in Indonesia. *Marine Policy*, 34 (6), 1215–1225.
Goñi, R., Badalamenti, F., and Tupper, M. H., 2011. Fisheries–Effects of marine protected areas on local fisheries: evidence from empirical studies. *Marine protected areas: A multidisciplinary approach*, 72, 73.
Greenbaum, T. L., 1998. *The handbook for focus group research*. Sage.
Grimm, K. E. and Needham, M. D., 2012. Moving beyond the "I" in motivation: Attributes and perceptions of conservation volunteer tourists. *Journal of Travel Research*, 51 (4), 488–501.
Grúňová, M., Brandlová, K., Svitálek, J., and Hejcmanová, P., 2017. Environmental education supports conservation action by increasing the immediate and long-term environmental knowledge of children in West Africa. *Applied Environmental Education & Communication*, 16 (1), 3–16.
Gurney, G. G., Cinner, J., Ban, N. C., Pressey, R. L., Pollnac, R., Campbell, S. J., Tasidjawa, S., and Setiawan, F., 2014. Poverty and protected areas: an evaluation of a marine integrated conservation and development project in Indonesia. *Global Environmental Change*, 26, 98–107.
Hein, M. Y., Vardi, T., Shaver, E. C., Pioch, S., Boström-Einarsson, L., Ahmed, M., Grimsditch, G., and McLeod, I. M., 2021. Perspectives on the use of coral reef restoration as a strategy to support and improve reef ecosystem services. *Frontiers in Marine Science*, 299.
Heron, S. F., Eakin, C. M., Douvere, F., Anderson, K. L., Day, J. C., Geiger, E., Hoegh-Guldberg, O., Van Hooidonk, R., Hughes, T., and Marshall, P., 2017. Impacts of climate change on World Heritage coral reefs: a first global scientific assessment.
Hofman, K., Hughes, K., and Walters, G., 2020. Effective conservation behaviours for protecting marine environments: The views of the experts. *Journal of Sustainable Tourism*, 28 (10), 1460–1478.
Howlett, L., Camp, E. F., Edmondson, J., Edmondson, J., Agius, T., Hosp, R., Coulthard, P., Edmondson, S., and Suggett, D. J., 2022. Adoption of coral propagation and out-planting via the tourism industry to advance site stewardship on the northern Great Barrier Reef. *Ocean & Coastal Management*, 225, 106199.
Hughes, T. P., Baird, A. H., Morrison, T. H., and Torda, G., 2023. Principles for coral reef restoration in the anthropocene. *One Earth*.
Hughes, T. P., Kerry, J. T., Álvarez-Noriega, M., Álvarez-Romero, J. G., Anderson, K. D., Baird, A. H., Babcock, R. C., Beger, M., Bellwood, D. R., and Berkelmans, R., 2017. Global warming and recurrent mass bleaching of corals. *Nature*, 543 (7645), 373–377.
Hughes, T. P., Kerry, J. T., Baird, A. H., Connolly, S. R., Dietzel, A., Eakin, C. M., Heron, S. F., Hoey, A. S., Hoogenboom, M. O., and Liu, G., 2018. Global warming transforms coral reef assemblages. *Nature*, 556 (7702), 492–496.
Hungerford, H. R. and Volk, T. L., 1990. Changing learner behavior through environmental education. *The journal of environmental education*, 21 (3), 8–21.
Ilie, G., 2014. Positive versus negative effects of foreign direct investments on host countries. *Knowledge Horizons. Economics*, 6 (4), 162.
IPCC, 2021. *The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment*

Report of the Intergovernmental Panel on Climate Change.
Kellmereit, B., 2015. Focus groups. *International Journal of Sales, Retailing & Marketing*, 4 (9), 42–52.
Khamdevi, M. and Bott, H., 2018. Rethinking tourism: Bali's failure. *In*: *IOP Conference Series: Earth and Environmental Science*. IOP Publishing, 12171.
Kim, M. J., Hall, C. M., and Kim, D.-K., 2020. Predicting environmentally friendly eating out behavior by value-attitude-behavior theory: does being vegetarian reduce food waste? *Journal of Sustainable Tourism*, 28 (6), 797–815.
Kostić, M. and Petrović, M., 2013. The importance of biodiversity conservation as a factor of ecotourism development at the Goc mountain. *Hotel and Tourism Management*, 1 (1), 56–66.
Kruglov, L. P. and Davidson, H. H., 1953. The willingness to be interviewed: a selective factor in sampling. *The Journal of Social Psychology*, 38 (1), 39–47.
Kumar Chaudhary, A., Warner, L. A., Lamm, A. J., Israel, G. D., Rumble, J. N., and Cantrell, R. A., 2017. Using the Theory of Planned Behavior to Encourage Water Conservation among Extension Clients. *Journal of Agricultural Education*, 58 (3), 185–202.
Kuper, A., Lingard, L., and Levinson, W., 2008. Critically appraising qualitative research. *Bmj*, 337.
Kusumawati, I. and Huang, H.-W., 2015. Key factors for successful management of marine protected areas: A comparison of stakeholders' perception of two MPAs in Weh island, Sabang, Aceh, Indonesia. *Marine Policy*, 51, 465–475.
Lapointe, B. E., Brewton, R. A., Herren, L. W., Porter, J. W., and Hu, C., 2019. Nitrogen enrichment, altered stoichiometry, and coral reef decline at Looe Key, Florida Keys, USA: a 3-decade study. *Marine Biology*, 166 (8), 1–31.
Lavelle, M. J., Rau, H., and Fahy, F., 2015. Different shades of green? Unpacking habitual and occasional pro-environmental behavior. *Global Environmental Change*, 35, 368–378.
Legare, C. H., Akhauri, S., Chaudhuri, I., Hashmi, F. A., Johnson, T., Little, E. E., Lunkenheimer, H. G., Mandelbaum, A., Mandlik, H., and Mondal, S., 2020. Perinatal risk and the cultural ecology of health in Bihar, India. *Philosophical Transactions of the Royal Society B*, 375 (1805), 20190433.
Leisher, C., Mangubhai, S., Hess, S., Widodo, H., Soekirman, T., Tjoe, S., Wawiyai, S., Larsen, S. N., Rumetna, L., and Halim, A., 2012. Measuring the benefits and costs of community education and outreach in marine protected areas. *Marine Policy*, 36 (5), 1005–1011.
Lenihan, H. S., Gallagher, J. P., Peters, J. R., Stier, A. C., Hofmeister, J. K. K., and Reed, D. C., 2021. Evidence that spillover from Marine Protected Areas benefits the spiny lobster (Panulirus interruptus) fishery in southern California. *Scientific Reports*, 11 (1), 1–9.
Lestari, P. and Trihadiningrum, Y., 2019. The impact of improper solid waste management to plastic pollution in Indonesian coast and marine environment. *Marine pollution bulletin*, 149, 110505.
Littledyke, M., 2008. Science education for environmental awareness: approaches to integrating cognitive and affective domains. *Environmental education research*, 14 (1), 1–17.
Liu, J., Ouyang, Z., and Miao, H., 2010. Environmental attitudes of stakeholders and their perceptions regarding protected area-community conflicts: A case study in China. *Journal of environmental management*, 91 (11), 2254–2262.
Di Lorenzo, M., Guidetti, P., Di Franco, A., Calò, A., and Claudet, J., 2020. Assessing spillover from marine protected areas and its drivers: A meta-analytical approach. *Fish and Fisheries*, 21 (5), 906–915.
Lundquist, C. J. and Granek, E. F., 2005. Strategies for successful marine conservation: integrating socioeconomic, political, and scientific factors. *Conservation Biology*, 19 (6), 1771–1778.
Mahyuni, L. P., 2016. Corporate social responsibility of hotels in Bali–Indonesia: practices, drivers and barriers.
Mainzer, S. and Luloff, A. E., 2017. Informing environmental problems through field analysis: Toward a community landscape theory of pro-environmental behavior. *Community Development*, 48 (4), 483–498.
Maki, A., Carrico, A. R., Raimi, K. T., Truelove, H. B., Araujo, B., and Yeung, K. L., 2019. Meta-analysis

of pro-environmental behaviour spillover. *Nature Sustainability*, 2 (4), 307–315.
Mangubhai, S., Sykes, H., Manley, M., Vukikomoala, K., and Beattie, M., 2020. Contributions of tourism-based Marine Conservation Agreements to natural resource management in Fiji. *Ecological Economics*, 171, 106607.
Marshall, B., Cardon, P., Poddar, A., and Fontenot, R., 2013. Does sample size matter in qualitative research?: A review of qualitative interviews in IS research. *Journal of computer information systems*, 54 (1), 11–22.
Mason, M., 2010. Sample size and saturation in PhD studies using qualitative interviews. *In*: *Forum qualitative Sozialforschung/Forum: qualitative social research*.
McGehee, N. G. and Andereck, K., 2008. 'Pettin'the critters': exploring the complex relationship between volunteers and the voluntoured in McDowell County, West Virginia, USA, and Tijuana, Mexico. *In*: *Journeys of discovery in volunteer tourism: International case study perspectives*. Cabi Wallingford UK, 12–24.
McKinley, E. and Ballinger, R. C., 2018. Welsh legislation in a new era: A stakeholder perspective for coastal management. *Marine Policy*, 97, 253–261.
McLeod, E. and Palmer, M., 2015. Why conservation needs religion. *Coastal Management*, 43 (3), 238–252.
McLeod, E., Szuster, B., and Salm, R., 2009. Sasi and marine conservation in Raja Ampat, Indonesia. *Coastal Management*, 37 (6), 656–676.
Milfont, T. L., 2007. Psychology of environmental attitudes. *Unpublished doctoral thesis, The University of Auckland, Auckland, NZ*.
Monbiot, G., 2017. *Out of the wreckage: A new politics for an age of crisis*. Verso Books.
Moran, E. F., 2016. *People and nature: An introduction to human ecological relations*. John Wiley & Sons.
Morse, J. M., 2000. Determining sample size.
Moura, C. O. de, Silva, Í. R., Silva, T. P. da, Santos, K. A., Crespo, M. da C. A., and Silva, M. M. da, 2021. Methodological path to reach the degree of saturation in qualitative research: grounded theory. *Revista Brasileira de Enfermagem*, 75.
Naiman, S. M., Stedman, R. C., and Schuldt, J. P., 2023. Latine culture and the environment: How familism and collectivism predict environmental attitudes and behavioral intentions among US Latines. *Journal of Environmental Psychology*, 85, 101902.
Nordfjærn, T., Şimşekoğlu, Ö., and Rundmo, T., 2014. The role of deliberate planning, car habit and resistance to change in public transportation mode use. *Transportation Research Part F: Traffic Psychology and Behaviour*, 27, 90–98.
O. Nyumba, T., Wilson, K., Derrick, C. J., and Mukherjee, N., 2018. The use of focus group discussion methodology: Insights from two decades of application in conservation. *Methods in Ecology and evolution*, 9 (1), 20–32.
Pandolfi, J. M., Connolly, S. R., Marshall, D. J., and Cohen, A. L., 2011. Projecting coral reef futures under global warming and ocean acidification. *science*, 333 (6041), 418–422.
Parker, L. and Prabawa-Sear, K., 2019. *Environmental Education in Indonesia: Creating Responsible Citizens in the Global South?* Routledge.
Parsons, E. C. M., MacPherson, R., and Villagomez, A., 2017. Marine "conservation": you keep using that word but I don't think it means what you think it means. *Frontiers in Marine Science*, 4, 299.
Patton, M. Q., 2014. *Qualitative research & evaluation methods: Integrating theory and practice*. Sage publications.
Pedju, F. M. K., 2018. Stakeholders' perceptions of the impacts of tourism on the social and ecological resilience of marine protected areas in Bali, Indonesia.
Prasetia, D., Supriharyono, M., Anggoro, S., and Sya'Rani, L., 2017. Coral Bleaching on Lembongan Island, Nusa Penida, Bali. *In*: . Atlantis Press.
Principe, P., Bradley, P., Yee, S., Fisher, W., Johnson, E., Allen, P., and Campbell, D., 2012. Quantifying

coral reef ecosystem services. US Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC.
Puspasari, R., Wiadnyana, N. N., Hartati, S. T., Rachmawati, R., and Yahya, Y., 2020. The Effectiveness of Artificial Reefs in Improving Ecosystem Health to Increase Coral Reef Resilience. *Jurnal Segara*, 16 (2), 115–126.
Ramos, J., Lino, P. G., Himes-Cornell, A., and Santos, M. N., 2019. Local fishermen's perceptions of the usefulness of artificial reef ecosystem services in Portugal. *PeerJ*, 6, e6206.
Rizzi, F., Annunziata, E., Contini, M., and Frey, M., 2020. On the effect of exposure to information and self-benefit appeals on consumer's intention to perform pro-environmental behaviours: A focus on energy conservation behaviours. *Journal of Cleaner Production*, 270, 122039.
Rokicka, E., 2002. Attitudes toward natural environment: A study of local community dwellers. *International Journal of Sociology*, 32 (3), 78–90.
Saldaña, J., 2021. *The coding manual for qualitative researchers*. sage.
dos Santos, M. N., Monteiro, C. C., and Lasserre, G., 2005. Observations and trends on the intra-annual variation of the fish assemblages on two artificial reefs in Algarve coastal waters (southern Portugal). *Scientia Marina*, 69 (3), 415–426.
Santos, M. N. and Monteiro, C. C., 1997. The Olhao artificial reef system (south Portugal): fish assemblages and fishing yield. *Fisheries Research*, 30 (1–2), 33–41.
Schwartz, S. H., 1977. Normative influences on altruism. *In*: *Advances in experimental social psychology*. Elsevier, 221–279.
Schwartz, S. H., 1992. Universals in the content and structure of values: Theoretical advances and empirical tests in 20 countries. *In*: *Advances in experimental social psychology*. Elsevier, 1–65.
Schwartz, S. H., 2012. An overview of the Schwartz theory of basic values. *Online readings in Psychology and Culture*, 2 (1), 919–2307.
SER, 2004. Society for ecological restoration international science & policy working group.
Shani, A., Polak, O., and Shashar, N., 2012. Artificial reefs and mass marine ecotourism. *Tourism Geographies*, 14 (3), 361–382.
Sigit, D. V, Miarsyah, M., Komala, R., Suryanda, A., Fadrikal, R., and Ichsan, I. Z., 2019. Improvement of knowledge and attitude in conservation of mangrove and coral reefs through environmental education community network model. *In*: . IOP Publishing, 12201.
Stafford, R. and Jones, P. J. S., 2019a. Viewpoint–Ocean plastic pollution: A convenient but distracting truth? *Marine policy*, 103, 187–191.
Stafford, R. and Jones, P. J. S., 2019b. We should not separate out environmental issues, but the current approach to plastic pollution can be a distraction from meaningful action. A response to Avery-Gomm et al. *Marine Policy*, 107, 103585.
Steg, L. E. and De Groot, J. I. M., 2018. *Environmental psychology: An introduction.* Wiley.
Steg, L. and Vlek, C., 2009. Encouraging pro-environmental behaviour: An integrative review and research agenda. *Journal of environmental psychology*, 29 (3), 309–317.
Stem, C. J., Lassoie, J. P., Lee, D. R., and Deshler, D. J., 2003. How'eco'is ecotourism? A comparative case study of ecotourism in Costa Rica. *Journal of sustainable tourism*, 11 (4), 322–347.
Stern, P. C., Dietz, T., Abel, T., Guagnano, G. A., and Kalof, L., 1999. A value-belief-norm theory of support for social movements: The case of environmentalism. *Human ecology review*, 81–97.
Strieder Philippssen, J., Soares Angeoletto, F. H., and Santana, R. G., 2017. Education level and income are important for good environmental awareness: a case study from south Brazil. *Ecología austral*, 27 (1), 39–44.
Sujarwoto, S., Tampubolon, G., and Pierewan, A. C., 2018. Individual and contextual factors of happiness and life satisfaction in a low middle income country. *Applied Research in Quality of Life*, 13 (4), 927–945.
Sully, S., Burkepile, D. E., Donovan, M. K., Hodgson, G., and Van Woesik, R., 2019. A global analysis of coral bleaching over the past two decades. *Nature communications*, 10 (1), 1–5.
Suparno, N., Currier, K., Milner, C., Alling, A., and Dustan, P., 2019. Ecological changes in the coral

reef communities of Indonesia's Bali Barat National Park, 2011–2016. *Atoll Res Bull* [online], 620, 1–35. Available from: http://10.0.21.103/si.0077-5630.620.

Susiloningtyas, D., Handayani, T., and Amalia, A. N., 2018. The impact of coral reefs destruction and climate change in Nusa Dua and Nusa Penida, Bali, Indonesia. *In*: . IOP Publishing, 12054.

Suteja, Y., Atmadipoera, A. S., Riani, E., Nurjaya, I. W., Nugroho, D., and Cordova, M. R., 2021. Spatial and temporal distribution of microplastic in surface water of tropical estuary: Case study in Benoa Bay, Bali, Indonesia. *Marine pollution bulletin*, 163, 111979.

Syam, A. R., Edrus, I. N., and Hartati, S. T., 2017. Coral fish population changes in the surrounding artificial reefs of the Lebah coastal waters, Karangasem, Bali. *Indonesian Fisheries Research Journal*, 13 (2), 101–116.

Taplin, J., Dredge, D., and Scherrer, P., 2014. Monitoring and evaluating volunteer tourism: A review and analytical framework. *Journal of Sustainable Tourism*, 22 (6), 874–897.

Terrier, L. and Marfaing, B., 2015. Using social norms and commitment to promote pro-environmental behavior among hotel guests. *Journal of Environmental Psychology*, 44, 10–15.

Tito, C. K., Ampou, E. E., and Wibawa, T. A., 2019. Stressor-Response of Reef-Building Corals to Climate Change in the Menjangan Island, West Bali National Park, Indonesia. *In*: . IOP Publishing, 12011.

Tomomi, I., 2010. Ecotourism in Bali: Backgrounds, present conditions and challenges. *Unpublished doctoral thesis). Ritsumeikan University, Kyoto, Japan*.

Trialfhianty, T. I., 2017. The role of the community in supporting coral reef restoration in Pemuteran, Bali, Indonesia. *Journal of Coastal Conservation*, 21 (6), 873–882.

Ulhasanah, N. and Goto, N., 2018. Assessment of citizens' environmental behavior toward municipal solid waste management for a better and appropriate system in Indonesia: a case study of Padang City. *Journal of Material Cycles and Waste Management*, 20 (2), 1257–1272.

Varela-Candamio, L., Novo-Corti, I., and García-Álvarez, M. T., 2018. The importance of environmental education in the determinants of green behavior: A meta-analysis approach. *Journal of cleaner production*, 170, 1565–1578.

Waayers, D., Lee, D., and Newsome, D., 2012. Exploring the nature of stakeholder collaboration: A case study of marine turtle tourism in the Ningaloo region, Western Australia. *Current Issues in Tourism*, 15 (7), 673–692.

Wager, E. and Williams, P., 2013. "Hardly worth the effort"? Medical journals' policies and their editors' and publishers' views on trial registration and publication bias: quantitative and qualitative study. *Bmj*, 347.

Wardana, A., 2019. *Contemporary Bali: Contested Space and Governance*. Palgrave Macmillan.

Wicaksana, I. B. A., 2020. 2020 Bali's Coral Conservation: from pandemic challenge to Government coral garden project. *Bali Tourism Journal*, 4 (2), 35–39.

Williams, K. C. and Page, R. A., 2011. Marketing to the generations. *Journal of behavioral studies in business*, 3 (1), 37–53.

Woo, E. and Kim, Y. G., 2019. Consumer attitudes and buying behavior for green food products: From the aspect of green perceived value (GPV). *British Food Journal*.

Woodhead, A. J., Hicks, C. C., Norström, A. V, Williams, G. J., and Graham, N. A. J., 2019. Coral reef ecosystem services in the Anthropocene. *Functional Ecology*, 33 (6), 1023–1034.

Yunitawati, D. and Clifton, J., 2019. Governance in the early stages of marine protected area development: A case study of Nusa Penida District Marine Conservation Area, Indonesia. *Marine Policy*, 103653.

Zhao, Q., Stephenson, F., Lundquist, C., Kaschner, K., Jayathilake, D., and Costello, M. J., 2020. Where Marine Protected Areas would best represent 30% of ocean biodiversity. *Biological Conservation*, 244, 108536.

Chapter 7: Discussion

Summary of thesis aims and key findings

The overall aim of this PhD thesis has been to investigate if coral reef conservation projects (specifically those that deploy artificial reefs) in Bali have restored ecological communities, ecosystem processes, and generated localised societal benefits (such as ecotourism and pro-environmental behaviours). Broadly, I found that coral reef conservation programmes in Bali have been effective in capturing some of the benefits of natural coral reefs, although they were not yet shown to act as a direct replacement for natural reefs. More specifically, chapters 2 and 3 showed that fish communities on artificial reefs became similar (ecologically equivalent) to those on a nearby coral reef over a 3 year period, whilst benthic populations remained quite different. Chapter 4 showed that artificial reefs displayed early signs of functioning as natural coral reefs, but did not directly resemble natural coral reefs in terms of the dynamics and storage of some key nutrients, likely because of the differences between benthic communities. Chapters 5 and 6 showed coral reef restoration programmes had generated localised social benefits through perceived additional marine tourism, and had also promoted the generation of pro-environmental behaviours within the community, through education and empowerment by local leaders. This discussion draws comparisons and links between these key benefits of coral reef restoration (as highlighted by my thesis), and makes suggestions on how these benefits can be maximised through holistic reef management measures.

Ecosystem services associated with coral reef restoration

A coral reef restoration programme can provide multiple ecosystem services, including provisioning services (e.g. subsistence and commercial fisheries, medicines, ornamental resources and building materials), regulating services (e.g. erosion control, climate regulation and storm prevention), cultural services (e.g. tourism, education, recreation, social traditions) and supporting services (e.g. biogeochemical cycling; Burke et al. 2008; Laurans et al. 2013). Although it was found that coral reef conservation programmes in Bali have been effective in capturing some of the ecosystem of natural coral reefs, it was also clear that in each location studied, there were conflicts between the natural and social benefits provided by coral reef restoration programmes (and their subsequent restored reefs). As documented by relevant literature, generally the social aspect of environmental conservation has been shown to weaken the natural aspect of it, and vice versa (Tomićević et al. 2010; Rahman et al. 2017; Wu et al. 2020).

My research found that reef restoration programmes in Bali were associated with the provisioning of multiple ecosystem services, with examples including:
1) Increasing the abundance and biodiversity of local fish stocks (as highlighted through chapters 2 and 3), which can lead to higher fishing yields for fishers who directly fish on restored reefs. If the restored reef sits within a marine protected area which prohibit fishing (such as most of the sites in this research), fishers may still benefit from the 'spill over effect' by fishing outside of it (as demonstrated by Abesamis and Russ (2005) and Lenihan et al. (2021)).
2) Biogeochemical cycling and carbon storage (chapter 4); the rapid circulation of key nutrients (nitrates, nitrites, ammonium and phosphates) as well as storage of carbon by the microbial, benthic and mobile communities associated with coral reefs.
3) Tourism (chapters 5 and 6); coral reef restoration was perceived to have led to additional tourists visits to a given area, which can create local new jobs in the reef-based tourism sector.
4) Education (chapter 6); coral reef restoration programmes often increase education and knowledge (related to protecting the marine environment) amongst the communities where they are based, which can lead to improvements in environmental attitudes and increase environmental behaviours.

Conflicts between the ecosystem services that coral reefs provide

There were found to be conflicts between the ecosystem services associated with coral reef restoration in Bali. These were generally split between the ecological and socioeconomic services, which were shown to be conflicting in situations which exploited the socioeconomic benefits associated with coral reefs. For example, most of the restoration sites within my PhD were located within a no-take-zone MPA, which likely provided important contributions to the demonstrated successes (in terms of ecological and biogeochemical benefits). There is a large body of literature highlighting that MPAs, especially those with no-take zone designation, are associated with benefits to ecology (in terms of enhancing fish stocks and protecting biodiversity; Costello 2014; Edgar et al. 2014; Zhao et al. 2020; Marcos et al. 2021), as well as biogeochemical cycling and carbon sequestration (Potts et al. 2014; Jacquemont et al. 2022). However, the establishment of MPAs may cause tension and opposition from local stakeholders, notably fishers, who sometimes argue that MPAs result in lower fishing (West et al. 2006; Mora and Sale 2011; Stafford 2018). Indonesia is one of the most fish dependent countries in the world (Teh et al. 2016) with coral reefs estimated to supply around 30% of its total fish catch (Kramer et al. 2002). Aside from food provision, artisanal fisheries also represent one of the primary livelihoods for Indonesian coastal communities (Halim et al. 2020). The establishment of fishing regulations therefore cause notable concerns for fishers who work within the industry, especially if they prohibit fishing in a previously productive fishing area (Goñi et al. 2011). This concern was documented by chapter 6 in a quote from FG1: "*We hope we can still do our job as fishers. If we are marginalised, I think we will fight*".

Managing fisheries within areas of conservation interest continues to be a global governance challenge (Stafford 2018), in large part because the greatest coral reef conservation successes are achieved in the absence of fishing (Agardy 1994; Maestro et al. 2019). This is greatly due to the important role of fish in coral reef functioning (Cole et al. 2008; Brandl et al. 2019; Boakes et al. 2023a). Whilst there is no one-size-fits-all solution to this issue, literature has highlighted that involving local stakeholder as participants within establishing MPA frameworks has been associated with greater acceptance and (Crawford et al. 2004; Taylor et al. 2013; Islam et al. 2017). Studies have also highlighted that well managed MPAs can lead to net-positive outcomes for fisheries, whereby the increased fish biomass from inside the protected area will 'spill over' to outside of it, thus increasing total fish catch over time, as documented by Roberts et al. (2001), Russ et al. (2003) and Abesamis and Russ (2005). As discussed in chapter 6, most fishers described their fishing yield as being 'stable' (no noticeable increase or decrease) since the restoration work and the no take MPA was established, likely because the species they target are often caught far from the protected area. Despite this, it is important to note that 'no change' in fishing yields may actually be positive sign, as it means that coral reef restoration and MPAs have been established without noticeable negative impacts for local fishers, in a 'win-win' situation for both the ecology and economy.

Another potential conflict between the ecosystem services associated with coral reef restoration was the issue of unsustainable reef tourism, whereby the socioeconomic benefits of tourism (notably job creation) led to the exploitation of coral reefs. Again, in cases where tourism negatively affects reefs, this would result in ecological consequences, and would therefore reduce the provisioning of other associated ecosystem services. This was discussed by chapter 5, which found both positive and negative examples of tourism, with regards to how it impacted reef health. The highlighted literature in chapter 5, as well as results from the research itself, showed that reef-based tourism has been associated with physical damage to corals (Davenport and Davenport 2006; Gladstone et al. 2013), as well as responsible for additional threats such as pollution (Diedrich 2007; Gladstone et al. 2013), sedimentation (Diedrich 2007) and disturbing reef biota (Rudd and Tupper

2002; Hayes et al. 2017; Giglio et al. 2019). Some of the study locations in chapter 5 (notably location 2; Kalibukbuk) were shown to have experienced similar threats to their reefs as a consequence of mass tourism, and stakeholders linked this with observable declines in reef health. In contrast, the research also highlighted that reef-based tourism (and its associated socio-economic benefits) can exist without negatively impacting reefs, and can actually in some cases (notably in location 1; Tianyar), provide substantial positive contributions towards restoring reefs that have previously been destroyed. Chapter 5 discussed that this 'win-win' situation can be achieved through the establishment of effective management measures, which facilitate a symbiotic relationship between ecology and economy.

Conclusion

My PhD aimed to evaluate the extent to which coral reef restoration programmes in Bali have restored reef ecology, ecosystem functioning and generated localised social benefits. I found that fish communities on artificial reefs became similar (ecologically equivalent) to those on a nearby CR over a 3 year period in terms of number of species, whilst benthic populations remained quite different. My research highlighted that artificial reefs displayed some resemblance to CRs in terms of functioning, but were not functioning fully as natural reefs. I also found coral reef restoration was perceived to have led to additional jobs in marine tourism, and had also promoted the generation of pro-environmental behaviours within the community, through education and empowerment of local leaders which create 'ocean empathy' amongst their people. Overall, the findings of my PhD found that coral reef conservation programmes in Bali had been effective in capturing some of the benefits of natural CRs, although they were not yet shown to act as a direct replacement for the natural reefs which they may aim to replace.

It is clear that conflicts can exist between the ecosystem services derived from coral reef restoration. If the socio-economic benefits are exploited too much, there will be consequences for the ecological and biogeochemical benefits associated with coral reefs. However, the implementation of effective management measures, which aim to holistically protect all ecosystem services associated with coral reefs, can facilitate a 'win-win' situation for coral reef ecology and economy, provided that short-term economic benefits are not fully exploited. A mutually beneficial situation such as this is likely to be supported by most local stakeholders, and therefore has a far greater potential for successful ecological restoration than one with poor community engagement.

From researching this topic over the past three years, I have come to learn that the field of coral reef ecosystem functioning science is still very much in its infancy. More specifically, there are large gaps in the literature in terms of our understanding the flora and fauna that provide important contributions to the functioning of coral reef ecosystems. Given that functioning is the biophysical foundation of the ecosystem services provided by coral reefs, improving our understanding of these key components and processes, will allow us to better conserve coral reefs as they face increasing anthropogenic threats over the coming decades.

References

Abesamis, R. A. and Russ, G. R., 2005. Density-dependent spillover from a marine reserve: long-term evidence. *Ecological applications*, 15 (5), 1798–1812.

Agardy, M. T., 1994. Advances in marine conservation: the role of marine protected areas. *Trends in ecology & evolution*, 9 (7), 267–270.

Boakes, Z., Suryaputra, I. G. N. A., Hall, A. E., Franklin, D. J. and Stafford, R., 2023. Nutrient dynamics, carbon storage and community composition on artificial and natural reefs in Bali, Indonesia. *Marine Biology*, 170 (10), 130.

Brandl, S. J., Tornabene, L., Goatley, C. H. R., Casey, J. M., Morais, R. A., Côté, I. M., Baldwin, C. C., Parravicini, V., Schiettekatte, N. M. D. and Bellwood, D. R., 2019. Demographic dynamics of the smallest marine vertebrates fuel coral reef ecosystem functioning. *Science*, 364 (6446), 1189–1192.

Burke, L., Greenhalgh, S., Prager, D. and Cooper, E., 2008. Coastal capital: economic valuation of coral reefs in Tobago and St. Lucia. *Coastal capital: economic valuation of coral reefs in Tobago and St. Lucia.*

Cole, A. J., Pratchett, M. S. and Jones, G. P., 2008. Diversity and functional importance of coral-feeding fishes on tropical coral reefs. *Fish and Fisheries*, 9 (3), 286–307.

Costello, M. J., 2014. Long live Marine Reserves: A review of experiences and benefits. *Biological Conservation*, 176, 289–296.

Crawford, B. R., Siahainenia, A., Rotinsulu, C. and Sukmara, A., 2004. Compliance and enforcement of community-based coastal resource management regulations in North Sulawesi, Indonesia. *Coastal Management,* 32 (1), 39–50.

Davenport, J. and Davenport, J. L., 2006. The impact of tourism and personal leisure transport on coastal environments: a review. *Estuarine, coastal and shelf science*, 67 (1–2), 280–292.

Diedrich, A., 2007. The impacts of tourism on coral reef conservation awareness and support in coastal communities in Belize. *Coral Reefs*, 26 (4), 985–996.

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., Barrett, N. S., Becerro, M. A., Bernard, A. T. F. and Berkhout, J., 2014. Global conservation outcomes depend on marine protected areas with five key features. *Nature*, 506 (7487), 216–220.

Giglio, V. J., Ternes, M. L. F., Kassuga, A. D. and Ferreira, C. E. L., 2019. Scuba diving and sedentary fish watching: effects of photographer approach on seahorse behavior. *Journal of Ecotourism*, 18 (2), 142–151.

Gladstone, W., Curley, B. and Shokri, M. R., 2013. Environmental impacts of tourism in the Gulf and the Red Sea. *Marine pollution bulletin*, 72 (2), 375–388.

Goñi, R., Badalamenti, F. and Tupper, M. H., 2011. Fisheries–Effects of marine protected areas on local fisheries: evidence from empirical studies. *Marine protected areas: A multidisciplinary approach*, 72, 73.

Halim, A., Loneragan, N. R., Wiryawan, B., Fujita, R., Adhuri, D. S., Hordyk, A. R. and Sondita, M. F. A., 2020. Transforming traditional management into contemporary territorial-based fisheries management rights for small-scale fisheries in Indonesia. *Marine Policy*, 116, 103923.

Hayes, C. T., Baumbach, D. S., Juma, D. and Dunbar, S. G., 2017. Impacts of recreational diving on hawksbill sea turtle (Eretmochelys imbricata) behaviour in a marine protected area. *Journal of Sustainable Tourism*, 25 (1), 79–95.

Islam, G. M. N., Tai, S. Y., Kusairi, M. N., Ahmad, S., Aswani, F. M. N., Senan, M. K. A. M. and Ahmad, A., 2017. Community perspectives of governance for effective management of marine protected areas in Malaysia. *Ocean & Coastal Management*, 135, 34–42.

Jacquemont, J., Blasiak, R., Le Cam, C., Le Gouellec, M. and Claudet, J., 2022. Ocean conservation boosts climate change mitigation and adaptation. *One Earth*, 5 (10), 1126–1138.

Kramer, R. A., Simanjuntak, S. M. H. and Liese, C., 2002. Migration and fishing in Indonesian coastal villages. *AMBIO: A Journal of the Human Environment*, 31 (4), 367–372.

Laurans, Y., Pascal, N., Binet, T., Brander, L., Clua, E., David, G., Rojat, D. and Seidl, A., 2013. Economic valuation of ecosystem services from coral reefs in the South Pacific: Taking stock of recent experience. *Journal of environmental management*, 116, 135–144.

Lenihan, H. S., Gallagher, J. P., Peters, J. R., Stier, A. C., Hofmeister, J. K. K. and Reed, D. C., 2021. Evidence that spillover from Marine Protected Areas benefits the spiny lobster (Panulirus interruptus) fishery in southern California. *Scientific Reports*, 11 (1), 1–9.

Maestro, M., Pérez-Cayeiro, M. L., Chica-Ruiz, J. A. and Reyes, H., 2019. Marine protected areas in the 21st century: Current situation and trends. *Ocean & coastal management*, 171, 28–36.

Marcos, C., Díaz, D., Fietz, K., Forcada, A., Ford, A., García-Charton, J. A., Goñi, R., Lenfant, P., Mallol, S. and Mouillot, D., 2021. Reviewing the ecosystem services, societal goods, and benefits of marine protected areas. *Frontiers in Marine Science*, 8, 613819.

Mora, C. and Sale, P. F., 2011. Ongoing global biodiversity loss and the need to move beyond protected areas: a review of the technical and practical shortcomings of protected areas on land and sea. *Marine ecology progress series*, 434, 251–266.

Potts, T., Burdon, D., Jackson, E., Atkins, J., Saunders, J., Hastings, E. and Langmead, O., 2014. Do marine protected areas deliver flows of ecosystem services to support human welfare? *Marine Policy*, 44, 139–148.

Rahman, M. M., Mahmud, M. A. Al and Shahidullah, M., 2017. Socioeconomics of biodiversity conservation in the protected areas: a case study in Bangladesh. *International Journal of Sustainable Development & World Ecology*, 24 (1), 65–72.

Roberts, C. M., Bohnsack, J. A., Gell, F., Hawkins, J. P. and Goodridge, R., 2001. Effects of marine reserves on adjacent fisheries. *science*, 294 (5548), 1920–1923.

Rudd, M. A. and Tupper, M. H., 2002. The impact of Nassau grouper size and abundance on scuba diver site selection and MPA economics. *Coastal Management*, 30 (2), 133–151.

Russ, G. R., Alcala, A. C. and Maypa, A. P., 2003. Spillover from marine reserves: the case of Naso vlamingii at Apo Island, the Philippines. *Marine Ecology Progress Series*, 264, 15–20.

Stafford, R., 2018. Lack of evidence that governance structures provide real ecological benefits in marine protected areas. *Ocean & coastal management*, 152, 57–61.

Taylor, E., Baine, M., Killmer, A. and Howard, M., 2013. Seaflower marine protected area: Governance for sustainable development. *Marine Policy*, 41, 57–64.

Teh, L. S. L., Lam, V. W. Y., Cheung, W. W. L., Miller, D., Teh, L. C. L. and Sumaila, U. R., 2016. Impact of high seas closure on food security in low income fish dependent countries. *PloS one*, 11 (12), e0168529.

Tomićević, J., Shannon, M. A. and Milovanović, M., 2010. Socio-economic impacts on the attitudes towards conservation of natural resources: Case study from Serbia. *Forest policy and economics*, 12 (3), 157–162.

West, P., Igoe, J. and Brockington, D., 2006. Parks and peoples: the social impact of protected areas. *Annu. Rev. Anthropol.*, 35, 251–277.

Wu, J., Guo, Y. and Zhou, J., 2020. Nexus between ecological conservation and socio-economic development and its dynamics: Insights from a case in China. *Water*, 12 (3), 663.

Zhao, Q., Stephenson, F., Lundquist, C., Kaschner, K., Jayathilake, D. and Costello, M. J., 2020. Where Marine Protected Areas would best represent 30% of ocean biodiversity. *Biological Conservation*, 244, 108536.