

Interdisciplinary Approaches to Shark and Ray Conservation



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Abstract

Human actions are causing pervasive declines in biodiversity and ecosystem services. Sharks, rays and their cartilaginous relatives (Class Chondrichthyes, herein ‘sharks’) are amongst the most threatened taxa on earth, primarily due to overfishing. Oceanic shark populations have declined by more 70% in the past 50 years, and over one third of shark species are threatened with extinction. Technologies and practices that reduce impacts of fisheries on sharks are relatively well documented, yet little is known about how to incentivise adoption across different types of fisheries while also managing trade-offs between conservation objectives and the important socio-economic functions of fisheries. This thesis seeks to fill this knowledge gap, by exploring approaches which can simultaneously deliver conservation outcomes for sharks and well-being outcomes for people. I begin by critically assessing current approaches for understanding and managing threats to sharks, and highlight gaps relating to socio-economic issues. I then develop a general decision-making framework, based on the mitigation hierarchy, which can integrate socio-economic issues to support least-cost shark conservation. The remainder of the thesis then focuses on research methods and conservation approaches for understanding and addressing socio-economic implementation gaps in shark conservation. I take a dual approach, exploring: 1) fisher behaviour and behavioural interventions in small-scale fisheries (SSFs) in Indonesia, and 2) broader structural interventions, with a focus on market-based approaches. In the SSFs, I empirically investigate the socio-economic drivers of shark fishing, and barriers to pro-conservation behaviour; assess the impact of an existing intervention for manta ray conservation, and document lessons learned; and use predictive methods to explore the cost-effectiveness of hypothetical incentive-based interventions. I find that a range of micro- and macro-level drivers influence shark fishing mortality in SSFs, including: basic needs for food and livelihoods, socio-cultural values, incidental catches, and profit motivations. These drivers interact and vary across fisheries. I also find that incentive-based approaches could tackle these drivers, whilst also maintaining the well-being of coastal communities, provided they are well-designed and implemented as part of an intervention mix. In terms of structural interventions, I explore financing mechanisms for shark conservation, based on operationalising the beneficiary-pays and polluter-pays principles. I demonstrate that bycatch levies could offer a market-based approach for incentivising bycatch reduction in commercial fisheries, and generating revenue for compensatory conservation which could support delivery of no net loss or net gain for marine biodiversity. I also find that marine tourists who benefit from healthy shark populations are willing to pro-actively pay for shark conservation outcomes. Together, these two income streams could generate billions of dollars in ocean finance, and fund direct investments or performance-based payments for marine conservation in SSFs. I conclude by demonstrating how these behavioural and structural interventions could be drawn together as part of an integrated strategy, and offer some recommendations for future research and practice. Overall, my findings highlight the importance of a more holistic and interdisciplinary approach to shark conservation, and illustrate some feasible strategies and next steps for achieving better outcomes for sharks and people. If well implemented, these findings could contribute towards delivering the post-2020 global biodiversity strategy, the sustainable development goals, and a sustainable and equitable ocean economy.

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Declaration of originality

I declare that this thesis is entirely my own work. Contributions by other authors are stated in section 1.4.
None of the work has been submitted, in whole or in part, for any previous degree application.

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1. Introduction



Photo: A blacktip reef shark in a thriving coral reef ecosystem. Raja Ampat, Indonesia.

“Laut adalah rantai, hilang satu, hilang semu” (the ocean is a chain, lose one [link], lose all)
- fisher in Lhok Rigaih.

*“And I held the breath inside my lungs for days
And I saw myself as one of many waves
And when I knew I’d become the ocean’s slave
I just stayed”*
- Bahamas.

1.1. Problem statement

Ocean ecosystems, marine biodiversity and the services they provide to humanity, are threatened by overexploitation (Halpern et al., 2008; Worm et al., 2006). Large, long-lived marine animals ('marine megafauna') are especially threatened, with slow life history traits which make their populations more susceptible to overfishing (Dulvy, Fowler, et al., 2014; McClenachan et al., 2012; Musik, 1999). In particular, sharks, rays and their cartilaginous relatives (Class Chondrichthyes, herein 'sharks') are one of the world's most threatened vertebrate taxa, with population declines and extirpations well documented for coastal and oceanic species alike (Dulvy et al., 2021; MacNeil et al., 2020; Pacoureau et al., 2021).

There are several reasons to be concerned about losses of sharks from the oceans. As a biological and genetic resource, cartilaginous fishes are one of the world's most ancient and diverse taxa, embodying millions of years of unique evolutionary history (Stein et al., 2018). With over 1,100 described species, they have also evolved to fulfil a wide variety of ecological niches, which contribute to supporting and regulating ecosystem services. For example, sharks' high mobility and migratory behaviour can support nutrient transfer and cycling; and as apex and meso-predators, sharks can play important roles in regulating trophic cascades, and maintaining the condition, diversity and health of coral reef ecosystems (Barley et al., 2017; Ferretti et al., 2010; Hammerschlag et al., 2019; McKibben & Nelson, 1986; Roff et al., 2016). Sharks are also intrinsically linked to the livelihoods and identity of many communities around the world, with consumptive use of shark products providing sources of income, nutrition and socio-cultural value (Clarke, 2015; Glaus et al., 2018; Jaiteh, Loneragan, et al., 2017; Leeney & Poncelet, 2015). Healthy shark populations can also generate millions of dollars in marine tourism value, particularly via scuba-diving and snorkelling (Gallagher & Hammerschlag, 2011; Mustika et al., 2020; Vianna et al., 2018). Therefore, supporting the recovery of shark populations and preventing species extinctions is essential for maintaining a range of ecosystem services, values and constituents of well-being (Figure 1.1).

Sustaining healthy shark populations means tackling overfishing. Overfishing is the primary direct threat to sharks, which occurs via cumulative pressures from targeted and incidental fisheries throughout the world's oceans (Dulvy et al., 2021; Dulvy, Fowler, et al., 2014). Fishing pressure is driven partially by demand for shark-derived commodities (fins, meat, cartilage and liver oil), coupled with a general expansion of global fisheries, with high levels of bycatch mortality (Clarke et al., 2006; Dent & Clarke, 2014; James et al., 2016; Oliver et al., 2015). Since many shark species have conservative life-history traits, which make them intrinsically more vulnerable to overexploitation relative to most teleost fishes, these levels of fishing mortality exceed the average rebound potential of most shark populations (Cortés, 2000; Worm et al., 2013). For example, it is estimated that oceanic shark populations have declined by over 70% in the past 50 years, which is attributed to an 18-fold increase in relative fishing pressure (Pacoureau et al., 2021), and 37% of species are now at risk of extinction (Dulvy et al., 2021).

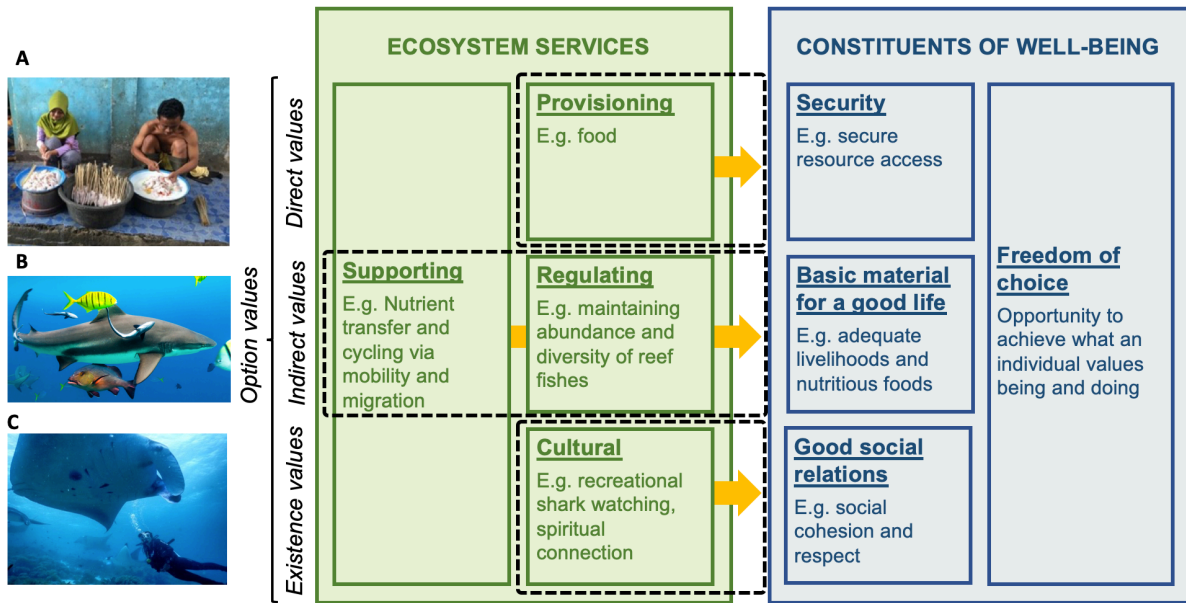


Figure 1.1 Examples of ecosystem services, values and constituents of well-being from sharks. A: shark meat skewers being prepared for sale (provisioning), B: a reef shark interacting with reef fish on a coral reef (maintaining), C: a scuba-diver with a manta ray (cultural)

Shark overfishing is exacerbated by a variety of socio-economic drivers, including local and international markets, governance complexities, lack of political will, and poverty (Dulvy et al., 2017; Glaus et al., 2018; Jaiteh, Loneragan, et al., 2017; MacNeil et al., 2020). Unlike other commercially important fish species, such as tuna, or charismatic marine megafauna with similar life histories and ecotourism potential, such as cetaceans, shark populations are exceptionally under-managed (Dulvy et al., 2017; Lack & Sant, 2011). Moreover, the diversity of shark species and the plurality of values associated with them creates conflicts and trade-offs with respect to how they should be managed: for some people, sharks are fisheries resources to be harvested for their consumptive use value; for others, they are a pest, to be controlled; for others, they are a conservation priority, to be protected and restored for their non-consumptive use and existence values (Carlson et al., 2019; Iwane et al., 2021; Molony & Thomson, 2020; Shiffman & Hammerschlag, 2016a). Therefore, like many conservation problems, tackling the root causes of shark overfishing requires addressing socio-economic issues (Balmford & Cowling, 2006; Díaz et al., 2019; D. R. Williams et al., 2020).

However, to date, much shark science has focused on describing the state of shark species and populations, and understanding the threatening mechanisms (i.e., overfishing), with relatively little research on understanding socio-economic drivers and trade-offs; and designing, implementing and testing responses: a pattern which is arguably reflected in conservation science literature in general (Simpfendorfer et al., 2011; D. R. Williams et al., 2020). This means that technologies and practices that reduce fisheries' impacts on sharks are relatively well documented (e.g., BMIS, 2021), yet less is known about how to encourage their adoption. Doing so is challenging, because it can create direct trade-offs

between conservation objectives and the important socio-economic functions of fisheries. These trade-offs are particularly acute in small-scale fisheries (SSFs), where endangered shark species can provide important sources of income and nutrition for ocean-dependent coastal communities (Glaus et al., 2018; Jaiteh, Loneragan, et al., 2017; Milner-Gulland, Ibbett, et al., 2020). However, mitigating shark catches is also costly in the commercial sector. For example, sharks may be difficult to avoid due to range overlaps with productive fishing grounds for high-value teleost fishes, or bycatch reduction technologies (BRTs) may be cumbersome to use or result in lost target catches (Campbell & Cornwell, 2008; M. A. Hall, 1996). These barriers, combined with valuable markets for shark-derived commodities (Hau et al., 2018; O'Malley et al., 2017; Wu, 2016), mean that fishers are rarely incentivised to mitigate mortality of sharks. These complexities and trade-offs create a triple challenge for ocean management, of providing food and nutrition for a growing population, and livelihoods for millions of coastal communities, while conserving marine biodiversity. Solving this challenge requires interdisciplinary and inclusive approaches, which can identify and address the underlying drivers of shark overfishing, and allow people and marine biodiversity to thrive together.

This is particularly important in the context of international policy objectives - such as those under the Sustainable Development Goals (SDGs) (e.g., Zero Hunger and Life Below Water), the UN Convention on Biological Diversity's (CBD's) post-2020 framework ("*Living in harmony with nature by 2050*"), and the Convention on the International Trade of Endangered Species (CITES) ("*to ensure that international trade in specimens of wild animals and plants does not threaten the survival of the species*") – which are also met with ambitious plans for 'blue growth', and calls for blue justice and a sustainable and equitable ocean economy (N. J. Bennett et al., 2019, 2021; CBD, 2020a; OECD, 2016). There are clear synergies and trade-offs within and between these ambitious goals (Fader et al., 2018; G. G. Singh et al., 2018), many of which are characterised by shark conservation. For example, blue growth aims to deliver economic growth through exploitation of marine resources, yet economic activities in the ocean also inherently jeopardize marine ecosystems, as exemplified by shark bycatch (Halpern et al., 2015; Nash et al., 2017). In this sense, sharks can serve as flagship species for improved ocean management across the globe, and, since shark conservation is arguably orders of magnitude more complex than that for other species of conservation concern (Dulvy et al., 2017), valuable lessons can surely be learned and applied to other species and ecosystems.

1.2. Study systems

The empirical work within this thesis focuses on Indonesia. Indonesia is a global priority for developing and testing interdisciplinary approaches to shark conservation for several reasons. Firstly, Indonesia lies at the heart of the Coral Triangle – the global epicentre of marine biodiversity, and a hotspot of shark diversity and endemism (Davidson & Dulvy, 2017; Selig et al., 2014). Secondly, Indonesia is the world's largest shark fishing nation, and a top exporter of shark-derived commodities (Dent & Clarke, 2014;

Dulvy et al., 2017). Third, Indonesia is characterised by high dependence on marine resources, particularly in coastal areas, with approximately 99% of fisheries classified as small-scale (Golden et al., 2016; Selig et al., 2018). Therefore, interdisciplinary approaches are needed to effectively and equitably address the ‘triple challenge’ of providing food, supporting livelihoods and protecting biodiversity within and from marine ecosystems. Within Indonesia, this thesis focuses on three case study small-scale fisheries, which represent contrasting case types in terms of fishery characteristics and socio-economic contexts (Figure 1.2). These fisheries are:

1. Lamakera in East Flores, East Nusa Tenggara Province: a coastal fishery, which historically targeted mobulid rays using harpoons (Lewis et al., 2015).
2. Tanjung Luar in Lombok, West Nusa Tenggara Province: an off-shore fishery, which specifically targets a range of pelagic and demersal sharks using longlines (I. Yulianto et al., 2018).
3. Lhok Rigaih in Aceh Jaya, Aceh Province: a coastal fishery, which targets a variety of reef and demersal fishes, and regularly takes endangered sharks as incidental catch (Simeon et al., 2020).



Figure 1.2 The study sites covered in this thesis. A. a map of Indonesia showing the location of the study sites; B-D examples of the vessels and catches in each site (B. Lhok Rigaih, C. Tanjung Luar, D. Lamakera).

All these fisheries are characterised by high dependence on marine resources, with the capture, consumption and sale of threatened shark species contributing to fishers' overall livelihood strategies and well-being. In all cases, the utilisation of sharks can be considered 'semi-commercial', in that some shark-derived commodities are consumed locally for subsistence purposes (usually meat), while other commodities are traded nationally or exported internationally (including excess meat, fins, morbid gill plates, skin, oil and cartilage) (Prasetyo et al., 2021). Despite these similarities, the study sites employ different fishing methods, which impact different (but overlapping) taxa and/or life history stages. This results in complex and diverse drivers of shark catches across the case study sites, with interactions between local and global market demand, as well as other social and practical determinants of fisher behaviour. Each of the fisheries are also somewhat representative of other similar fisheries elsewhere in Indonesia, and the tropics more generally (e.g., Acebes & Tull, 2016; Gupta et al., 2020; Harry et al., 2011; Smart et al., 2020). Therefore, some of the findings and lessons learned may be applicable to other places and countries.

1.3. Aims and objectives

Within this context, the overarching aim of this DPhil is to contribute to more effective design of shark and ray conservation interventions, which can simultaneously deliver conservation outcomes for sharks (with a particular focus on threatened species) and well-being outcomes for people (with a particular focus on vulnerable coastal communities).

To achieve this aim, the objectives of my research are:

1. To critically assess current approaches for understanding and managing threats to sharks, and highlight gaps relating to understanding and addressing socio-economic issues
2. To develop general decision-making frameworks, which can better integrate socio-economic issues, and support interdisciplinary approaches to shark conservation and fisheries management
3. To investigate the key socio-economic drivers of shark fishing and related behaviours in small-scale fisheries in Indonesia, and barriers to pro-conservation behaviour
4. To assess the impact of an existing integrated conservation intervention for sharks in Indonesia, and document lessons learned for designing future interventions
5. To use predictive methods to explore how hypothetical conservation interventions could impact shark conservation and human well-being in small-scale fisheries, and offer design options for cost-effective interventions in the future
6. To explore long-term sustainable financing mechanisms for shark conservation.

Each of these objectives will provide a set of management-relevant research findings, to meet the overarching aim.

1.4. Thesis outline

Chapter 1: Introduction. In this first chapter, I introduce the thesis and its contributions to conservation science, including a problem statement, an overview of the study system, the thesis outline, other research and impact that derived from my DPhil, and a positionality and ethics statement.

Chapter 2: The neglected complexities of shark fisheries, and priorities for interdisciplinary risk-based management. Here I provide an overview of typical measures for understanding and managing threats to sharks in fisheries, and highlight critical gaps relating to incorporating socio-economic factors into research and management. I then propose a framework for assessing feasibility in a shark management context, which could be integrated with traditional fisheries risk assessments in order to bridge this gap.

This chapter is published as: Booth, H., Squires, D., & Milner-Gulland, E. J. (2019). The neglected complexities of shark fisheries, and priorities for holistic risk-based management. *Ocean & Coastal Management*, 182(September), 104994. <https://doi.org/10.1016/j.ocecoaman.2019.104994>. I was responsible for conception, design and drafting of the chapter and its intellectual content. DS and EJMJ reviewed drafts, provided critical revisions, and gave final approval of the version to be published.

Chapter 3: The mitigation hierarchy for sharks: a risk-based framework for reconciling trade-offs between shark conservation and fisheries objectives. Here I build on Chapter 2 to propose the mitigation hierarchy as a novel framework for supporting complex fisheries management decisions relating to shark conservation. Through providing examples from real-world fishery management problems, I illustrate how the mitigation hierarchy can be applied to a range of species, fisheries and contexts, and can incorporate fishing mortality risks and socio-economic feasibility.

This chapter is published as: Booth, H., Squires, D., & Milner-Gulland, E. J. (2019). The mitigation hierarchy for sharks: A risk-based framework for reconciling trade-offs between shark conservation and fisheries objectives. *Fish and Fisheries*, November, 1–21. <https://doi.org/10.1111/faf.12429>. I was responsible for conception, design and drafting of the chapter and its intellectual content, with guidance from DS and EJMJ on mitigation hierarchy concepts. DS and EJMJ reviewed drafts, provided critical revisions, and gave final approval of the version to be published.

Chapter 4: Impact and lessons learned from an integrated conservation intervention for manta rays. Here I describe an integrated intervention to reduce manta hunting and mortality in a small-scale targeted mobulid fishery in Eastern Indonesia, and assess its impact over a five-year period (2013–2018) using a theory-based research design. Based on the results I make several recommendations for designing interventions to mitigate trade-driven over-exploitation of marine megafauna.

This chapter is published as Booth, H., Mardhiah, U., Siregar, H., Hunter, J., Giyanto, Putra, M. I. H., Marlow, J., Cahyana, A., Boysandi, Demoor, A. Y. L., Lewis, S., Adhiasto, D., Adrianto, L., & Yulianto, I. (2020). An integrated approach to tackling wildlife crime: impact and lessons learned from the world's largest targeted manta ray fishery. *Conservation Science and Practice*, 1–18. <https://doi.org/10.1111/csp2.314>.

I conceptualised and designed the paper and methods, conducted the overarching theory-based impact analysis, and co-ordinated writing of the manuscript. UM provided support with statistical analysis; AC provided support with spatial analysis and figures; JH provided guidance on crime science; JM provided inputs on livelihoods and outreach components; HS, MIHP, SL, B and AYLD were involved in data collation; myself, JH, G, MIHP, JM, B, AYLD, SL, DA and IY were involved in intervention design and implementation; LA, DA and IY supervised intervention design and project implementation. Dr Matthew Linkie, Prof. E.J. Milner-Gulland and Dr David Wilkie also provided helpful inputs on drafts of the manuscript.

Chapter 5. A socio-psychological approach for understanding and managing bycatch in small-scale fisheries. Here I present a novel approach for understanding bycatch in small-scale fisheries, drawing on well-established theories from behavioural and social sciences. I first typify bycatch as a spectrum rather than a clearly delineated component of catch, where the position of a species on this spectrum depends on fishers' beliefs regarding the outcomes of bycatch-relevant behaviour. I then outline an approach to diagnose the underlying socio-psychological drivers of bycatch, based on the Theory of Planned Behaviour. Finally, I illustrate the approach using an empirical case study, exploring fishers' beliefs regarding bycatch-relevant behaviour for three endangered species in a small-scale gill net fishery in Indonesia (Lhok Rigaih, Aceh Jaya), which regularly interacts with three endangered shark species: wedgefish, hammerheads and whale sharks. I show how a socio-psychological approach can help to identify conflicts and synergies between bycatch mitigation and fishers' beliefs, thus informing more effective and socially-just interventions for marine megafauna conservation.

This chapter has been submitted for peer review in *Marine Policy*, and published as a pre-print: Booth, H., Ichsan, M., Hermansyah, R. F., Rohmah, L. N., Naira, K. B., Adrianto, L., & Milner-Gulland, E. J. (2021). A socio-psychological approach for understanding and managing bycatch in small-scale fisheries. *OSF Preprints*. <https://doi.org/10.31219/OSF.IO/P4AHZ>. I conceptualised and designed the approach, methods and research instruments, with guidance and supervision from EJMG. I conducted and co-ordinated data collection, with assistance from MI, RFH, LNR and KBN; and conducted analysis and write-up, with supervision and review from EJMG. LA provided supervision for RFH and LNR.

Chapter 6: Exploring cost-effective management measures for reducing risks to threatened sharks in a problematic longline fishery.

Here I analyse five-years of fishery landings and profit data from a small-scale targeted shark fishery (Tanjung Luar, Lombok) to identify and assess cost-effective fisheries management measures for reducing the risk of capture for conservation priority (i.e., threatened and CITES-listed species). I used a Boosted Regression Tree (BRT) method to assess the relative influence of different plausible management measures (e.g., effort restrictions, gear restrictions, spatio-temporal closures) on: 1) risk of capture for each priority taxon, and 2) trip profit, and develop predictive models for these outcome variables. I then use the prediction datasets to conduct a semi-quantitative assessment of the hypothetical cost-effectiveness of different plausible management measures, based on the estimated conservation benefit (as reduced risk of capture) and socio-economic cost (as relative profit forgone) of each potential measure.

This chapter has been submitted for peer review in *Ocean and Coastal Management* as Booth, H., Powell, G., Yulianto, I., Simeon, B., Muhsin, Adrianto, L., Milner-Gulland, E. J. (IN REVIEW). Exploring cost-effective management measures for reducing risks to threatened sharks in a problematic longline fishery. *Ocean & Coastal Management*. I conceptualised the study and methods, conducted the analysis and led on writing the manuscript, with supervision and critical inputs from EJMG. GP provided support with methods and statistical analysis. BS and M collected and collated the data, with supervision from IY and LA.

Chapter 7: Estimating economic losses to small-scale fishers from shark conservation: a hedonic price analysis. Following on from Chapter 5, with the aim of improving my understanding of the potential economic opportunity costs of shark conservation for small-scale fishers in Tanjung Luar, I applied a revealed preference method of economic valuation—hedonic price analysis (HPA)—to understand local shark markets and values. The results gave significant marginal price estimates for five species of conservation and commercial importance, and I use these estimates to predict the economic opportunity costs of conservation measures - such as catch limits for endangered and CITES- listed species - for low-income fishers.

This chapter is published as Booth, H., Squires, D., Yulianto, I., Simeon, B., Muhsin, Adrianto, L., & Milner-Gulland, E.J (2021). Estimating economic losses to small-scale fishers from shark conservation: A hedonic price analysis. *Conservation Science and Practice*, June, e494. <https://doi.org/10.1111/csp2.494>. I conceptualised the study and methods, and conducted formal analysis with supervision and guidance on econometrics from DS and EJMG. IY, BS and M conducted and coordinated data collection and management. I led on writing of the manuscript, with review and supervision from LA, DS, EJMG.

Chapter 8. Designing locally-appropriate conservation incentives for small-scale fishers. Here I use a novel combination of methods - scenario interviews with contingent valuation (CV) - to investigate how incentive-based interventions might influence fisher behaviour and reduce mortality of Critically Endangered taxa (hammerhead sharks and wedgefish) in two case study SSFs (Tanjung Luar and Lhok Rigaih). I use the results to estimate and compare the impact of different hypothetical interventions on conservation outcomes and fisher well-being, and the potential cost and cost-effectiveness of a performance-based reward for reducing landings of the study taxa.

This chapter has been submitted for peer review in *Biological Conservation*, and published as a pre-print: Booth, H., Ramdhan, M. S., Hafizh, A., Wongsopatty, K., Mourato, S., Pienkowski, T., Adrianto, L., & Milner-Gulland, E. J. (2021). Designing locally-appropriate conservation incentives for small-scale fishers. *OSF Preprints*. <https://doi.org/10.31219/OSF.IO/BXZFS>. I conceptualised the study and methods and conducted formal analysis, with guidance on statistical analysis from TP, and overall supervision and guidance from SM and EJMG. I conducted and coordinated data collection, with assistance from SMR, AH and KW. I led on writing of the manuscript, with review by SM, TP and EJMG. LA provided supervision for AH and KW.

Chapter 9. Tourism levies: operationalising the beneficiary-pays principle for just and equitable marine conservation. Here I operationalise the beneficiary-pays principle via marine tourism levies, as a socially-progressive sustainable financing mechanism for community-based marine conservation. I use an online contingent valuation (CV) study to measure international tourists' willingness-to-pay (WTP) towards community-based marine conservation, where the payment would be used to directly incentivise local fishers to reduce catches of endangered sharks and rays (as per the scenarios explored in Chapter 8). I validate the results using regression models against widely-accepted determinants of WTP, and by comparing CV results to real donation behaviour by the respondents. I then combined the CV results with quantitative and qualitative data from two real marine tourism hotspots in Indonesia (Lombok and Pulau Weh) to explore the feasibility of implementing tourism levies in these sites, to directly incentivise pro-conservation behaviour via payments for ecosystem services (PES) in two local SSFs (Tanjung Luar and Lhok Rigaih). The results indicate that income from marine tourism levies could be several times greater than the estimated costs for PES schemes in local SSFs, and shows that the global marine tourism industry offers an under-utilised revenue stream for marine conservation, which, when coupled with well-designed PES schemes, could have direct positive outcomes for marine biodiversity and coastal communities.

This chapter has been submitted for peer review in *Ecological Economics* as Booth, H., Mourato, S., & Milner-Gulland, E. J. (IN REVIEW). Operationalising tourism levies for just and equitable marine conservation. *Ecological Economics*. I conceptualised the study and methods, collected data, and

conducted formal analysis, with guidance on survey design and analysis from SM and EJMG. I led on writing the manuscript, with review and supervision from SM and EJMG.

Chapter 10. Bycatch levies: operationalising the polluter-pays principle to reconcile trade-offs between blue growth and biodiversity conservation. Here I explore bycatch levies as a market-based instrument for reconciling trade-offs between blue growth and marine biodiversity conservation. I outline the theory and practice of bycatch levies to demonstrate how they could incentivize bycatch prevention and raise revenue for compensatory conservation, provided they are well designed, as part of a policy mix for sustainable and equitable ocean governance. I then explore ways forward for mainstreaming bycatch levies into the blue economy, and demonstrate that they could raise billions of dollars towards closing global biodiversity financing gaps, delivering net outcomes for biodiversity under the United Nations Post-2020 Global Biodiversity Framework while enabling blue growth, and moving towards win–wins for economic welfare and biodiversity conservation.

This chapter has been published as Booth, H., Arlidge, W. N. S., & Squires, D. (2021). Bycatch levies could reconcile trade-offs between blue growth and biodiversity conservation. *Nature Ecology & Evolution*. <https://doi.org/10.1038/s41559-021-01444-w>. I conceptualised the paper and led on writing and subsequent revisions. W.N.S.A., D.S. and E.J.M.-G. provided substantial inputs, guidance and mentorship throughout, supporting supervision, review and editing.

Chapter 11. Discussion. This chapter synthesises the main findings of this thesis. I revisit the aim and objectives of the research, and outline novel contributions. I identify cross-cutting themes, limitations and future directions for research and practice.

1.5. Additional research

Throughout my DPhil I have had the opportunity to lead and contribute to several other projects relating to conservation science and practice. Outputs from these efforts include a further four lead-author publications, and eight co-authored publications (listed below). A further co-authored publication is in press and two are in review (not listed below).

1.5.1. Lead authorship

Booth, H., Clark, M., Milner-Gulland, E.J., Amponsah-Mensah, K., Antunes, A.P., Brittain, S., Castilho, L.C., Campos-Silva, J.V., Constantino, P. de A.L., Li, Y., et al. *Current Biology* (2021).

Investigating the risks of removing wild meat from global food systems. 1–10.

<https://doi.org/10.1016/j.cub.2021.01.079>

Booth, H., Arias, M., Brittain, S., Challender, D., Khanyari, M., Kupier, T., Li, Y., Castro, A.O., Oyanedel, R., Pienkowski, T., et al. (2021). “Saving lives, protecting livelihoods, and safeguarding nature”:

risk-based wildlife trade policy for sustainable development outcomes post-COVID-19. *Front Ecol Evol.* <https://doi.org/10.3389/fevo.2021.639216>

Booth, H., Chaya, F., Ng, S., Tan, V., Rao, M., Teepol, B., Matthews, E., Lim, A., & Gumal, M. (2021). Elasmobranch fishing and trade in Sarawak, Malaysia, with implications for management. *Aquatic Conservation: Marine and Freshwater Ecosystems*, January, 1–16. <https://doi.org/10.1002/aqc.3688>

Booth, H. (2019). Knowing and doing - conservation charities must spend wisely. *Nature* 568. <https://doi.org/10.1038/d41586-019-01058-0>

1.5.2. Co-authorship

Zhou, X., Booth, H., Macmillan, D. C., & Veríssimo, D. (2021). Leveraging shark-fin consumer preferences to deliver sustainable fisheries. *Conservation Letters*, e12842(August), 1–8. <https://doi.org/10.1111/conl.12842>

Petrovan, S., Aldridge, D., Bartlett, H., Bladon, A., Booth, H., Broad, S., Broom, D., Burgess, N., Cleaveland, S., Cunningham, A., et al. (2021). Post COVID-19: a solution scan of options for preventing future zoonotic epidemics. *Biological Reviews*. <https://doi.org/10.1111/brv.12774>

Maron, M., Bignoli, D. J., Krueger, L., Kiesecker, J., Kumpel, N. F., ten Kate, K., Milner-Gulland, E. J., Arlidge, W. N. S., Booth, H., Bull, J. W., et al. (2021). Setting robust biodiversity goals. *Conservation Letters*. <https://doi.org/10.1111/conl.12816>

Castellanos-Galindo, G. A., Herrón, P., Navia, A.F., Booth, H. (2021). Shark conservation and blanket bans in the eastern Pacific Ocean. *Conservation Science & Practice*. <https://doi.org/10.1111/csp2.428>

Milner-Gulland, E.J., Addison, P., Arlidge, W., Baker, J., Booth, H., Brooks, T., Bull, J., Burgass, M., Ekstrom, J., Ermgassen, S. zu, et al. (2020). Fthe Steps for the Earth: mainstreaming the post-2020 Global Biodiversity Framework. *One Earth* 2050, 75–87. <https://doi.org/10.1016/j.oneear.2020.12.011>

Gupta, T., Booth, H., Arlidge, W., Rao, C., Manoharakrishnan, M., Namboothri, N., Shanker, K., and Milner-Gulland, E.J. (2020). Mitigation of elasmobranch bycatch in trawlers: A case study in Indian fisheries. *Front Mar Sci* 7, 571. <https://doi.org/10.3389/fmars.2020.00571>

Mustika, P.L.K., Ichsan, M., and Booth, H. (2020). The Economic Value of Shark and Ray Tourism in Indonesia and Its Role in Delivering Conservation Outcomes. *Front Mar Sci* 7, 261. <https://doi.org/10.3389/fmars.2020.00261>

Arlidge, W.N.S., Squires, D., Alfaro-shigueto, J., Booth, H., Mangel, J.C., and Milner-Gulland, E.J. (2020). A Mitigation Hierarchy Approach for Managing Sea Turtle Captures in Small-Scale Fisheries. *Front Mar Sci* 7. <https://doi.org/10.3389/fmars.2020.00049>

1.6. Positionality and ethics

This interdisciplinary thesis draws on natural and social sciences, and includes the collection and analysis of both objective and subjective data. It therefore incorporates elements of both positivist and interpretivist research philosophies, i.e., attempts to objectively explain and predict phenomenon, as well as understanding relative and socially-constructed meanings.

It is widely acknowledged - particularly in social science disciplines and interpretivist research philosophies – that relationships between researchers and subjects are interactive, and that a researcher's positionality influences the research process. Positionality refers to a researcher's unique combination of social identities (e.g., background, education, gender, ethnicity, language) and how they interact with other people and places, which in turn shapes the nature of inquiry by influencing how a researcher gathers, evaluates, synthesizes and communicates information (Moon et al., 2019). Since my identities and experiences have uniquely influenced the knowledge production processes during this thesis, it is important to practice reflexivity, as it can identify sources of bias and potential ethical dilemmas, and contribute to the advancement of conservation science as a more pluralistic and inclusive discipline (Brittain et al., 2020; Moon et al., 2019).

To appropriately situate the contents of this thesis within my positionality, I present here: a brief reflection on my social identities and background, my overall intellectual position and conservation philosophy, and a statement on research ethics and the subject-researcher relationship, with some reflections on how each has potentially influenced the knowledge production process during my thesis.

1.6.1. My social identities, and the journey to my DPhil

I am a British, white working-class female who grew up on the outskirts of Birmingham – the UK's second largest city. As a child, I wasn't exposed to tropical climates or charismatic megafauna, however my family (in particular, my dad and his parents) instilled a fascination for 'backyard biodiversity' (mostly birds and butterflies) and a love of the outdoors from an early age. For me, all forms of nature hold strong intrinsic value, for their inherent beauty and evolutionary ingenuity. As a teenager, I was lucky to excel in mathematics and the sciences, and I was always interested in understanding how the world worked – from particles to planets and everything in between. This led to me studying maths, biology, chemistry and physics for my A-levels, and eventually an undergraduate degree in Natural Sciences at Cambridge University (2007-2011). During this time, my epistemology was firmly grounded in positivism – I enjoyed the absolute truth of mathematics (it was always possible to arrive at a correct answer), and was fascinated with understanding objective laws of physics, physiology, neuroscience and ecology. I eventually specialised in Zoology, with a focus on animal behaviour and neuroscience. At this point, I had never been explicitly exposed to social sciences or different philosophies of knowledge. However, I had grown passionate about issues of global economic inequality and injustice, and wanted to find ways to

help people less fortunate than myself. In the final year of my Zoology specialisation the animal physiology course was cancelled due to staff illness, and I ended up opting for ‘conservation biology’ instead. As a student indoctrinated in positivism, I found it all quite wishy-washy and unscientific. That is until one day, while revising for my final year exams, when it all seemed to fall in to place. I realised that conservation was where my passion for scientific enquiry, my love for nature, and my desire to tackle economic inequalities all overlapped. From there began my journey from a positivist natural scientist, to embracing social science principles and diverse epistemologies.

There are several notable experiences that have shaped this journey, and my world view today. Firstly, studying Management Studies in my final year at Cambridge, where I had my first formal social sciences tuition, and learned about economics, organisational behaviour and environmental policy. Secondly, working in the Ecosystem Assessments Program at UNEP-WCMC, where I was exposed to the concept of ecosystem services, as an approach for communicating the value of nature to decision-makers and the general public. Thirdly, spending time volunteering and working in rural communities and wilderness areas in Africa (mostly Kenya, Ethiopia and Tanzania), where I was exposed to life below the poverty line, the front lines of elephant and rhino conservation, and the conflicts and ethical dilemmas that occur between the two. Fourth, my Masters in Conservation Science at Imperial College London (2015-16) helped me to become a fully-fledge interdisciplinary researcher. I learned of the terms ‘epistemologies’ and ‘reflexivity’, the difference between induction and deduction, and was able to formally apply social sciences research techniques in my Master’s thesis, which explored illegal manta ray trade in Indonesia, and the associated moral and practical dilemmas. Fifth, before embarking on my DPhil, I spent over 3 years working as a shark and ray conservation practitioner in Indonesia. This gave me first-hand exposure to the lives, perspectives and struggles of shark fishers, and the moral and practical imperative to include people in conservation decision-making and ensure they are no worse off. Finally, I think it is worth noting that while I certainly think sharks are fascinating and worthy of conservation attention, sharks were never my favourite animal, and I did not set out to dedicate my career to shark conservation. My career has been quite diverse in terms of issues, ecosystems and species, and I don’t hold any particularly strong intrinsic or moral values for any one issue. Rather, I am interested in solving complex problems and having positive impact, at the intersection of nature and human well-being. It struck me that shark conservation was particularly complex, and perhaps lagging in terms of interdisciplinary thinking, therefore it represented an area where I could perhaps add some value.

1.6.2. My intellectual position and conservation philosophy

My unique combination of identities and experiences means that while I have a deep intrinsic love for nature, and a strong background in natural sciences and positivist research, I am also empathetic towards other people’s world views, and care deeply about preventing injustices and ‘doing no harm’ to vulnerable communities who depend on nature. Perhaps the fact that I’m not a ‘shark fanatic’ also allows me to be

somewhat more neutral when it comes to emotional reactions and values regarding their plight, and thus somewhat sympathetic towards the positions of fishers. I am also a pragmatist rather than an idealist, and this comes across in my embrace of market-based approaches, private sector engagement and the search for win-win solutions (which work with capitalism and principles of economic development, rather than more radical approaches which propose dismantling these systems all together). The urgency of conservation means there is a need to find solutions that can be widely accepted and implemented immediately. I do wish that everyone cared as deeply as I do about nature, and would be willing to adopt radical change, but conservation needs a pragmatic middle ground to move us from the business-as-usual trajectory, in which economic entities continue to damage nature unabated.

Overall, according to the Future of Conservation typologies (Sandbrook et al., 2019), I am more skewed towards ‘people-centred conservation’ and ‘conservation through capitalism’, and relatively less skewed towards ‘science-led ecocentrism’ (though I do believe that conservation should be evidence-based, however the evidence should be pluralistic and acknowledge diverse values) (Figure 1.3). These values and philosophies have shaped my DPhil – from objectives to methodologies to interpretation – and are thus strongly reflected in this thesis.

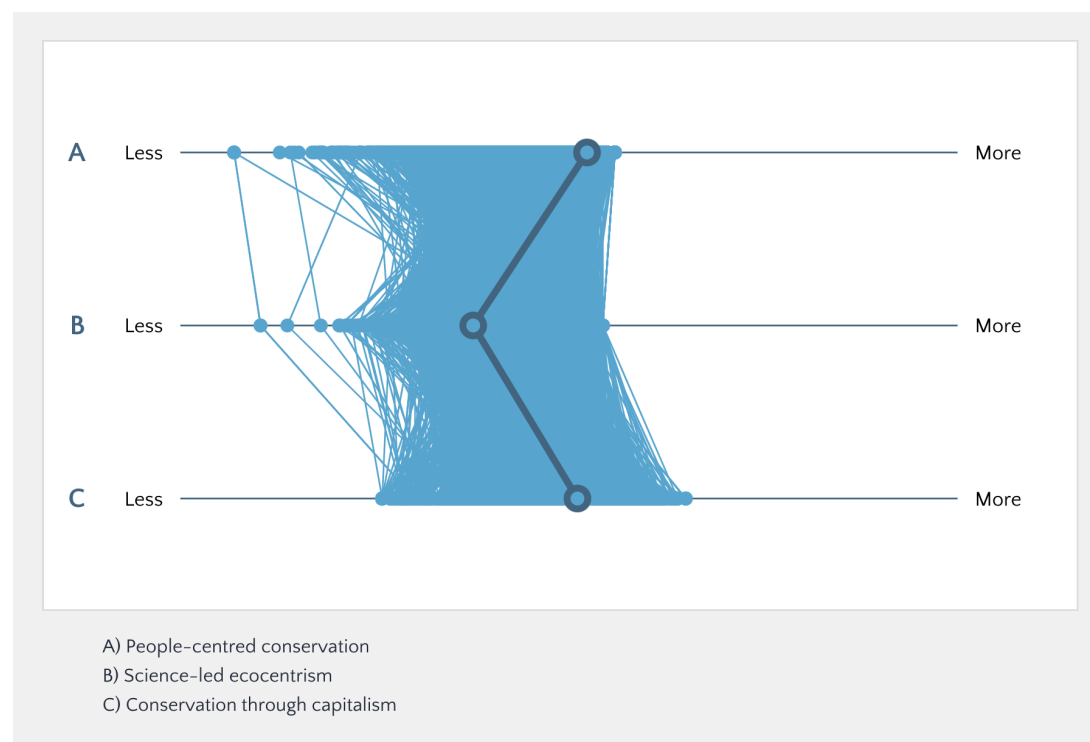


Figure 1.3 My results from the Future of Conservation Survey

It is also important to position my identities and philosophies with the research context of coastal communities in Indonesia. Being a white female researcher can cause a range of biases, from fishers being reluctant to talk with me due to suspicion of outsiders, to fishers being particularly enthusiastic to talk

with me due to novelty and interest. In general, I found fishers to be friendly, open and bemused that I was interested in their lives and livelihoods. All participation was voluntary, and fishers were free to leave at any time, such that any fishers who appeared shy or hesitant did not participate. Of course, it is possible that fishers may have given strategic responses, especially regarding questions about hypothetical payment schemes and conservation values, however this was somewhat pre-empted when local research assistants collected data.

Language barriers and interpretation of language, especially for terms that do not directly translate across languages, create a further potential source of bias. I am relatively fluent in Bahasa Indonesia, but I do not speak local Indonesian languages, and can struggle with colloquialisms or technical terminology. I was supported by a team of local research assistants, with fluency in Bahasa Indonesia, English and local languages, however I acknowledge that my own subjectivities, and those of the research assistants, will have influenced translation and interpretation of fisher responses.

1.6.3. Research ethics and the subject-researcher relationship

All research involving human subjects was approved by the University of Oxford Medical Sciences Interdivisional Research Ethics Committee (MS IDREC) (ref. R66416/RE001). All research in Indonesia was conducted under a foreign research permit (No. Surat Izin: 407/E5/E5.4/SIP/2019), with permissions from the Indonesian Ministry of Research and Technology (RISTEK). Before commencing research in any village, I gained the free, prior, and informed consent (FPIC) of the heads of the local fisheries departments and villages. All respondents and participants also gave their free prior and informed consent verbally before participation, during which we (myself and the research assistants) explained our identities, the purpose of the research, and guarantees of confidentiality. Verbal consent was considered appropriate due to high levels of illiteracy amongst coastal communities. All data were anonymous, i.e., we did not collect names or individual identifiers from participants.

Since shark fishing is almost entirely legal and socially-accepted in Indonesia (aside from a few exceptions for protected species, such as manta rays) the data collected for this thesis do not present any special ethical concerns in terms of sensitive information. Nonetheless, there may have been misperceptions, social responses biases and power imbalances, which create ethical dilemmas and biases. For example, some fishers may misunderstand existing conservation laws, or fear that they will get in to trouble for sharing information on their catches. Though participation was voluntary, and respondents were free to leave any time, fishers may have felt obliged to participate. The interview will have also used up their valuable time (when not at sea fishers spend most of their time resting and preparing for their next trip), and could create unrealistic expectations, both positive and negative – such as concerns that new rules will be set in the future, or expectations that payment schemes will be established. The FPIC process should have mitigated some of these risks but may not have eliminated them. I also gave small gifts (*oleh*

oleh) to interview participants at the end of each interview (keyrings from the UK) as a sign of gratitude and goodwill, and will return to the villages to present findings, and explore implications and next steps.

Finally, the wider benefits and beneficiaries of this research warrant ethical consideration. I myself am a primary beneficiary of this research, since I will attain a DPhil qualification and publications, which will support my career as an interdisciplinary researcher. The research assistants I worked with also gained short-term employment, training and experience in novel research methods, and four Masters students from IPB were able to collect and analyse data for their Masters theses, which will support their career development and employment prospects. The research assistants are also co-authors on publications from this thesis, which will help to build their international reputations as conservation researchers. The Indonesian Ministry of Marine Affairs and Fisheries and conservation NGOs are also potential beneficiaries, since the results can be used to inform conservation and fisheries management strategies. Regarding the research participants – whether they directly benefit from this research primarily depends on if and how the results are used to inform future conservation interventions. It is possible that they will directly benefit in the medium to long-term (e.g., if performance-based conservation payments are introduced, and help to deliver well-being and biodiversity outcomes), but this remains uncertain. They may also indirectly benefit from the opportunity to express their voice and concerns regarding conservation interventions that impact them, and thus socio-economic issues in marine management may be given more due attention by NGOs and government bodies in the future. I also hope this research will benefit threatened sharks and rays of Indonesia, by helping to design more effective conservation interventions in Indonesia and beyond, especially: wedgefish, hammerhead sharks and mobulid rays.

2. The neglected complexities of shark fisheries, and priorities for holistic risk-based management

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Photo: A man carrying a Critically Endangered juvenile hammerhead shark. He's taking it home for his family to eat. Lhok Rigaih, Aceh.

"If you want to truly understand something, try to change it" - Kurt Lewin.

2.1 Introduction

Reversing the decline of sharks requires that exploitation of populations is appropriately managed. Several multi-lateral environmental agreements, such as the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and Regional Fisheries Management Organisations (RFMOs), have created international frameworks to regulate fishing and trade for several species. Yet in order to deliver meaningful conservation outcomes, these efforts must translate into fisheries management action at national and local levels, which can also be adopted at scale. Specifically, management actions are required that reduce shark fishing mortality, particularly for threatened species and in countries with the largest catches. However, robust shark fisheries management remains the purview of a few market-oriented more economically developed countries (MEDCs) (e.g., Australia, New Zealand, USA) (Simpfendorfer & Dulvy, 2017). Yet fisheries in many lower-income countries, which constitute the majority of global shark production, remain under-managed (Momigliano & Harcourt, 2014; Simpfendorfer & Dulvy, 2017). As such, sharks continue to be overfished across most of the world (Davidson et al., 2016; Dulvy et al., 2021).

Management in lower-income countries can be hampered by limited resources and capacity (Dharmadi et al., 2015; Momigliano & Harcourt, 2014). Regulatory action is further complicated by the prevalence of small-scale mixed-species fisheries, which are ubiquitous throughout the coastal waters of fishery-dependent developing nations. Small-scale fisheries (SSFs) can be responsible for significant proportions of shark fishing mortality. Yet these fisheries are often informal, unmonitored and unmanaged, with socially-oriented governance. Further, the coastal communities that are dependent on SSFs are often poor and socio-economically vulnerable (Glaus et al., 2018; Jaiteh, Loneragan, et al., 2017; Lestari et al., 2017; I. Yulianto et al., 2018). However, this does not have to be a barrier to achieving marine conservation outcomes. In some cases, high dependence on coastal resources may facilitate better management, when coupled with strong local engagement in management and robust socio-cultural governance systems. In these cases, 'bright spots' in lower-income countries can far outperform management in developed nations (Cinner et al., 2016). As such, lack of resources may not necessarily preclude good management, provided there is sufficient political will and effective institutions for environmental leadership (Steinberg, 2001). What is more, reducing shark fishing mortality ultimately requires changing human behaviour. In particular, it requires influencing the decisions of fishers and skippers at the point of catch. As such, there is a need for a social sciences perspective on shark fisheries: a perspective which can facilitate the design of local-level bottom-up approaches to complement macro-scale policy interventions, and enable implementation. Despite this, socio-economic factors are rarely incorporated into shark fisheries management research, and are not typically considered in shark risk assessments or conservation strategies (Mizrahi et al., 2019). With significant research gaps on the human dimensions of shark conservation, there are calls for greater inclusion of local people in shark management planning (MacKeracher et al., 2018; Mizrahi et al., 2019; Rigby, Appleyard, et al., 2019; Simpfendorfer et al., 2011).

This plurality of contexts also necessitates over-arching frameworks that can integrate complex, multifarious fisheries and their diverse management goals to create overall positive impact at scale. No net loss (NNL) of biodiversity has been put forward as a goal for the Convention on Biological Diversity's (CBD) post-2020 agenda, and a global mitigation hierarchy for nature has been proposed as a framework for delivering this, which allows a multitude of goals and targets to be set at different levels, and integrated into an ambitious whole (Arlidge et al., 2018; CBD, 2020a; Milner-Gulland, Addison, et al., 2020). However, measuring and accounting for social impacts and incentivising compliance within this framework remains poorly understood, and is a priority for the up-coming re-negotiation of the CBD (Fulton et al., 2011; Milner-Gulland et al., 2018).

Acknowledging these needs, this chapter explicitly demonstrates and describes the socio-economic implementation gap in shark fisheries management and conservation, explores why it occurs, and outlines some of the implications. Building on these findings, I propose a practical solution for filling this gap, which is focused on including socio-economic feasibility assessments as an integral part of management decision-making for sharks. I also outline research priorities for adopting feasibility assessments in the future, and propose how feasibility assessments could be incorporated into risk-based decision-making at multiple scales, to create better outcomes for sharks and people. I take a risk-based approach, since risk assessments are commonly used to understand the impacts of economic development activities on natural resources (e.g., through Environmental and Social Risk Assessments (ESRA)) and threats to sharks in fisheries (e.g., Cortés *et al.*, 2010; Dulvy *et al.*, 2014; Griffiths *et al.*, 2019); and provide a practical, data-driven means for prioritising management action, which can be used flexibly in data-poor contexts, as is needed for sharks (Arrizabalaga et al., 2011; Braccini et al., 2006; Cortés et al., 2010; S. Griffiths et al., 2019).

2.2 Typical measures for managing shark mortality

2.2.1 Biological and technical risk

Over the past decade, much applied research for shark management has focused on understanding biological (i.e., intrinsic physiological and life history characteristics) and technical (i.e., fisheries operations and technology) factors that influence overfishing and extinction risk in sharks. There is now a considerable body of evidence describing these factors, and their roles in the risk of shark overfishing. Biological factors include the influence of size, fecundity, habitat preference, depth range, and geographic range of on risk of capture and overexploitation (Dulvy *et al.*, 2014); and the influence of morphology, locomotor performance, and respiratory and metabolic physiology on post-capture mortality (Braccini et al., 2012; Gallagher et al., 2014; Manire et al., 2001). Technical factors include those relating to overall fishing pressure, as well as the fishing process and technology used, such as gear type and associated modifications (e.g., leader material, use of BRTs and electro-sensory deterrents), set depth, soak time,

fishing ground, fishing time, fishing season, target species, bait and post-capture handling practices (Dapp et al., 2016; Dulvy et al., 2021; Gallagher et al., 2014; James et al., 2016; Craig P. O'Connell et al., 2014; Oliver et al., 2015; Pacoureau et al., 2021; Patterson et al., 2014; Poisson et al., 2010; Thorpe & Frierson, 2009; Ward et al., 2008) (Table 2.1).

These factors represent varying degrees of risk to different shark species in different fisheries contexts. They can be incorporated into semi-quantitative ecological risk assessments for sharks, such as Productivity-Susceptibility Analyses (PSA) or EASI-Fish (Ecological Assessment of the Sustainable Impacts by Fisheries). These approaches quantify the relative vulnerability of shark species to fisheries by combining productivity (i.e., biological) and susceptibility (i.e., technical) variables, and can be particularly useful in data-limited situations (Arrizabalaga et al., 2011; Cortés et al., 2010, 2015; Gallagher et al., 2012; S. Griffiths et al., 2019).

Understanding these biological and technical risk factors is important because they allow scientists and managers to assess the vulnerability of different species within a comparative framework. The results can then be used in prioritisation, management strategy design and identification of technical measures to reduce the risk of fishing mortality for sharks (Table 2.1). For example, use of nylon leaders and circle hooks can reduce shark mortality in pelagic longline fisheries (Cooke & Suski, 2004; Ward et al., 2008). Modifying mesh sizes and net tension can minimise susceptibility of certain species and life history stages to meshing and entanglement gillnets (Harry et al., 2011; Thorpe & Frierson, 2009). Attractants, deterrents or backdown procedures can reduce capture of pelagic sharks in purse seine vessels fishing on fish aggregation devices (FADs) (Restrepo et al., 2017). The use of exclusion or escape devices are effective for reducing capture of large sharks and rays from trawls (Brewer et al., 2006). However, many of these technical measures, while scientifically tested, are yet to be fully incorporated into fisheries policy and practice.

2.2.2 Macro-economic risk

At the other end of the supply chain, it is widely acknowledged that international demand for shark-derived consumer products creates a macro-economic driving force for shark mortality. In particular demand for fins for shark fin soup (Clarke et al., 2007). This high value market is a driver for targeted shark fishing, finning, and the retention of incidentally caught sharks as marketable secondary catch (Clarke et al., 2007; Davidson et al., 2016). Davidson et al. (2016) also found that the scale of the meat trade influences shark overfishing, while McClenachan et al. (2016) found that economic value is the key factor explaining extinction risk for large-bodied marine species, once they reach a certain threshold value. Species above this threshold include whale sharks (*Rhincodon typus*), hammerhead sharks (*Sphyrna* spp.) and sawfish (*Pristidae* spp.) (McClenachan et al., 2016). Understanding these factors is important, because they can inform high-level international policy and trade-based interventions, such as those under

CITES, as well as interventions in trading and consumer countries to reduce demand and market value. Recognising the importance of international demand, measures have been introduced to regulate markets and trade at the macro-economic level. These measures include fin bans, species-specific trade bans, or countries banning all commercial fishing and trade of sharks and shark products (i.e., 'shark sanctuaries') (Friedman et al., 2018; Shiffman & Hammerschlag, 2016b).

Anthropogenic factors, such as population size and accessibility, and governance factors, such as regulation and marine protected area networks, also play a role in moderating macro-economic drivers (Cinner et al., 2018; Davidson et al., 2016). For example, Cinner *et al.*, (2018) found that coral reefs with lower human impacts consistently supported higher numbers of top predators, including sharks, while Davidson *et al.*, (2016) found that human coastal population size was a key factor in explaining shark overfishing trajectories. As such, understanding broader socio-demographic trends is important for contextualising macroeconomic risk.

2.2.3 Managing risk through direct regulation

Where they are in place, management measures for biological, technical and macro-economic risks tend to be implemented through direct regulation. Direct regulation focuses on mandating specific behaviours or outcomes, usually through technology, process or performance standards, and enforcement of their adoption. Technology standards focus on gear and equipment, while process standards relate to how technology is employed in a fishing operation (i.e., input-orientated). Performance standards focus on the outcomes of a fishing operation, such as catch or mortality (i.e., output-orientated). In the case of managing shark mortality, direct regulations may be imposed on the fishery causing shark mortality, or on the supply chain fuelling the fishery.

In fisheries, input-oriented instruments prescribe alterations to the fishing operation itself. Indeed, one of the most widely adopted approaches for shark conservation is direct regulation of fishing locations through marine reserves or shark no take zones (NTZs) (MacKeracher et al., 2018; Shiffman & Hammerschlag, 2016b; Ward-Paige & Worm, 2017). Other input-orientated measures include regulation of fishing effort, or authorised gears and gear specifications. For example, the shark fisheries management plans for the North West Atlantic and Gulf of Mexico established gear restrictions to reduce bycatch/bycatch mortality, while all trawl nets in Western Australia are required to be fitted with bycatch reduction devices (Table 2.1). Fisheries regulations may also take the form of output-orientated policies, which are based on performance standards, such as the size or amount of catch. Examples include fishing quotas, such as those set for sandbar shark (*Carcharhinus plumbeus*) stocks in the Fishery Management Plan for Sharks of the Atlantic Ocean (Momigliano & Harcourt, 2014). Similarly, the U.S. Atlantic Highly Migratory Species shark fishery has a total trip limit of 36 large coastal sharks, with several species managed as a species complex (Shiffman & Hammerschlag, 2016b). These policies may also restrict

Table 2.1 Summary of biological, technical and macro-economic risks to sharks

Category	Factors	Role in risk	Examples of uses in management	Key references
Biological (i.e., intrinsic physiological and life history characteristics of sharks)bycatch	Size (max length)	Risk of capture in fisheries	Used in fisheries risk assessments, and area-based management associated with critical habitat or aggregations – e.g., Australia enacts time-area closures to protect gummy sharks migrating to pupping grounds.	Manire <i>et al.</i> , 2001; Thorpe and Frierson, 2009; Harry <i>et al.</i> , 2011; Braccini, Van Rijn and Frick, 2012; Dulvy <i>et al.</i> , 2014; Gallagher <i>et al.</i> , 2014; Braccini and Waltrick, 2019
	Depth (min and depth range)			
	Geographic range			
	Habitat-type			
	Morphology (e.g., cephalphoil)			
	Locomotor performance	Risk of post-capture mortality	Used in understanding extinction risk for global conservation prioritisation – e.g., many countries have species-specific restrictions on catch and retention of endangered species, including sawfishes and manta rays.	
	Segregation and schooling (e.g., by size, sex, reproductive stage)			
	Habitat-type (i.e., bottom-dwelling, pelagic, demersal)			
	Respiratory and metabolic physiology			
	Locomotor performance			
Length	Risk of overfishing/ extinction risk			
Slow growth, low fecundity				
Technical (i.e., operational characteristics of fisheries, including fishing process and technology)	Fishing effort	Risk of capture	Used in fisheries risk assessments and in designing fisheries regulations – e.g. all trawl nets in Western Australia are required to be fitted with bycatch reduction devices; all vessels with bottom longline gears operating in NW Atlantic and Gulf of Mexico (U.S.) must have non-stainless-steel corrodible hooks to improve post-release survival of released sharks.	Brewer <i>et al.</i> , 1998; Thorpe and Frierson, 2009; Poisson <i>et al.</i> , 2010; Gallagher <i>et al.</i> , 2014; O’Connell and He, 2014; O’Connell, Stroud and He, 2014; Patterson, Hansen and Larcombe, 2014; Oliver <i>et al.</i> , 2015; Dapp <i>et al.</i> , 2016
	Gear type and modifications			
	Target species			
	Set depth			
	Fishing ground			
	Fishing time (I.e., time of day)			
	Fishing season			
	Soak time			
	Use of deterrents	Risk of post-capture mortality		Braccini, Van Rijn and Frick, 2012; Gallagher <i>et al.</i> , 2014
	Post-capture handling			
	Soak time			
	Target species			
Gear type, and modifications				
Set depth				
Macro-economic (i.e., factors influencing local, national or global trade)	Scale of meat trade	Risk of overfishing and extinction	Used for informing international and national trade and fishing regulations - e.g., Listings on CITES and CMSs; 11 EEZ’s declared as shark sanctuaries’; national export and trade bans enacted.	Clarke, Milner-Gulland and Bjørndal, 2007; Oliver <i>et al.</i> , 2015; Davidson, Krawchuk and Dulvy, 2016; James <i>et al.</i> , 2016; McClenachan, Cooper and Dulvy, 2016; Ward-Paige, 2017; Cinner <i>et al.</i> , 2018
	Scale/value of fin trade			
	Economic value of species			
	Human population and markets			
	Regulatory context			

fishing for threatened species, through a low quota or fishing ban. For example, it is illegal to land whale sharks and manta rays in Indonesia, Malaysia and The Philippines (Friedman et al., 2018).

Macro-economic risks are most commonly managed through performance standards via trade controls, such as species-specific trade bans or low quotas for international export, or domestic bans on the sale of fins or on the commercial sale of all shark products (Friedman et al., 2018; Shiffman & Hammerschlag, 2016b).

2.3 The neglected complexities of managing shark mortality

Implementing measures to address biological, technical and macro-economic risks can undoubtedly reduce fishing mortality and facilitate population recovery. For example, regulations in the Hawaiian longline swordfish fishery require vessels to use a specific combination of technical input controls, reduced shark bycatch by 36% (Gilman *et al.*, 2007). Similarly, science-based management of sandbar shark (*Carcharhinus plumbeus*) stocks in the U.S., involving quotas, permits, time-area closures and species-specific retention restrictions, has supported recovery of this species (Momigliano & Harcourt, 2014).

However, these examples of success come from a handful of high-income countries (Shiffman & Hammerschlag, 2016b; Simpfendorfer & Dulvy, 2017), with resources and fisheries management infrastructures that enable development and enforcement of science-based policies. However, this is atypical for much of the world's shark fishing pressure. The majority of global recorded shark production is derived from lower-income countries (Dent & Clarke, 2014), which are dominated by diverse, unmonitored and unmanaged small-scale fisheries. These governments often possess limited resources for monitoring and compliance management, and uptake of available technical measures is limited (Dulvy et al., 2017; Momigliano & Harcourt, 2014). Where management is in place, it tends to be relatively simplistic, with a focus on trade bans or total bans, and limited evidence of measurable reductions in shark mortality at the stock or fishery level (Friedman et al., 2018; Shiffman & Hammerschlag, 2016b). Regulatory action is further complicated by the socio-economic vulnerability of fishers, and their high dependence on marine resources for income, nutrition and well-being (Glaus et al., 2018; Golden et al., 2016; Jaiteh et al., 2016; Jaiteh, Loneragan, et al., 2017). In short, most approaches to shark management have been developed in high-income market-orientated countries, where scientific and resource capacity is high (Momigliano & Harcourt, 2014; Simpfendorfer & Dulvy, 2017). Yet in the highest priority countries for shark conservation, managing shark fishing is much more than a biological, technical and macro-economic issue; it is also a human issue. Effective management in these contexts necessitates a holistic approach, which acknowledges the need to foster compliance with rules, develop supportive institutions, understand socio-economic barriers to implementation, and consider human behaviour as a key source of uncertainty in fisheries management (Dutton & Squires, 2008; Fulton et al., 2011; Milner-Gulland et al., 2018; Squires & Garcia, 2018).

I propose that current approaches for managing risks to sharks neglect these complexities through three implicit and interlinked assumptions (Figure 2.1):

Assumption 1: the mandated technical measure is the most effective measure that can be adopted to achieve the associated shark management goal.

Assumption 2: fishers, fishing fleets and industry have sufficient capacity and motivation to adopt these mandated measures. This assumption implies that fishers are willing and able to change their behaviour to take up these measures and comply with rules. Taking an instrumental perspective, this assumes that the (positive or negative) economic incentives created by direct regulation favour uptake and compliance, leading to:

Assumption 3: that shark mortality is driven primarily by economic incentives.

However, there is a wealth of evidence that these assumptions are flawed. While legal obligations can be a factor driving fisher behaviour, uptake of technical measures and compliance with regulations depend on a wide range of factors, which are often context-specific (Arias, 2015; Arias et al., 2015; Bergseth & Roscher, 2018; Campbell & Cornwell, 2008; M. A. Hall et al., 2007). Direct regulation rarely creates sufficient incentives to drive compliance, while economic incentives alone are rarely sufficient to change human behaviour (Campbell & Cornwell, 2008; Dutton & Squires, 2008; M. A. Hall et al., 2007; Milner-Gulland et al., 2018). This is especially pertinent to less market-oriented, lower technology fisheries, which are ubiquitous in the world's largest shark fishing nations, and often governed through local social norms and trust-based institutions (R. Q. Grafton, 2005; Kosamu, 2015).

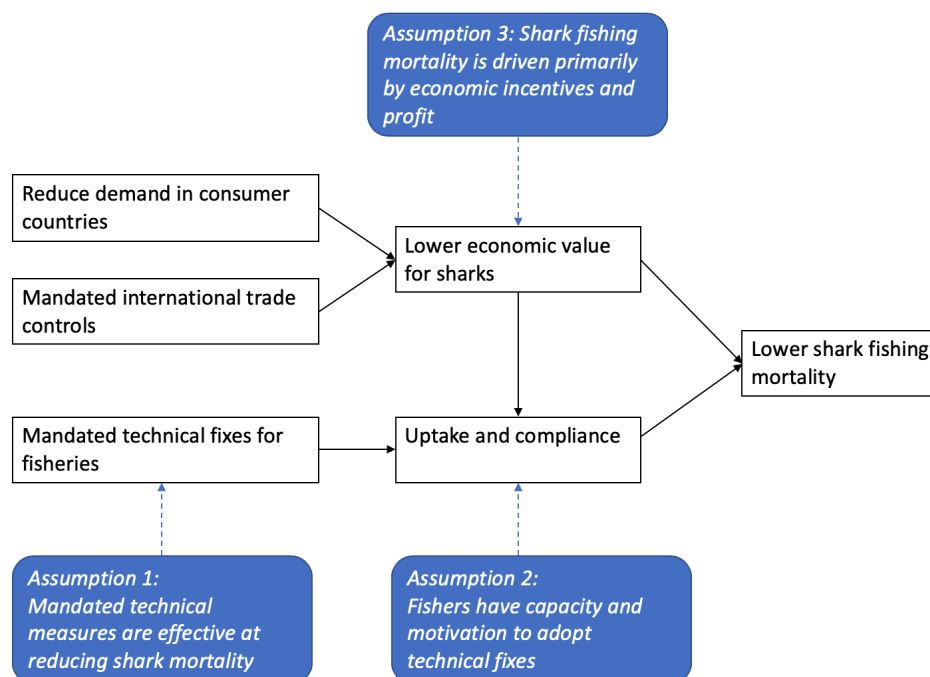


Figure 2.1 A simplified implied theory of change underlying current approaches to reducing shark fishing mortality through technical measures and macro-economic interventions

2.3.1 Assumption 1: Technical measures are effective

Shark management based on direct regulation assumes that the prescribed measure is the most effective approach for reducing risk to sharks in the regulated fishery and context (assumption 1). Yet this is not always the case. The appropriateness of several commonly-applied measures for shark management has been questioned (e.g., Shiffman and Hammerslag, 2016). Excessively prescriptive technical measures can be biologically ineffective, ecologically and socially problematic, difficult and costly to enforce, or insufficiently robust to dynamic changes in the ocean and its users (Jaiteh et al., 2016; MacKeracher et al., 2018; Maxwell et al., 2015; Shiffman & Hammerslag, 2016b; Tolotti et al., 2015).

Ineffective measures

There are several examples of existing mandated measures for shark management that may be of limited effectiveness for reducing fishing mortality. For example, area-based management is one of the most widely advocated and adopted strategies for shark management (MacKeracher et al., 2018), which comes in many forms. These may include time-area closures for specific taxa, though are more commonly applied in the form of strict no-take marine reserves, marine protected areas (MPAs) with some limitations on fishing activity, or ‘shark sanctuaries’ which specifically ban catch and retention of sharks throughout an entire jurisdiction. However, while these approaches represent conceptually-appealing policy wins, the benefits of area-based management for sharks remain species- and context-specific. For example, benefits are likely to be limited to a small number of coastal, small-ranging species or specific life history stages (Jaiteh et al., 2016; Knip et al., 2012; Yates et al., 2016). Benefits of MPAs to larger migratory species, which are often those in need of more urgent conservation action, are rare (F. Graham et al., 2016; Howey-Jordan et al., 2013). Generally, MPAs need to be large, isolated, strictly enforced no-take zones in order to benefit sharks (Edgar et al., 2014). Yet even within some large, isolated MPAs, shark populations continue to decline (N. A. J. Graham et al., 2010; E. R. White et al., 2015). This may be due to insufficient enforcement leading to continued fishing within MPAs (Carr et al., 2013), or displacement of fishing effort to other places, species or life history stages, with sharks remaining vulnerable to fishing pressure in the parts of their range outside of MPAs (O’Keefe et al., 2014). Further, MPAs that are home to a higher abundance of sharks are usually those in more remote locations, which experience low human pressure (Cinner et al., 2018; Edgar et al., 2014). This may be due to MPA establishment being biased towards locations that are ‘residual’ to fishing pressure, with placement favouring ease of establishment over their ability to mitigate threats to sharks (Devillers et al., 2015). Spatial closures for sharks can also lead to unintended negative social consequences (Jaiteh et al., 2016), with social issues often neglected, despite the wide-held belief that social outcomes are essential for enhancing the benefits of MPAs to sharks (MacKeracher et al., 2018; Mizrahi et al., 2019).

Similarly, species-specific or total bans are not always effective, because implementation at the point of catch depends on target species, gear and the species of management concern. In many fisheries, a certain

level of incidental shark catch is unavoidable, and sharks may already be dead or dying before release is feasible, rendering total bans biologically ineffective (Braccini & Waltrick, 2019; Gallagher et al., 2014; Tolotti et al., 2015). This is particularly problematic for highly mobile pelagic species which exhibit ram ventilation, such as scalloped hammerhead sharks (*Sphyrna lewini*), spinner sharks (*Carcharhinus brevipinna*), mako sharks (*Isurus* spp.) and thresher sharks (*Alopias* spp.). These species are commonly caught as incidental catch in longline and purse seine fisheries, and have very high levels of post-capture mortality (Braccini & Waltrick, 2019; Gallagher et al., 2014). As such, blanket bans need to be accompanied with practical fisheries management measures that effectively avoid or minimise capture, or promote live release. In general, ‘one-size-fits-all’ approaches, which apply one set of prescribed rules to sharks as a homogenous group, are of limited effect (Dulvy et al., 2017; Shiffman & Hammerschlag, 2016b). Differences in life history strategies and susceptibility to fishing will influence the effectiveness of different management strategies for different species (Braccini & Waltrick, 2019; Harry et al., 2011; Yates et al., 2016). The consequences of ineffective measures can also have wider-reaching impacts on management, through affecting perceived legitimacy and the likelihood of uptake by fishers (Hall *et al.*, 2007, See Assumption 3, Section 4.3).

Prescriptive technical measures also fail to consider that the effectiveness of a measure will vary based on fine-scale biophysical characteristics within a fishery or fishing trip, such as temperature, season and time of day (Maxwell *et al.*, 2015). Research has shown that the effectiveness of some technical measures (e.g., leader lines and hook types) varies in space, time and under different operational and environmental circumstances, as well for different shark species (Branstetter & Musick, 1993; Bromhead et al., 2012; Cooke & Suski, 2004; Serafy et al., 2012). In many cases, there is a need for dynamic decision-making at sea, based on prevailing biophysical conditions (Maxwell *et al.*, 2015). Therefore, if appropriately incentivised, fishers themselves may have better information for making optimal, adaptive decisions, rather than behaviour being prescribed (Hall *et al.*, 2007).

Unintended consequences

There are also examples where mandated technical measures have unintentionally increased levels of bycatch for the species they are attempting to protect (Sarmiento, 2006) or for other species of conservation concern (Baum et al., 2003; Weimerskirch et al., 1999). Unintended consequences can also occur at the macro-economic level, with bans creating black markets, and in some cases stimulating demand for more rare, luxurious and high-price commodities. Effectiveness depends on monitoring and enforcement capacity, as well as the nature of demand in consumer markets (Challender et al., 2015b; Courchamp et al., 2006; R. J. Hall et al., 2008).

2.3.2 Assumption 2: Capacity and motivation for adoption

Mandated measures assume that fishers, fishing fleets and industry have sufficient capacity and motivation to adopt them (Assumption 2). That is, they are willing and able to change fishing behaviour and decision-making to uptake measures and comply with rules. However, if technical measures for shark management are to be adopted in practice, they need to be appropriately incentivised, either positively (e.g., through economic benefits) or negatively (e.g., through enforcement and putative action), with efforts to ensure that such measures are as cost-effective as possible (Gjertsen et al., 2014; M. A. Hall et al., 2007; Hilborn et al., 2005). These factors are rarely considered in contemporary shark management design, or indeed in bycatch reduction research more broadly (Campbell and Cornwell, 2008). Yet failure to consider them can result in unacceptable implementation costs and negative socio-economic impacts to fishers and fishing fleets (Innes & Pascoe, 2008; Jaiteh et al., 2016; O’Keefe et al., 2014; Rausser et al., 2009). In turn, this can lead to a lack of compliance and implementation failure (Fulton et al., 2011; Gezelius, 2002).

Unrealised benefits

In bycatch reduction literature, positive economic incentives are believed to arise through a range of efficiencies (Campbell and Cornwell 2008). Examples of efficiencies include less time sorting unwanted or low value catch (Broadhurst, 2000; Fonseca, Campos, Larsen, Borges, and Erzini, 2005) and less damage to gear (Bache, 2003; Brewer et al., 1998). Economic benefits may include higher total catch value/catch per unit effort because bait, space and trips are not taken up by non-target catch (Fonseca et al., 2005; Gilman et al., 2003, 2006), or potential for higher sales value through marketing eco-friendly seafood (Bache, 2000; Gilman et al., 2005). However, the benefits of technical measures demonstrated in theory or under research conditions may not be replicated in practice. For example, bycatch reduction devices can be cumbersome, difficult to introduce and operate, and may malfunction or be costly to maintain (Campbell & Cornwell, 2008; M. A. Hall et al., 2007; Kaplan et al., 2007). What is more, benefits may be captured further up the supply chain, by boat owners or investors, as opposed to the fishers implementing the measures. These benefits are even harder to realise for sharks, since sharks are often valuable, marketable catch.

Hidden costs

As well as unrealised benefits some measures may be unacceptably costly to implement, due to foregone catches and revenues. For example, introduction of bycatch reduction technologies (BRTs) in the Gulf of Mexico shrimp fishery resulted in significant shrimp loss (Margavio & Forsyth, 1996). Similarly, input controls in the Hawaiian longline fishery reduced bycatch, but also caused significant reduction in catch rates for tunas and several other commercial species (Gilman et al., 2007). Such opportunity costs are particularly relevant for shark management, where species of conservation concern may have a high market value. For example, a semi-commercial pelagic shark fishery in Indonesia takes a mixture of

species, which include species of low fecundity and international management concern such as hammerheads (*Sphyrna* spp.) and silky sharks (*Carcharhinus falciformis*), and species with higher productivity such as blue sharks (*Prionace glauca*), milk sharks (*Rhizoprionodon acutus*) and dogfish (*Squalidae* spp.) (Yulianto et al., 2018). While it may be desirable from a conservation perspective to reduce catch of hammerhead sharks and silky sharks, these species are also some of the most economically valuable in the fishery (Lestari et al., 2016). Limiting catch of these species would result in a significant decline in total catch value and household income for fishers and traders. Similarly, even species reportedly taken as bycatch in non-target fisheries represent considerable economic value. For example, several small-scale coastal gill net fisheries in Indonesia, which target shrimp and small demersal teleost, also incidentally take wedgefishes (*Rhinidae* spp.) (Simeon et al., 2020). Yet despite being a small volume of the total catch, wedgefishes can make up a significant proportion of total catch value, since their market value is high relative to other species (Hau et al., 2018).

In the absence of market-based incentives for sustainable management, or alternative sustainable income streams, management can lead to unacceptable negative consequences. These may be socio-economic, in terms of reduced income, employment, and food security; or ecological, with displacement of effort towards other vulnerable or overexploited species and stocks. For example, area-based restrictions and declining fin prices in Eastern Indonesia reportedly displaced small-scale fishers, and drove uptake of risky, illegal income generation activities, such as people smuggling (Jaiteh et al., 2016). While a ban on manta ray fishing resulted in a three-fold increase in devil ray catch in one fishery in Indonesia, due to a shift in effort to non-protected species (Chapter 4; Booth, Mardiah, et al., 2020). It is also plausible that regulation-induced declines in market value for silky sharks (*Carcharhinus falciformis*), hammerhead sharks (*Sphyrna* spp.) and other CITES-listed species could drive an increase in shark fishing pressure to replace lost economic value. For example, in a socio-economic survey of shark fishers in Tanjung Luar, Indonesia, 75% of fishers stated that they would continue to fish as normal or increase their fishing effort should their shark catch decline, rather than change target species or livelihood (Lestari et al., 2016). Other intangible costs, such as a loss of cultural values, may also be common (see Assumption 3, section 4.3).

Identifying and understanding the costs of shark management is further complicated by the mixed capture of multiple species, the fuzzy distinction between target and bycatch species in small-scale tropical fisheries, and the fluid and opportunistic nature of fishing within the broader livelihood strategies of rural coastal communities (Allison & Ellis, 2001; Bene, 2006; C. Carter & Garaway, 2014). Many small-scale fishing communities already face structural poverty and vulnerability to shocks, with instable income and high reliance on marine resources for nutrition and food security (Golden et al., 2016). In these communities sharks are not only caught to generate income, but are also caught for food. Sharks can provide an important source of animal protein and micronutrients, particularly as catches in traditional

food fishes decline (Golden et al., 2016; Glaus et al., 2018). Fishing may therefore serve an important welfare function, creating a labour buffer or safety net for structurally poor or vulnerable households (Bene, 2006; Jul-Larsen et al., 2003). Therefore, some costs of shark management may be hidden or intangible, such as increased vulnerability to shocks or reduced access to micronutrients. These costs may also disproportionately affect poorer households. If predicated on an incomplete understanding of livelihoods and the pro-poor functions of small-scale fisheries, management measures may be incompatible with both conservation and the social and economic goals of fisheries management (Allison and Ellis, 2001). This underlines the practical and ethical impetus to consider the direct and indirect opportunity costs to fishers when designing management approaches.

The limitations of enforcement

Incentives may also be negative, due to the costs – theoretical, actual and perceived – of enforcement. When technical measures are mandated, enforcement is assumed to incentivise uptake through avoidance of putative action. There is empirical evidence that risk of enforcement plays some role in shaping fisher behaviour (Arias et al., 2015). However, little attention has been paid to what kinds of regulations produce economic incentives for uptake, the investments required in monitoring and enforcement to ensure compliance (and whether these are realistic, given budgetary constraints), and in what ways they function in the contexts of different shark fisheries.

Economic models theorise that the cost of enforcement is a function of probability of an act of non-compliance being detected and punished, and the severity of punishment that results (Becker, 1968). This suggests that penalties must at least balance the illegal gains from catch, the threat of enforcement must be credible, and that cost-effective monitoring information is available for detecting non-compliance. However, shark catch can be highly valuable (e.g., Hau et al., 2018), penalties in fisheries law can be weak or non-existent (Sumaila et al., 2006), and managers and other fishers may be reluctant to deliver sanctions against non-compliant fishers for social or cultural reasons (Gezelius, 2002). Fisheries enforcement often fails in practice because of low detection probabilities in extensive and remote fishing grounds, which are monitored by enforcement agencies with limited resources (Gilman et al., 2003). This makes marine areas vulnerable to illegal fishing by international vessels (e.g., Reuters World News, 2017). Enforceability challenges are also exacerbated in small-scale fisheries, which are ubiquitous in the coastal waters of lower-income countries, and almost entirely unregistered and unmonitored. Regulatory action in these contexts is further complicated by the socio-economic vulnerability of fishers, with ethical concerns and limited political will to strictly enforce laws.

Even in a world of high detection probabilities and severe penalties, the costs of enforcement may fail to incentivise sustainability or change behaviour in the desired way. Fishers may respond by taking measures to avoid putative enforcement action rather than to fish more sustainably. For example, mandated

bycatch reduction technology (BRT) in a shrimp fishery in Texas led to fishers attempting to ‘beat the system’ by tying off their BRTs in the water, looking for loopholes in the regulations and simply not employing BRTs until caught without them (Jenkins, 2006). These situations can create costly ‘arms races’ between enforcement agencies and fishers (Campbell and Cornwell, 2008).

Overall, the negative incentives created by enforcement can support uptake of technical measures, but only when the probability and costs of being caught are high, and even then, only to a certain point (Arias, 2015; Jenkins, 2006). In reality, compliance with marine and fisheries regulations are shaped by a range of individual and social factors beyond economic costs and benefits, including knowledge, perceptions, social networks, perceived legitimacy and governance structures (Arias et al., 2015; Bergseth & Roscher, 2018; Oyanedel, Gelcich, & Milner-Gulland, 2020) (See Assumption 3, Section 4.3). As such, the success of enforcement will be influenced by the specifics of the fishery, the measure being regulated, and the socio-economic context.

Ultimately, the extrinsic incentives for adopting a fisheries management measure will depend on a complicated balance of costs and benefits. These include: the benefits arising through catch efficiencies and market-based rewards, the fixed and variable economic costs of adopting and maintaining a technical measure, the opportunity cost of lost valuable catch, and the risk and cost of enforcement. Hidden or intangible costs may also arise, and adoption of measures may be strongly influenced by intrinsic motivations and social influences (See Assumption 3, Section 4.3).

2.3.3 Assumption 3: Economic incentives are sufficient

Finally, even in cases where prescribed technical measures are seemingly effective and sufficiently incentivised, they may not be widely implemented (Damalas & Vassilopoulou, 2013; Orphanides & Palka, 2013; Radzio et al., 2011). As such, even if they do exist, shark fishers may not respond to incentives by reducing catch (Assumption 3). This is because economic models of how people make decisions are unrealistic – “*Individuals may have bounded rationality, limited by cognitive resources, and employ a variety of heuristic procedures to achieve outcomes that are ‘good enough’ rather than truly optimal*” (Conlisk, 1996). A range of emotional, social, cultural and cognitive biases shape people’s conservation-relevant decisions (Cinner, 2018), thus influencing uptake of technical measures and compliance with regulations. What is more, extrinsic incentives can have complex interactions with social norms and intrinsic motivations (Gneezy et al., 2011). As such, introducing extrinsic motivations in an unsuitable social context can create conflicts between different types of motivations. This can lead to unexpected or unintended impacts on behaviour. For example, economic incentives can crowd-in or crowd-out intrinsic motivations for prosocial behaviour, or damage trust and institutions (Bowles & Polanía-Reyes, 2012; Cinner et al., 2021; Gneezy et al., 2011; Grillos et al., 2019). Understanding the decision-making context is therefore crucial for

designing suitable management interventions, which can effectively modify fisher behaviour in the desired direction, improve management outcomes and reduce regulatory costs (Grafton, 2005).

Cognitive biases

Lessons from the field of behavioural economics indicate that that responses to incentives are shaped by mental heuristics, such as loss aversion, as opposed to rational costs-benefit calculations. Therefore, the framing of an issue or incentive can be more important than its absolute magnitude (Cinner, 2018; Hossain & List, 2012). Symbolic or social rewards may also be more effective and efficient at encouraging a desired behaviour than direct economic incentives, particularly in a public goods or social context (Gallus, 2017; Pentland, 2014). People often act in ways that are shaped by sub-conscious cues, such as emotional associations, ego, priming or anchoring. Decisions are also strongly influenced by social context, such as who communicates information to them (e.g., trusted messengers and block leaders), what they normally do (e.g., the status quo bias), what most people do (e.g., peer pressure and social norms), and what other people see (e.g., observability) and what other people expect of them (e.g., public commitments, reputation and recognition) (Abrahamse & Steg, 2013; Cinner, 2018; Gallus, 2017; Mbaru & Barnes, 2017; Thaler, 2018).

Social influences

Research into the social aspects of fisheries management has shown that social networks; trust and social capital; local leadership and role models; governance and institutional structures; social norms and peer pressure; perceived legitimacy of regulations; perceived effectiveness of proposed measures; and even the skill, experience and motivation of individual fishers and captains shape uptake of technical measures (M. L. Barnes et al., 2016; Cinner et al., 2016; Gutiérrez et al., 2011; M. A. Hall et al., 2007; Mbaru & Barnes, 2017). Beyond the natural resource management literature there is also a broad body of research on how new ideas and technologies spread, which indicate that innovation (i.e., in this case, adopting new management measures or fishing practices) depends heavily on communication channels and social networks (Dearing, 2009; Rogers, 2003).

For example, social networks have been identified as a key factor in shaping uptake of shark bycatch mitigation measures in Hawaii's tuna longline fishery (M. L. Barnes et al., 2016). While in Indonesia, many shark fishers inherit their gears and fishing practices from their fathers and grandfathers, and take considerable pride in their way of life (Lestari et al., 2016). As such, adopting shark management measure may violate social and cultural norms, which can lead to widespread non-compliance (e.g., Gezelius, 2002). Similarly, Margavio and Forsyth (1996) described how resistance of shrimp fishers to mandated BRTs in Louisiana, USA was a manifestation of defence of traditional cultural practices, fear of eroding independence, and anger at the marginalisation of shrimping in the face of competing economic activities. These issues are analogous to those documented in the human-wildlife conflict literature, where social

factors, intangible costs or underlying human-human conflicts may be more important for effectively resolving conflict than technical measures (Dickman, 2010; Redpath et al., 2013; Thirgood & Redpath, 2008).

In identifying opportunities for engaging people in land-based conservation, Knight et al. (2010) and Selinske et al. (2015) also found that human and social capital defined people's willingness to engage. The most salient factors included conservation knowledge, entrepreneurial orientation, local sense of belonging or attachment, confidence in governance, local networks, willingness to collaborate and social learning (Knight et al., 2010; Selinske et al., 2015). Human capital has been found to be similarly important for successful fisheries management, particularly leadership and social cohesion (Gutiérrez et al., 2011). Fishers and skippers also differ in their knowledge, experience, risk tolerance, and ability or willingness to adjust. As such, imposing the same standard on all vessels does not necessarily achieve optimal management goals in an efficient, least-cost manner (Hall et al., 2007; Squires and Garcia, 2018). Management measures that acknowledge and capitalise on this heterogeneity have a greater chance of being accepted, and achieving socially efficient outcomes (Hall et al., 2007; Knight et al., 2010; Squires and Garcia, 2018). Given their large repository of practical knowledge and experience, fishers themselves may be better placed to make at-sea decisions to avoid shark capture, as opposed to policy-makers (Hall et al., 2007).

Similarly, the perceived legitimacy of a rule, in terms of its effectiveness, justness and confidence in regulating institutions, can affect uptake and compliance (M. A. Hall et al., 2007; Levi et al., 2009; McClanahan et al., 2006; Oyanedel, Gelcich, & Milner-Gulland, 2020). Lessons from bycatch mitigation efforts for other species indicate that fishers need to understand the importance of the management problem, and believe that proposed solutions are effective (Hall et al., 2007). Failing to recognise fisher knowledge or getting a technical measure wrong may therefore damage perceived legitimacy of a regulation or regulating institution, and negatively impact management efforts.

In addition, local institutional context and tenure regimes influence the success of fisheries management (Hilborn et al., 2005). Community-based management interventions that engage with local or traditional institutions, build upon cultural values, provide rewards and equitable benefit distribution, and provide opportunities for social learning are more likely to succeed (Brooks et al., 2012; Hilborn et al., 2005; Oldekop et al., 2010; Waylen et al., 2010). Compliance can also emerge and persist through group dynamics if individuals cooperate and enforce rules by social pressure (Fehr & Gächter, 2002; Fowler, 2005); or can break down where rules do not align with the social norms of the group (Gezelius, 2002). As such, novel policy instruments that can foster peer pressure and group-level cooperation may be more efficient and effective than direct regulation and enforcement (Fehr & Gächter, 2002; Gezelius, 2002; Keane et al., 2008; Kotchen & Segerson, 2020). What is more, since social context is dynamic, factors that

encourage initial uptake of management measures may be different from those that maintain use and engagement in the long-term (Selinske et al., 2015).

Social-physiological models

Social-psychological models of human behaviour consider that a combination of behavioural beliefs, normative beliefs, and perceived behavioural control are crucial in shaping behavioural intentions (Ajzen, 1991; Fishbein & Ajzen, 1975). Behavioural beliefs are based on the evaluation of a likely outcome of a behaviour, while normative beliefs are based on perceptions about how others will judge a behaviour. Perceived behavioural control relates to perceptions of self-efficacy and autonomy regarding a behaviour. The resulting behavioural intention is in turn moderated by intervening factors, which may create barriers to a behaviour even when an intention exists. Individuals may also have multiple evaluations of a behaviour, some of which will be more salient than others (the Theory of Reasoned Action and the Theory of Planned Behaviour (Ajzen, 1991; Fishbein & Ajzen, 1975), see also norm activation theory, social norm, theory and self-determination theory). These models recognise that a combination of instrumental and normative, and extrinsic and intrinsic factors will shape behavioural intentions and outcomes (Deci & Ryan, 1985; Ryan & Deci, 2002).

In addition to theory, social psychology methods have been applied to conservation planning. For example, psychometric surveys have been used to understand motivations of individual resources users at the local level (Knight et al., 2010; Selinske et al., 2015, 2019), and to design and tailor policies and instruments, such as financial incentives, to meet diverse motivations of individual resources users (Selinske et al., 2017).

2.3.4 The socio-economic implementation gap

Overall, I have demonstrated that managing shark fishing is much more than a biological and technical issue: it is also a human issue. There are practical and ethical imperatives to consider the human dimensions of shark conservation, which echo previous calls for research into the social and economic aspects of shark fisheries (Dharmadi et al., 2015; Mizrahi et al., 2019; Simpfendorfer et al., 2011). What is more, socio-economic issues may be even more relevant to shark conservation than many other fields, due to the mixed fisheries, diverse contexts, conflicting human uses and values, and complex supply chains, which play a role in food and livelihood security in vulnerable coastal communities. The need to consider human issues is not new to conservation, yet it has been neglected in shark science and management (Dulvy et al., 2017; MacKeracher et al., 2018; Simpfendorfer et al., 2011). I argue this is creating a socio-economic implementation gap (Figure 2.2), which hinders effective management. This gap is particularly problematic for lower-income countries that are dominated by small-scale fisheries.

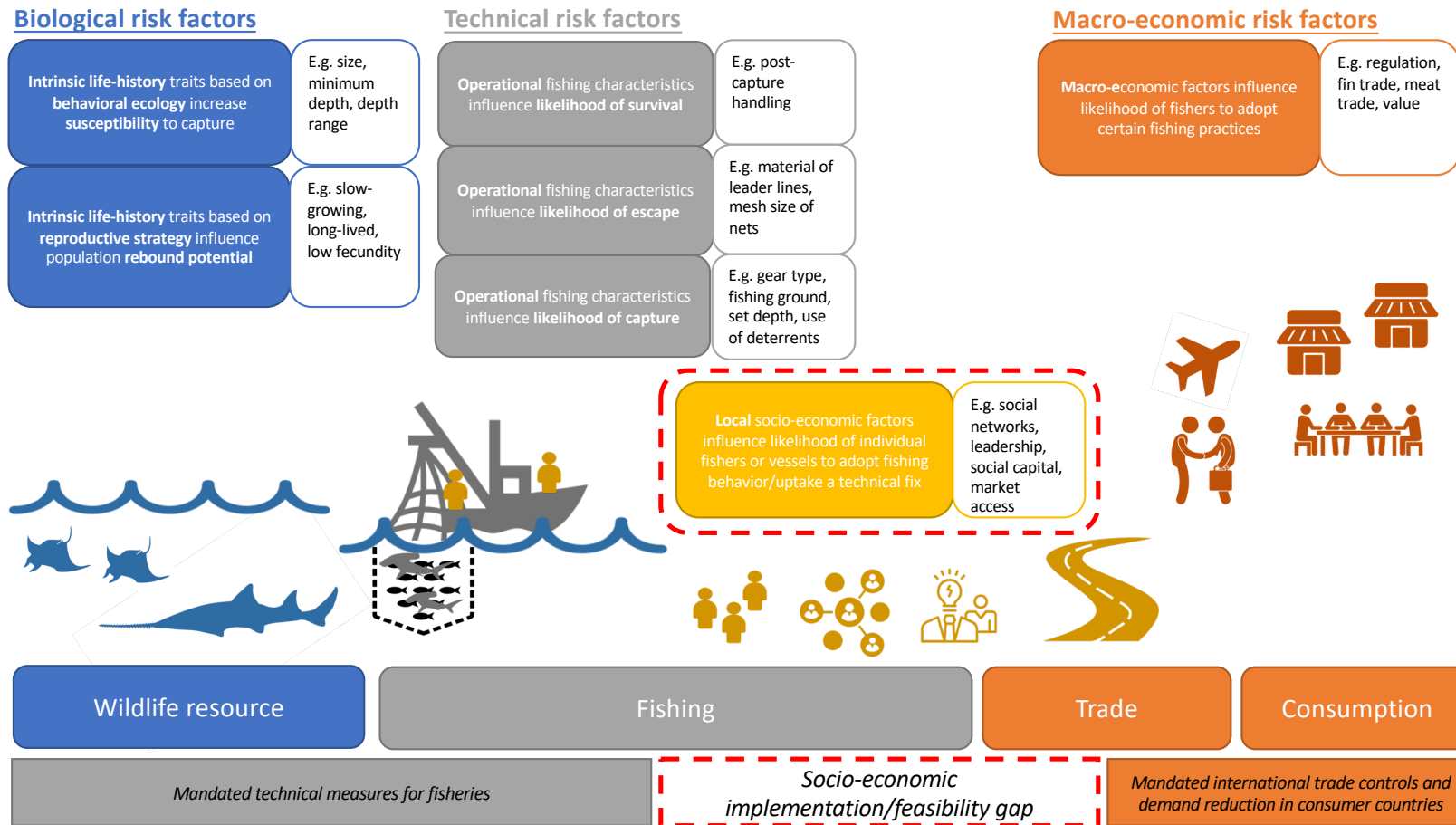


Figure 2.2 A conceptual diagram of all risk factors to sharks throughout the shark trade chain, with highlighted gaps in management related to local-/ micro-scale socio-economic factors

The complexities discussed here demonstrate that fisheries need to be managed within their specific ecological, economic and social contexts. Effective management requires a complementary mix of policies and instruments, which seek to converge the behavioural motivations and welfare of fishers, with conservation objectives (Brady & Waldo, 2009; Fulton et al., 2011). These policies and instruments must also be consistent with cost-effective monitoring and enforcement. Accordingly, there is a need to differentiate between different fishery types and the primary drivers of shark fishing mortality in each fishery, when making management decisions. For example, differentiating between industrial-scale fishing for profit, small-scale commercial fishing for food and profit, and subsistence fishing for food only; as well as between fisheries that take sharks as primary catch, valuable secondary fishing, or true bycatch. Understanding these drivers will be critical for designing management measures that are effective at reducing shark fishing mortality, whilst appropriately considering the needs and capacities of people (Barker & Schluessel, 2005; Dharmadi et al., 2015; Glaus et al., 2018). Further, an increased understanding of the attitudes, norms and underlying motivations of fishers, and their interactions and dynamics as a group, is needed to design policy instruments that can effectively change fishing behaviour (Ajzen, 1991; Battista et al., 2018; Milner-Gulland et al., 2018; Stern, 2018). The heterogeneity, dynamism and stochasticity of these socio-economic contexts implies that simple, generalizable solutions may be elusive, rather, diverse measures are required that are adequate for local socio-economic and institutional realities (Waylen et al., 2010).

Moving forwards, achieving reductions in shark fishing mortality will require researchers and practitioners to take a more holistic approach to risk-based management and decision-making. Such an approach should consider not only the biological and technical aspects of species and fisheries, but also the feasibility of management actions, given the socio-economic context (Figure 2.2). Explicit assessment of feasibility can support design of management measures, and identification of complementary policies and instruments that are tailored towards to places and people (Davidson & Dulvy, 2017; Dickman et al., 2015). This can help to ensure management measures are effective and ethical, and thus overcome the socio-economic implementation gap (Ostrom et al., 2007, Knight et al., 2010).

A holistic approach to risk will be particularly important for delivering shark conservation outcomes in lower-income countries, where shark fisheries management measures will need to address multiple objectives and manage difficult trade-offs. These competing objectives may include: protection of the most vulnerable shark species, sustainable offtake of co-occurring species and populations that can withstand it (shark and non-shark), and maintenance of the livelihoods and well-being of vulnerable coastal communities.

2.4 Discussion

Going forwards, I propose several priorities for bridging the socio-economic implementation gap for shark management. In particular, I propose that feasibility assessments be explicitly incorporated into fisheries risk assessments and decision-making processes. To support this, a deeper understanding of the human dimensions of shark fisheries is required, alongside holistic management frameworks that support the integration of socio-economic considerations from planning to implementation to monitoring.

2.4.1 Feasibility assessments

A holistic approach to understanding and managing shark fisheries requires that socio-economic factors and implementation costs be integrated throughout risk assessment processes. This is not a new concept in conservation. There is a substantial body of literature on the importance of incorporating cost and feasibility in to systematic conservation planning (Ban et al., 2009; Ban & Klein, 2009; Zhang & Vincent, 2019), and on evaluating the impacts of conservation interventions on human well-being (Bull et al., 2018; Milner-Gulland et al., 2014; Woodhouse et al., 2015). Yet these concepts are yet to be adopted and adapted to shark fisheries management. To do so, I suggest the addition of a new dimension to traditional risk assessments for sharks: feasibility (Figure 2.3). While these considerations may add an additional layer of complexity to an already complex problem (Dulvy et al., 2017), various established methods for the integration of socio-economic variables in to decision-making and management could be adapted for this purpose (Álvarez-Romero et al., 2018). For example, Davidson and Dulvy (2017) used a national-level conservation likelihood score for prioritising shark management needs for different countries (Davidson & Dulvy, 2017; Dulvy et al., 2017), while Knight et al. (2010) used five dimensions of conservation opportunity to schedule conservation action at the ecosystem-level.

Building on this, I propose six dimensions of feasibility for shark management. These include economics and well-being, which are associated with the monetary and non-monetary costs of shark management; while human capital, social capital, regulation and governance, and human pressure relate to the broader enabling environment. Each dimension may have multiple constituent components, which in turn can have positive or negative impacts on feasibility, depending on the context (Table 2.2). Economics includes the direct economic costs and benefits of adopting a management measure, in terms of losses or gains in catch and associated income, and the costs of putative action through enforcement. Well-being includes broader costs and benefits to people beyond changes in income, such as basic needs (e.g., food security, employment security, access to services); agency (e.g., participation in decision-making); and experienced quality of life (e.g., ability to pursue goals) (Bull et al., 2018). Human capital includes knowledge, skills and experience of fisher communities, and their access to technology and tools, which influence the capacity to uptake a management measure. Social capital includes social influences that may enable or disable implementation, such as social networks, leadership, local institutions, willingness to collaborate, peer pressure, public perceptions and trust. Regulation and governance include factors within

the broader regulatory context, such as policy frameworks to protect species or control trade, and how well these are implemented, through government effectiveness and rule of law (Table 2.2). Finally, human pressure relates to broader scale market and subsistence pressures on a fishery, such as gravity of human impacts, based on human population and travel time (Cinner *et al.*, 2018). This list is not necessarily exhaustive, and the most important factors would need to be identified and assessed based on the context of a given fishery. For example, factors such as food security, livelihoods, poverty and corruption will be less important in high-value commercial fisheries in wealthy, politically stable developed countries such as Australia and the USA, while they may be critical in defining the effectiveness of a management measure in a small-scale coastal fishery in a developing country such as India or Indonesia.

Once key feasibility issues are identified, quantitative or semi-quantitative assessments of the risk they pose could be conducted to determine a feasibility score. This information can then be used to supplement traditional risk assessments, for example through adding a third ‘feasibility’ dimension to traditional fisheries risk assessments, such as Productivity-Susceptibility Analyses (PSA) (Figure 2.3). In some cases, it may be challenging to gather quantitative data on all these factors, and the magnitude of the risk they pose. As such, a ranking, scoring or categorisation system could be adopted based on informed judgement or expert elicitation. These methods are commonly used for ecological risk assessments in data-poor contexts (Beauvais *et al.*, 2018; Mace *et al.*, 2008), and are already used for semi-quantitative biological and technical risk scores in PSAs for sharks, e.g., through 1-3 or high-to-low scoring systems (Hobday *et al.*, 2007; Gallagher *et al.*, 2012).

Overall, the adoption of feasibility and its explicit assessment alongside biological and technical factors would enable a more holistic understanding of risks to sharks in fisheries. This would build on a substantial body of work to systematically include socio-economic factors in conservation planning (Álvarez-Romero *et al.*, 2018; Knight *et al.*, 2010; Polasky, 2008), and address recent calls to include local people in conservation planning for sharks (MacKeracher *et al.*, 2018; Rigby, Appleyard, *et al.*, 2019).

Table 2.2 Proposed dimensions of feasibility of shark management measures, based on socio-economic factors that influence pressures and uptake of management measures in fisheries

Costs and benefits				Enabling or disabling environment											
Economics		Well-being		Human capital		Social capital		Regulation and governance		Human pressure					
Definition		Economic gains or losses of adopting management measures		Well-being gains or losses of adopting management measure		Knowledge, skills, and experience of fisher community, and availability of tools/technologies		Networks, relationships and cohesion within fisher community		Policies, rules, official or socio-cultural institutions, and their implementation		Broader scale market and subsistence pressures			
Examples		Positive impact on feasibility		- Increase in target catch - Increased value of target catch - Avoidance of costly putative measures - Lower operational costs/operational efficiencies - Incentives		- Increase in food security - Increase in employment security - Increase in access to other services		- Conservation values - Desire to learn		- Social networks - Leadership - Institutions - Public perceptions of conservation - Peer pressure to comply/not comply		- Higher-level policy frameworks in place for marine management. - Government resources and effectiveness - Political stability - Rule of law		- Remote locations with low human pressure	
		Negative impact on feasibility		- Reduction in target catch or other marketable species - Operational inefficiencies		- Lower food and livelihood security - Loss of cultural values - Reduced freedom of choice/agency - Loss of social cohesion		- Lack of skills - Limited adaptive capacity		- Trust and confidence in authorities - Willingness to collaborate		- Conflict - Corruption		- Markets - Human population - Accessibility/travel time	
Key References		Hall et al., 2007; Campbell and Cornwell, 2008; Dickman et al., 2011; Fulton et al., 2011; Gutiérrez et al., 2011.		Hall <i>et al.</i> , 2007; Campbell and Cornwell, 2008; Jaiteh <i>et al.</i> , 2016, 2017; Glaus <i>et al.</i> , 2018.		Hall <i>et al.</i> , 2007; Knight <i>et al.</i> , 2010; Selinske <i>et al.</i> , 2015, 2019.		Knight <i>et al.</i> , 2010; Oldekop <i>et al.</i> , 2010; Waylen <i>et al.</i> , 2010; Gutiérrez <i>et al.</i> , 2011; Barnes <i>et al.</i> , 2016; Cinner, 2018.		Dickman <i>et al.</i> , 2011; Cinner <i>et al.</i> , 2016; Davidson and Dulvy, 2017.		Dickman <i>et al.</i> , 2011; Edgar <i>et al.</i> , 2014; Davidson <i>et al.</i> , 2016; Cinner <i>et al.</i> , 2018.			

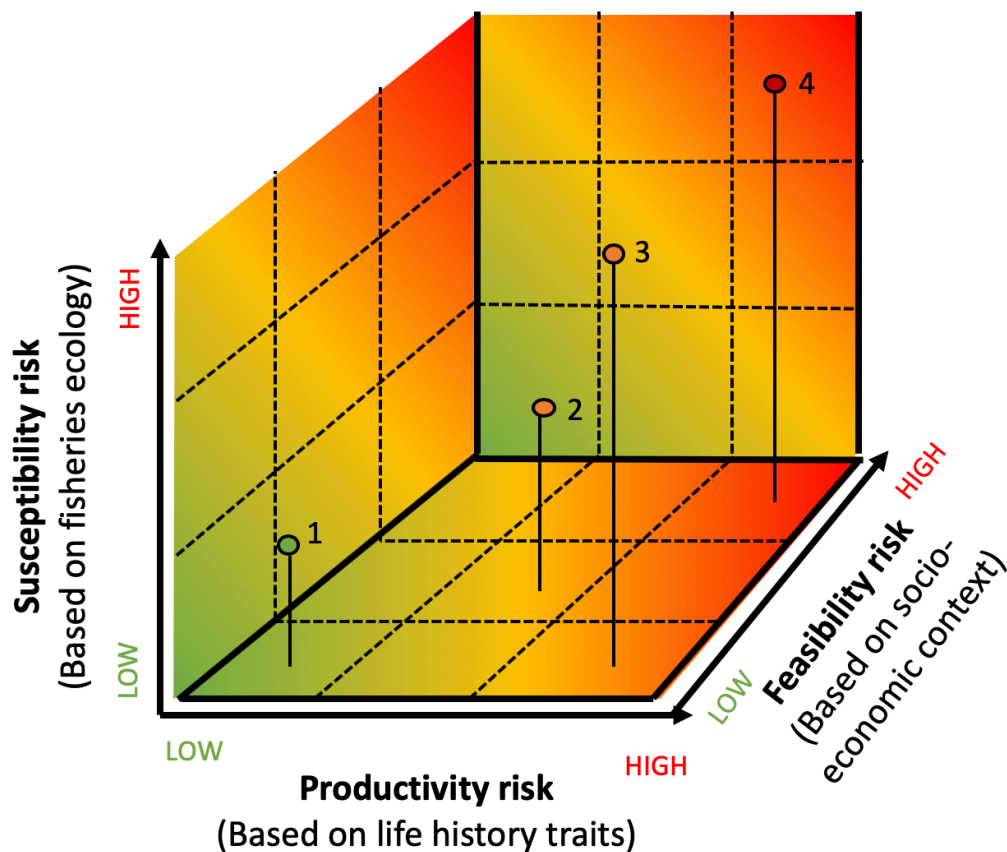


Figure 2.3 Schematic of a common productivity-susceptibility analysis (PSA) plot used in semi-quantitative ecological risk assessments for fisheries, with the added feasibility dimension. Point number 1 is an example of a low risk susceptibility-productivity-feasibility combination, such as the US Atlantic spiny dogfish population, while point 4 is an example of high risk across all dimensions, such as small-scale targeted fisheries for wedgefish and hammerhead sharks in Indonesia. Point 2 is moderate in all dimensions, while point 3 is high technical risk and low feasibility risk, which could represent the risks to blue sharks (2) and bigeye thresher sharks (3) caught as bycatch in high-value commercial tuna longlines.

2.4.2 Understanding the human dimensions of shark fisheries

Adopting feasibility assessments as part of shark fisheries management frameworks also requires a more in-depth understanding of the human dimensions of shark fisheries. Substantial gaps remain in the understanding of the local socio-economic factors that influence shark fishing behaviour, and how these interact with other risk factors, such as fishing technology and macro-level policy. To better manage fisheries and inform policy in the future, it would be informative to conduct detailed analyses of the drivers of shark fishing in different fishery contexts, including the relative importance of technical factors vs. social factors, and the degree to which global trade in shark-derived products drives local-level fishing behaviour and fishing mortality. This could be supported through socio-economic surveys (e.g., Lestari et al., 2016; Glaus et al., 2018) or psychometric methods (e.g., Selinske et al., 2015). Based on this understanding, cost-effectiveness analyses (Gjertsen et al., 2014; Wilcox & Donlan, 2007) and

participatory predictive methods (such as scenario analysis, experimental games or choice experiments) could be used to investigate the potential effectiveness and social acceptability of a management intervention (Moro et al., 2013; Travers et al., 2011; Travers, Selinske, et al., 2019). These approaches could help to provide quantitative scorings and weightings to feasibility assessments, and ultimately determine which management measures are likely to be most effective, acceptable and ethical, for both sharks and people.

2.4.3 Holistic management frameworks

Further, socio-economic factors need to be considered beyond the risk assessment phase, and systematically incorporated to prioritisation, decision-making, policy design, implementation and evaluation. The information gathered for feasibility assessments could support the setting of realistic management goals at the fishery level, which consider the constraints of the broader regulatory, cultural and economic conditions of a fishery. This can enable trade-offs between shark conservation objectives and socio-economic fisheries objectives to be made explicit when designing management measures. In turn, acknowledging trade-offs can encourage creative thinking regarding optimal mixes of policies and instruments. This can aid the design of policies which reduce costs, capitalise on heterogeneity in attitudes and motivations, and ultimately encourage wider compliance and cost-effective enforcement (Hall et al., 2007; Selinske et al., 2016). Management goals can also explicitly include socio-economic objectives and constraints, such as minimising cost or maintaining well-being of vulnerable fishers, to optimise outcomes for sharks and people. If quantitative targets are set as part of this process, the impacts of management on both sharks and people can be monitored and evaluated. In turn, this can support learning and adaptive management. This would be a valuable contribution to shark science, particularly if proofs of concept can be provided for effective management models in small-scale fishery developing country contexts.

Finally, given the magnitude of the shark conservation problem, and the nature of shared stocks and pressures from multiple fisheries, it is important to ensure that the tailored solutions advocated for here are not simply implemented through piecemeal projects. Management measures need to be adopted at scale, integrated into national-level plans and objectives, and contribute to the achievement of international biodiversity conservation goals such as those under CITES and the Convention on Biological Diversity (CBD). This necessitates over-arching frameworks that can integrate complex, multifarious fisheries and their diverse management goals to create net positive impact – i.e., healthy shark populations – at national- and global-scales. The mitigation hierarchy, for example, has been applied to achieving no net loss (NNL) of biodiversity in terrestrial ecosystems (Arlidge et al., 2018), and proposed as a step-wise precautionary approach for least-cost management of marine fisheries and bycatch mitigation (Milner-Gulland *et al.*, 2018; Squires and Garcia, 2018), with potential application to sharks (Milner-Gulland *et al.*, 2018). This could provide an overarching framework to set ambitious

management goals for sharks based on net impact. Thinking in net terms allows room for fishery-specific management, whilst ensuring aggregate impact at scale. Systematic assessment of the biological, technical and socio-economic dimensions of fisheries within this framework could support identification of national and international priorities and approaches for meeting management goals, by identifying the most problematic fisheries in terms of fishing mortality risk, and strategic leverage points for maximising conservation impact for sharks while minimising cost to people. This holistic approach could enable identification of feasible management measures, which facilitate the recovery and maintenance of healthy shark populations, whilst ensuring the socio-economic complexities of fisheries management are no longer neglected.

3. The mitigation hierarchy for sharks: a risk-based framework for reconciling trade-offs between shark conservation and fisheries objectives

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Photo: a woman in Tanjung Luar carrying a swordfish on her head.

“If you never miss a plane, you’re spending too much time in airports” – George Stigler, on risk.

3.1 Introduction

Fisheries management reforms for sharks need to be adapted to specific country and fishery contexts, so that they are effective at the local level (Chapter 2). Yet actions must also be scalable, to manage shark mortality at seascape, stock and global levels. This necessitates an overarching framework that can guide a coherent network of coordinated actions across multiple levels. Such a framework needs to incorporate the biological and operational complexities of shark fisheries (i.e., many species, mixed fisheries, multiple jurisdictions, compliance and enforcement challenges; Dulvy et al., 2017), and be capable of handling data paucity and uncertainty. Moreover, to support pragmatic policy making, management decision-making should consider socio-economic factors, budgetary constraints, and inevitable trade-offs between conservation objectives and human needs (e.g., food security, livelihoods, income). There is a need to think beyond silver-bullet technical solutions and direct regulation for shark conservation, towards creative approaches for feasible fisheries management, which can improve outcomes for sharks and people (Booth, Squires, & Milner-Gulland, 2019b; Dulvy et al., 2017; Shiffman & Hammerschlag, 2016a, 2016b). Sharks can also serve as a flagship species for improved fisheries management across the globe.

Acknowledging these challenges and opportunities, this chapter proposes the mitigation hierarchy (MH) as an overarching framework for holistic, risk-based fisheries management for sharks, which can be used to integrate technical and socio-economic considerations to guide multiple towards united biodiversity goals. The MH is a step-wise precautionary approach to reduce the impact of economic development activities on biodiversity (BBOP, 2012). It has been most commonly applied to development planning in terrestrial ecosystems, however it has recently been proposed as a framework for least-cost management of marine fisheries and bycatch mitigation (Milner-Gulland et al., 2018; Squires & Garcia, 2018). The MH has also been recommended as a global framework to mitigate all negative impacts of human activity on biodiversity, and implement the goal of No Net Loss (NNL) of biodiversity as part of the Convention on Biological Diversity's Post-2020 Global Biodiversity Framework (Arlidge et al., 2018; CBD, 2020a; Milner-Gulland, Addison, et al., 2020).

I build on efforts to translate the MH to marine fisheries (Milner-Gulland et al., 2018) and delve into the practical aspects of its application and operationalisation for sharks: a challenging species group in urgent need of better management. I develop a conceptual model for shark fishing mortality, which decomposes risk into several constituent elements. I propose a process for using the MH to make transparent, goal-oriented, data-driven management decisions for reducing these risks. To illustrate its utility, I explore how the process could be applied to a range of different species and contexts using examples from real-world fisheries. In doing so, I outline how existing shark management measures correspond to different stages of the MH, and how existing knowledge on the effectiveness of these measures can be synthesised to make informed management decisions. I also explore practical challenges in applying the MH to sharks, and offer workable solutions and priorities for future research. Overall, I demonstrate how the MH can

help to reconcile trade-offs between shark conservation goals and the important role of fisheries in national economies and coastal livelihoods

3.2 The mitigation hierarchy for sharks

The mitigation hierarchy (MH) is a risk-based precautionary approach for limiting the negative impacts of human activities on biodiversity (Arlidge et al., 2018). The MH was designed for infrastructure development projects in terrestrial ecosystems with effectively irreversible impacts (e.g., housing developments, roads, plantations). It is increasingly incorporated in to infrastructure planning policy, and is most commonly applied as part of Environmental Impact Assessments (EIAs), which seek to assess the environmental consequences of plans or projects prior to their implementation (G. Bennett et al., 2017).

The MH typically proceeds in four sequential steps: (1) avoid, (2) minimise, (3) remediate and (4) compensate. The first step involves avoiding negative impacts on biodiversity from the outset, such as setting damaging human activities away from biodiversity hotspots or critical habitat. The second step requires that the extent of the negative impacts on biodiversity are minimised whilst the damaging activity occurs. The third step involves remediating negative impacts on biodiversity within the footprint of the damaging activity. The final step requires that any residual negative impacts are compensated for, through off-site conservation actions which improve the status of the affected biodiversity elsewhere (Arlidge et al., 2018; CSBI, 2015; Milner-Gulland et al., 2018). If applied successfully, the MH can lead to no net loss (NNL) of biodiversity or even net gain (BBOP, 2012; Bull et al., 2013; Gardner et al., 2013; Milner-Gulland et al., 2018; zu Ermgassen et al., 2019). For example, wetland mitigation banks in the United States have shown to successfully achieve NNL of wetland area through protection, restoration or creation of wetlands in compensation for loss caused by development projects (Brown & Lant, 1999; zu Ermgassen et al., 2019).

Recently, the MH has been proposed as a framework for managing marine fisheries and mitigating marine megafauna bycatch (Milner-Gulland et al., 2018; Squires & Garcia, 2018). In traditional fisheries management the MH is not explicitly referred to and EIAs are rarely requested, yet the ethos and process share many similarities (Squires et al., 2018; Squires & Garcia, 2018). Building on these similarities, the MH has already been applied to identify and implement least-cost approaches for sea turtle bycatch mitigation (Squires et al., 2018; Squires & Garcia, 2018). However, there is a need to further empirically demonstrate the utility of the MH for other marine species and fisheries.

At the time of writing, the MH had not yet been applied to shark management (though has since been applied to a case study on bycatch in trawlers in India (Gupta et al., 2020)). However, risk assessments of the vulnerability of sharks to fisheries are already commonly conducted, such as: Productivity-

Susceptibility Analyses (PSAs), Sustainability Assessment for Fishing Effects (SAFE) and Ecological Assessment of the Sustainable Impacts by Fisheries (EASI-Fish) (S. Griffiths et al., 2019; Hobday et al., 2007; S. Zhou & Griffiths, 2008). These methods quantify the relative vulnerability of species to fisheries based on susceptibility and productivity parameters, where susceptibility is based on the risk of a species being captured, and productivity is based on intrinsic life history parameters of the affected species. Derived vulnerability scores quantify the extent to which fisheries exceed the species' biological ability to recover, which are used to prioritise management action and research (Arrizabalaga et al., 2011; Braccini et al., 2006; Cortés et al., 2010; S. Griffiths et al., 2019; Hobday et al., 2007). These assessments can be seen as analogous to EIAs in terrestrial development projects, and the MH an extension of these widely accepted methods to quantify and manage risk. However, the MH also offers several novel advantages. In particular, it provides a framework for defining measurable goals, and structuring existing knowledge about potential management measures to achieve those goals (Milner-Gulland et al., 2018). This can facilitate transparent science-based management decisions, and highlight data gaps and uncertainties which hinder decision-making. Through least-cost implementation, the MH also enables socio-economic trade-offs to be explicitly factored in to decisions (Squires & Garcia, 2018). The MH also provides room for tailored fishery-specific or location-specific management, which can be combined to achieve net goals over a larger area or jurisdiction. This can encourage creative thinking about management measures and their implementation, and a shift of focus towards proactive creation of net outcomes for biodiversity as opposed to reactive avoidance of losses. The setting of measurable targets from the outset can also support monitoring of progress towards goals, and adaptive management (Milner-Gulland et al., 2018). In this paper I seek to demonstrate these advantages, as well as highlighting some challenges in applying the MH to sharks.

3.2.1 A conceptual model for risk to sharks in fisheries

Applying the MH to sharks requires an appropriate conceptual model for quantifying fishing mortality and understanding risk. A general model for shark fishing mortality for species X at time t ($F_{X,t}$) can be defined as shark-relevant fishing effort ($E_{X,t}$) multiplied by shark mortality per unit of that effort ($MPUE_{X,t}$; Equation 3.1, Figure 3.1).

Equation 3.1
$$F_{X,t} = E_{X,t} * MPUE_{X,t}$$

These components can be further decomposed into several constituent variables (Figure 3.1). Shark-relevant fishing effort ($E_{X,t}$) is a subset of the overall effort of a fishery that results in volumetric overlap with a population of shark species X within a certain time-period (t). This is a function of the areal overlap of fishing activity with the range of shark species X (P_{Ax}) at time t, and the proportion of effort that will lead to an interaction between the gear and the population of species X (i.e., encounterability) (P_{Ex} ; Equation 3.2, Figure 3.1).

Equation 3.2

$$E_{X,t} = E_t * P_{A_{X,t}} * P_{E_{X,t}}$$

Once shark-relevant effort is present for species X, the shark mortality per unit of that effort (MPUE_X) depends on the probability of being captured per unit effort (CPUE_X) and the probability of mortality once captured (P_{M_X}) (Equation 3.3, Figure 3.1). Mortality in fisheries occurs when caught sharks are retained, discarded dead, or discarded alive but suffer post-release mortality (Worm et al., 2013). Collateral mortality also occurs when dead sharks drop out of gears, are depredated after capture, or escape but die later due to exhaustion or injury. The proportion of sharks suffering mortality can therefore be decomposed into the proportion arriving dead on the vessel (P_{DOA_X}), the proportion dying on the vessel (P_{DOV_X}), the proportion dying after release (P_{DPR_X}) and the proportion dying collaterally (P_{COL_X}). Mortality of sharks on the vessel (P_{DOV_X}) may be intentional (e.g., due to retention or finning) or unintentional (e.g., due to injury or exhaustion).

Equation 3.3.

$$MPUE_X = CPUE_X * \overbrace{(P_{DOA_X} + P_{DOV_X} + P_{DPR_X} + P_{COL_X})}^{\text{Post-capture mortality } (P_{M_X})}$$

The model can be used flexibly to account for targeted and non-targeted shark fishing, or multiple species and scales. For example, for targeted shark fisheries E_{X,t} may be equal to E_t, such that the proportion of fishing effort that overlaps with the range of species X approaches 1. E_{X,t} could also be used for species-complexes in the same area with similar characteristics, or the equation could be extended to sum across multiple species and gear types.

It should be noted that these equations do not represent bio-economic models. Rather I intend to illustrate the different risk factors contributing to shark fishing mortality. In reality these factors are unlikely have an additive, linear relationships, and shark mortality will also be subject to random fluctuations in environmental factors and variation in technical efficiency and skipper skill (Kirkley et al., 1998).

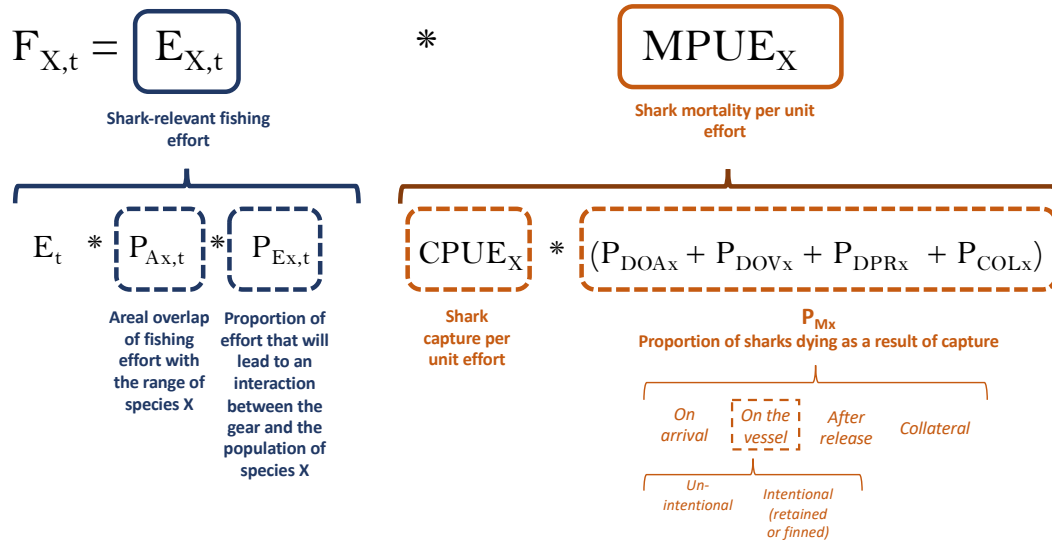


Figure 3.1 A conceptual model for shark fishing mortality, to decompose risks to sharks in fisheries

The components of equations 3.1-3.3 are further influenced by a range of direct and indirect factors, which may be operational, biophysical or socio-economic (Table 3.1). For example, shark-relevant fishing effort, likelihood of capture and likelihood of mortality directly depend on the operational characteristics of a fishery (e.g., fishing ground and gear specifications) the biophysical characteristics of a species (e.g., size, respiratory physiology, locomotor performance), and dynamic interactions between the two (Hobday et al., 2007) (Table 3.1). Operational factors are determined by active decisions made by fishers and skippers (Figure 3.2), while biophysical factors are primarily passive (i.e., not actively caused or influenced by fishers) (Table 3.1). Fisher decisions are in turn driven by indirect factors such as the market and regulatory environment, the perceived legitimacy of regulations, the risk of enforcement, social norms and individual beliefs (Arias et al., 2015; M. L. Barnes et al., 2016; Campbell & Cornwell, 2008; M. A. Hall et al., 2007) (Figure 3.2, Table 3.1). Together, these factors interact and combine to define the overall risk of mortality for a species in a fishery. The primary source of risk will vary for different species and fisheries, while different factors will act at different spatial and temporal scales. A holistic understanding of these different sources of risks, as well as their magnitudes, influenceability, and when and where they can be influenced, will help to identify points of leverage for effective mortality mitigation (Figure 3.2, Table 3.1).

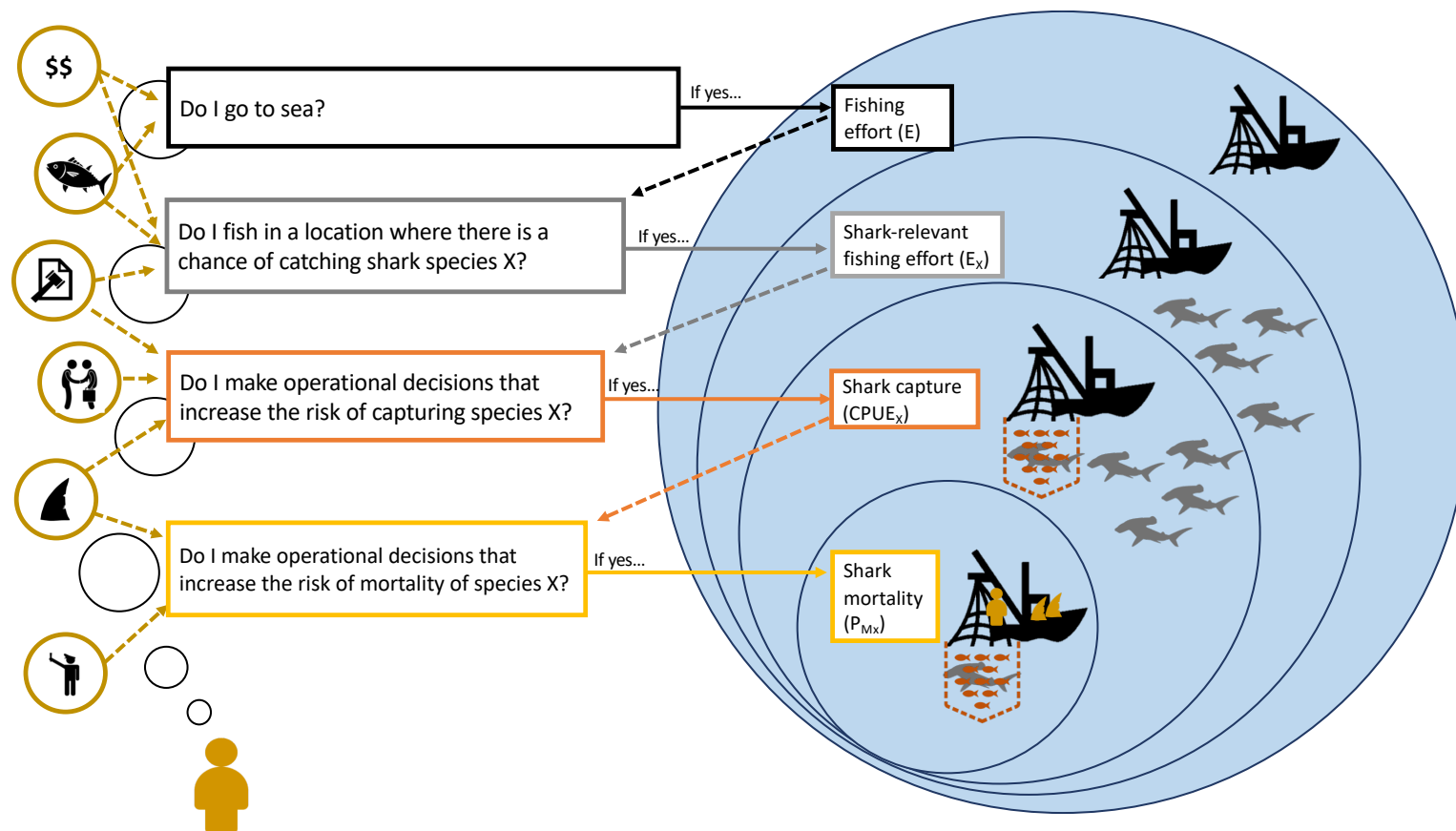


Figure 3.2 A schematic of the fisher decision-making process that leads to shark mortality. Fisher decisions influence the proximate technical causes of shark mortality, and fisher decisions are in turn influenced by a range of distal socio-economic factors (See Table 3.1)

Table 3.1 Direct and indirect factors affecting shark mortality at the point of catch

Equation components		Factors affecting components of fishing mortality		
		Operational (direct, active)	Biophysical (passive)	Socio-economic (indirect)
Shark-relevant fishing effort for species X ($E_{x,t}$)	Areal overlap of fishing activity with shark population ($P_{Ax,t}$).	- Target species - Fishing location	- Geographic range - Season - Climate	
	Encounterability. Proportion of effort that will lead to an interaction between gear and shark population ($P_{Ex,t}$).	- Set depth - Gear type and specifications - Soak time	- Maximum depth and depth range - Habitat-type - Habitat use (e.g., site fidelity, schooling)	
Mortality Per Unit Effort ($MPUE_x$)	Number of sharks captured by gear per unit of shark-relevant effort ($CPUE_x$)	- Gear type and specifications - Soak time - Mesh size - Hook size	- Size - Morphology - Locomotor performance	- Availability and value of marketable non-shark catch - Economic value and importance of sharks for income or subsistence
	Proportion of sharks that die due to capture (P_{Mx})	Proportion arriving dead on vessel (P_{DOAx})	- Morphology - Locomotor performance - Respiratory and metabolic physiology	- Regulations, perceived legitimacy and fairness of regulations, risk of enforcement
		Proportion dying on vessel (P_{DOVx})		- Economic costs - Incentives for compliance
		Unintentionally		
		Intentionally (due to retention or finning)		
		Proportion dying after release (P_{DPRx})	- Post-capture handling - Gear type and specifications - Hook type	- Locomotor performance - Respiratory and metabolic physiology
		Proportion dying collaterally (P_{DOLx})	- Gear type and specifications - Soak time	- Size - Locomotor performance - Predators

3.2.2 Operationalising the mitigation hierarchy for sharks

A proposed strength of the MH is that it provides a transparent framework for structuring knowledge and monitoring progress towards goals (Milner-Gulland et al., 2018). However, for these benefits to be realised, high-level concepts need to be operationalised in practical terms. User-friendly processes and definitions are required that allow managers to set goals and measurable targets, make informed decisions, and monitor progress. There is also a need for flexibility in order to handle complexity, data paucity and different management priorities.

I expand on the framework by Milner-Gulland et al. (2018) to suggest a process with five key stages: 1) Define the problem, 2) Explore potential management measures, 3) Assess hypothetical effectiveness of management measures, 4) Make decisions, 5) Implement, monitor and adapt (Table 3.2). This process draws on existing approaches for adaptive fisheries management, including Management Strategy Evaluation (Bunnefeld et al., 2011; Fulton et al., 2014) and feasibility assessments (Chapter 1, Booth, Squires, & Milner-Gulland, 2019b). I incorporate the MH into the process as a framework for structuring knowledge and making decisions.

Table 3.2 A multi-stage process for using the mitigation hierarchy to make science-based management decisions for sharks at the fishery level

Stage in the assessment	Key questions/considerations
1. Define the problem	
1.1. Understand the fishery	Fishery footprint, market-type, target species, targeting of sharks
1.2. Define the species of management concern	Single species, taxonomic group or species complex
1.3. Assess the risks	
1.3.1. Biological (species)	Size, fecundity, biological reference points, extinction risk
1.3.2. Technical (fishery)	Encounterability, catchability and survivability of species in fishery
1.3.3. Socio-economic (context)	Uses and values of sharks, target markets
1.3.4. Constraints (context)	Budget for monitoring, enforcement and implementation. Societal limits on acceptable damage to species or costs to people.
1.4. Set goals and quantitative targets	
1.4.1. Goal	Desired change in biodiversity (e.g., no net loss, net gain, population recovery, mortality minimisation, population stability, fishery sustainability).
1.4.2. Target	Quantitative target which operationalises the goal
1.4.3. Metric	Units to measure gains and losses in biodiversity to evaluate progress (e.g., population growth, total mortality, number of animals).
1.4.4. Baseline	Reference point against which progress is assessed.
1.4.5. Counterfactual	Projected change in metric in business-as-usual scenario.
2. Explore management measures	Which management options are available for achieving the target at each step? What data are available for estimating their impact on the target? What are the uncertainties?
2.1. Avoid	Options for avoiding encounters (i.e., reducing E _x)

2.2. Minimise	Options for minimising capture, given E_X is present (i.e., reducing $CPUE_X$)
2.3. Remediate	Options for minimising mortality, given sharks are captured (i.e., reducing $MPUE_X$)
2.4. Compensate	Options to compensate for residual mortality (i.e., increasing C_X)
3. Assess hypothetical effectiveness of management measures	
3.1. Technical assessment	To what degree could management measures reduce risks to the species, based on biophysical and operational factors?
3.2. Feasibility assessment	To what degree could management measures be feasibly implemented, given costs, benefits, social context and resources for implementation? Is there scope for incentives to address gaps?
4. Make a management decision	Which mix of measures and instruments are likely to have the greatest impact?
5. Implement, monitor and adapt	Implement measures and encourage uptake. Monitor progress towards target. Adapt management.

Defining the problem

Preliminary information

Milner-Gulland et al. (2018) start with defining a goal. The goal is the high-level desired change in biodiversity as a result of management. For sharks, the goal will depend on the level of the management unit and the species and fishery(s) of concern. As such, preliminary information on the fishery and species of concern will be required to set reasonable goals and targets. Useful preliminary information includes the species' biological characteristics, the fishery's operational characteristics, the socio-economic context, and constraints such as budget for monitoring, enforcement and implementation (Table 3.2). This information will help to define the overall mortality risk for a given species-fishery combination, as per equations 3.1-3.3 and Table 3.1. Preliminary information can be collected through a range of methods, including a review of available literature, or primary data collection via on-board observers, landings surveys, socio-economic surveys or key informant interviews (Rigby, Appleyard, et al., 2019; I. Yulianto et al., 2018).

Goals

Once background information is clear, a management goal can be set. Goal setting can take place at different scales, from global-, to national-, to fishery-level, or even as a joint goal for RFMOs, shared stocks or the High Seas. The goal can be defined in terms of NNL, net gain, population stability, population recovery, sustainability or simply catch minimisation, depending on what is practical given budgetary and operational constraints. For example, a national-level policy goal could be linked to CITES implementation for a species listed on Appendix II, such as silky sharks (*Carcharhinus falciformis*). The overall goal could be population stability, to avoid utilization of silky sharks that is incompatible with their survival. Another country may seek to restore populations of Critically Endangered species, such as sawfish (*Pristis spp.*, Pristidae), with a goal of net gain or population recovery. Corresponding goals can also be set at finer spatial scales, such as the fishery level. To achieve a national-level goal of silky shark population stability, the goals for all fisheries throughout a national jurisdiction could be no net loss of

silky sharks. Alternatively, by thinking in net terms, different goals can be set for different fisheries, acknowledging heterogeneity in fishery impacts, dependence on sharks and adaptive capacity of fishers. For example, vessels taking silky sharks as non-target catch in high-value commercial fisheries could be required to achieve net gain through additional or multiplicative compensatory actions. Small-scale fisheries that are more dependent on silky sharks for income and food security could then be permitted to have a net negative impact on the national silky shark stock, provided the gains and losses across all fisheries combine to achieve net population stability at the national level.

Targets

Goals must be operationalised through quantitative targets, for which metrics and baselines can be defined. Expanding on Equation 3.1 offers a general equation for a shark management target where $\Delta_{\lambda T}$ is the target level of net damage/recovery to the species of concern with respect to a baseline (Equation 3.4).

Equation 3.4

$$\Delta_{\lambda T} = \underbrace{f(M_X) - C_X}_{(E_X * MPUE_X)}$$

The term $f(M_X)$ is the net damage inflicted by fishing on species X, which is a function of the effort directed at species X and the mortality thus caused. C_X is the net effect of compensatory conservation efforts to improve the viability of the stock or species elsewhere (Milner-Gulland et al., 2018). Milner-Gulland et al. (2018) propose that targets be defined in terms of net change in population growth rate (the metric) with respect to an agreed baseline. A $\Delta_{\lambda T}$ of zero implies no change in population growth rate with respect to the baseline. A positive or negative $\Delta_{\lambda T}$ implies increases or decreases in population growth rate, respectively.

To return to the silky shark example, if the overall goal is population stability a suitable quantitative target could be $\Delta_{\lambda T} \geq 0$, with a static baseline set at zero population growth rate. At fishery levels, a uniform target of $\Delta_{\lambda T} \geq 0$ could also be set across all fisheries. Alternatively, to allow for heterogeneity in fisheries and goals as discussed above, commercial vessels that take silky sharks as non-target catch could be required to achieve $\Delta_{\lambda T} > 0$, while small-scale vessels more dependent on shark catch could be permitted $\Delta_{\lambda T} < 0$, with the net result summing to $\Delta_{\lambda T} \geq 0$. For sawfish recovery, net gain targets ($\Delta_{\lambda T} > 0$) could be set for specific species-fishery combinations, depending on the area of occurrence of different species and the fishery threats.

In theory, once a desired $\Delta_{\lambda T}$ is set, equation 3.4 can be solved to define acceptable levels of E_X and $MPUE_X$, which could in turn inform effort or catch quotas. Further decomposition of E_X and $MPUE_X$ into their constituent elements allows identification of management options to achieve to these targets (See Section 2.1.2).

The benefit of adopting targets based on population growth rates is that they focus on the aspirational goal of population health, with a direct relationship between the target and the conservation status of the species. However, such targets require a good understanding of the relationship between population growth rates and mortality. Yet sharks are a data poor group, with limited understanding of population dynamics and fishing mortality for many species (Cashion et al., 2019; Dulvy et al., 2017; Dulvy, Fowler, et al., 2014). Data paucity is particularly challenging in lower income countries, which represent many of the biggest priorities for management (Momigliano & Harcourt, 2014). As such, targets based on population growth rate may need to be considered the ‘gold standard’ for data rich, high-capacity situations. Simpler targets can be adopted in data poor, lower capacity situations where population models and stock assessments are lacking. Targets could be based on abundance, catch or catch per unit effort, depending on what data is available (Table 3.3). To return to the silky shark example, the target could be a total catch quota lower than the level required to yield MSY, based on known biological reference points. For sawfish recovery, the target could be based on abundance estimates. Crucially, the target should be quantitative and measurable. In very data poor situations where this is not possible, an aspirational target could be set while more data are collected to inform a revised target (Table 3.3). Targets can be adjusted over time as the situation changes.

Finally, acknowledging trade-offs and societal limits, some targets may need to be set based on regulatory, cultural and economic constraints. For example, ‘minimise mortality of species X whilst maintaining the economic viability of the fishery’ or ‘minimise mortality of species Y whilst maintaining income of vulnerable fishers’. For these targets, the equation for Δ_{NT} could be solved by expressing E_X , $MPUE_X$ and C_X as functions of cost, and including budgetary or socio-economic constraints. I discuss this further in Section 2.1.3.

Table 3.3 Examples of different goals and targets that could be used, depending on the fishery, data availability and capacity.

Example Fishery	Species of management concern	Data availability	Goal	Target	Methods	Key references
Commercial mixed gear fishery for spiny dogfish in Northwest Atlantic, USA	Spiny dogfish (<i>Squalus acanthias</i>)	Very good – population models, life-history and total fishing mortality	Fishery sustainability	Total fishing mortality $\leq F_{MSY}$	Define based on stocks and modelled projections of stocks under different fishing mortality rates. Monitor based on catch and mortality data.	Simpfendorfer & Dulvy, 2017; Sosebee & Rago, 2017
Commercial shrimp trawls taking sawfish as bycatch in Gulf of Mexico, USA	Smalltooth sawfish (<i>Pristis pectinata</i>)	Good – abundance estimates	Net gain	Abundance increases at 2% per year relative to baseline until 10% increase achieved.	Define and monitor based on estimated abundance from shark tagging studies.	NOAA Fisheries, 2019b
Commercial tuna purse seine taking pelagic sharks as bycatch in Western and Central Pacific Oceans	Silky sharks (<i>Carcharhinus falciformis</i>)	Moderate – catch and catch per unit effort time series	Net gain	Total fishing mortality $< F_{40\%}$	Defined based on precautionary biological reference points, monitor based on catch.	Restrepo et al., 2017
Small-scale longlines taking mixed pelagic sharks in Lombok, Indonesia	Scalloped hammerheads (<i>Sphyrna lewini</i>)	Moderate – catch and catch per unit effort time series	Population stability	Catch $\leq F_{40\%}$	Defined based on precautionary biological reference points, monitor based on catch.	Yulianto et al., 2018
Small-scale coastal gill nets taking wedgefish as secondary catch in Aceh, Indonesia	Wedgefish (<i>Rhynchobatus</i> spp.)	Poor – patchy catch data	Catch minimisation while maintaining household income of fishers	Total wedgefish catch and bycatch ratio decline by 30%, while maintaining total value of catch.	Define and monitor based on catch data and fisher interviews.	M. Ichsan pers. comm
Artisanal multi-gear fishers taking reef-associated species in Fiji	Reef sharks	Very poor – no catch data	Catch minimisation while maintaining food security	Shark catch declines by 10% each year, while maintaining total catch weight.	Define based on fisher interviews, monitor and refine based on catch data.	Glaus et al., 2018

key: F_{MSY} = fishing mortality that achieves maximum sustainable yield (MSY). $F_{40\%}$ = fishing mortality at 40% MSY.

More aspirational.
Suitable in data rich and high-capacity situations

More pragmatic.
Suitable in data poor and limited capacity situations

Exploring management measures

Once goals and targets are set, management measures need to be identified and assessed. If the data are adequate, this can be done quantitatively through solving equation 3.4 and considering the various determinants of M_x and C_x . However, in most cases, the data may be insufficient for a full quantitative assessment.

Existing measures for shark mortality mitigation can be categorised in to the first three steps in the MH: avoid, minimise and remediate, as outlined in Table 3.4. These steps also correspond to the different sources of fishing mortality risk outlined in equations 3.1-3.4 and Table 3.1, and the different steps in fisher decision-making (Figure 3.2). Avoidance strategies are measures to reduce the probability of encounter between potentially harmful gear and a potentially (by)-caught individual, by separating fishing activity from individuals or stocks of concern. This can be considered equivalent to a reduction in $E_{x,t}$. Examples of avoidance strategies include, no-fishing zones, depth restrictions or closed seasons (Milner-Gulland et al., 2018, Table 3.4). To translate avoidance into a reasonable risk-based definition for sharks, I propose that measures leading to <5% probability of a potentially harmful gear being within 1km of a shark stock of concern (for vessel i , during time t , operating in spatial extent j) are considered avoidance. While measures such as marginal reductions in fishing effort within an area of shark availability are minimisation. Using this definition, fishing zonation or closures for avoidance could be defined according to overlap between the spatial and temporal extent of the fishery and accepted habitat distribution maps for the species of concern (Table 3.4).

Where avoidance is neither feasible nor necessary, minimisation strategies can reduce the probability of sharks being captured, given that shark-relevant effort is present. These measures are equivalent to a reduction in $CPUE_x$. Minimisation strategies can reduce capture of species of concern, while allowing for sustainable exploitation of co-occurring species with healthier populations. Existing fisheries management measures that qualify as minimisation include reductions in effort or technology and gear specifications to reduce capture of particular species and sizes (Table 3.4). For example, in gill nets, modifications to net size and tension can minimise of susceptibility of certain species and life history stages to meshing and entanglement (Harry et al., 2011; Thorpe & Frierson, 2009). For purse seine vessels attractants, deterrents, backdown procedures and the design of Fish Aggregation Devices (FADs) can reduce capture of pelagic sharks (Restrepo et al., 2017) (Table 3.4).

Remediation strategies facilitate live release of individuals, their safe return to the sea, and their post-release survival (Table 3.4). Remediation includes pre- and post-haul measures that reduce the probability of mortality, given a shark is captured in a gear. This includes steps to increase pre-haul escape, and increase survival if brought on deck and subsequently released. Remediation is equivalent to reductions in P_{DOA} , P_{DOV} , P_{DPR} and P_{COL} . Examples of pre-haul remediation measures include use of nylon

monofilament leaders in pelagic longlines to allow sharks to bite off and escape before haul back (Ward et al., 2008), and the use of exclusion devices to allow escape of large sharks and rays from trawls (Brewer et al., 2006) (Table 3.4). Once on the vessel, post-capture handling such as reducing time out of the water, cutting the line off quickly and close to the hook, and gentle handling, can facilitate post-release survival (Kaplan et al., 2007) (Table 3.4). Use of circle hooks instead of J-hooks also promotes easy hook removal and reduces severity of injury, and corrodible hooks may minimise long-term damage or injury once sharks are released (Cooke & Suski, 2004). Finning bans or retention bans also apply to this category, since they effectively reduce the probability of sharks dying on-board vessels (Table 3.4).

Finally, compensation occurs to offset unavoidable residual damage to the population once all reasonable measures have been taken to avoid, minimise and remediate. Compensation may be particularly important for high vulnerability, low survivability pelagic species, which are caught in commercially important fisheries that cannot feasibly be closed. To my knowledge compensation has not been applied in a shark management context, though it is used for sea turtle bycatch mitigation. A voluntary bycatch tax is levied on tuna processors via the International Seafood Sustainability Foundation (ISSF), which then funds high-priority sea turtle conservation projects in the Atlantic, Indian, Eastern Pacific, and Western and Central Pacific Oceans, including nesting site protection, bycatch and subsistence take reduction in small-scale fisheries, education and research (Squires et al., 2018). These compensatory conservation efforts are estimated to have a higher conservation benefit, in terms of turtle population growth rate, per dollar cost than other measures to avoid and minimise capture (Gjertsen et al., 2014). A similar mechanism could be adopted for shark mortality mitigation, through bycatch taxes on commercial fisheries which are invested in conservation actions to improve the status of the fishing-affected population. For example, payments could be instituted to support the protection and management of pupping and nursery grounds, and reduce take in small-scale fisheries, as has been demonstrated for sea turtles (Gjertsen et al., 2014; Squires et al., 2018). Though in order to be true compensation, the increase in survival probability as a result of compensatory conservation must be at least equivalent to the mortality probability of the harmful gear. To address this uncertainty, high offset multipliers could be applied to bycatch taxes, as has proven to be a key success factor for delivering ecological outcomes in terrestrial applications of compensatory mitigation (zu Ermgassen et al., 2019).

Table 3.4 Summary of technical measures for managing shark mortality for each steps in the mitigation hierarchy, and examples of their use in existing fisheries management/policy for sharks, where applicable.

Operational fishery variables	Example effects on sharks (Applicable gears)	Examples of use in existing fisheries management plans and policy	Key references
Avoidance: Avoid encounters of sharks with fishing gear, given sharks are present. Equivalent to a reduction in Ex. (Avoid defined as <5% probability of a potentially harmful gear being within <1km of a shark of management concern)			
Spatial location of fishing activity	Spatial trends in catch rates related to habitat preferences, movement patterns and aggregating behaviour (LL, GN, PS, TR).	No-take MPAs (e.g., Raja Ampat, Indonesia), permanent closures to particular vessels (e.g., shark sanctuaries ban commercial shark fishing), species-specific area-based management (e.g., time-area closures to protect gummy sharks migrating to pupping grounds in Australia).	Afonso et al., 2011; Bromhead et al., 2012; Gray, Broadhurst, Johnson, & Young, 2005; Jaiteh et al., 2016; Oliver, Braccini, Newman, & Harvey, 2015; Poisson, Gaertner, Taquet, Durbec, & Bigelow, 2010; Sepulveda & Aalbers, 2018; Shiffman & Hammerschlag, 2016b; Sybersma, 2015; Ward-Paige & Worm, 2017; Yulianto et al., 2018
Depth of fishing activity	Depth trends in catch rates related to habitat preferences and movement patterns (LL, GN, PS, TR).	-	
Time of year or season of fishing activity	Seasonal time/area closures avoid seasonally migrating or aggregating species (LL, GN, PS, TR).	Direct regulation of fishing seasons (e.g., Canada's Atlantic Fisheries Regulation establishes closed seasons for commercial and recreational shark fishing), time-area closures once catch limits have been met (e.g., shark FMPs for Gulf of Alaska and NW Atlantic & Gulf of Mexico in USA).	
Minimisation: Minimise capture of individuals in fishing gear, given shark-relevant effort is present. Equivalent to a reduction in CPUE _{Ex} .			
Gear type	Different total catch and bycatch ratios for different gears (LL, GN, PS, TR).	Direct regulation of permitted gear (e.g., coastal GN ban in California in 1994 led to increases in soupfin shark (<i>Galeus galeus</i>) and leopard shark (<i>Triakis semifasciata</i>) numbers; ban on GN in Florida to minimize capture of smalltooth sawfish (<i>Pristis pectinata</i>)).	Afonso, Santiago, Hazin, & Hazin, 2012; BMIS, 2015; Brill et al., 2009; Gilman et al., 2008; Gray, Johnson, Broadhurst, & Young, 2005; Harry et al., 2011; NOAA Fisheries, 2019a; Ramírez-Amaro & Galván-Magaña, 2019; Restrepo et al., 2017; Thorpe & Frierson, 2009;
Gear deployment depth	Species-specific effects of fishing depth on catch rate (LL, GN, PS, TR).	-	
Gear deployment time	Species-specific effects of time of day on catch rate (LL).	-	
Bait	Mackerel style bait instead of squid bait reduces bycatch of pelagic sharks (LL).	-	

Attractants/deterrents	Species-specific effects of chemical cues, light cues and magnetic or electropositive metals on gear interactions (LL, GN, PS).	-	Wakefield et al., 2016; Ward et al., 2008; Watson, Epperly, Shah, & Foster, 2005; Yulianto et al., 2018
Mesh size, design and tension	Mesh size and tension influences selectivity for species and life history stage (GN)	-	
Fishing effort	Higher effort (vessels, gears, hook number) leads to higher catch rates (LL, GN, PS, TR).	Direct regulation of fishing effort through limited entry and permits (e.g., U.S. Atlantic Highly Migratory FMP for sharks requires fishers to obtain permits), direct regulation of fishing outputs through quotas and trip limits (e.g., U.S. Atlantic Highly Migratory Species shark fishery has a trip limit of 36 large coastal sharks).	
FAD management	Setting on FADs can cause higher levels of shark catch. Higher levels of collateral mortality associated with entangling FADs (PS).	Regulation of FAD design (e.g., several RFMOs require a transition to non-entangling FADs).	
Tickler chain	Tickler chain on bottom trawls increases catch rate of bottom-dwelling sharks and skates	-	
Remediation: Remediate individuals by ensuring their safe return to the ocean and post-capture survival, given capture has occurred. Includes steps to increase escape if captured, prior to being brought on deck; and increase survival if brought on deck and subsequently released. Equivalent to reductions in P_{DOA}, P_{DOV}, P_{DPR} and P_{COL}.			
Setting depth	Survival rates of some species vary with setting depth (LL, GN).	-	Braccini, Van Rijn, & Frick, 2012; Brewer et al., 2006; Brewer, Rawlinson, Eayrs, & Burridge, 1998; Cooke & Suski, 2004;
Soak time	Survival rates of some species vary with soak time (LL, GN).	-	Dapp, Huveneers, Walker, Drew, & Reina, 2016; Gallagher, Orbesen, Hammerschlag, & Serafy, 2014; Godin, Wimmer, Wang, & Worm, 2013; Kaplan et al., 2007; Kerstetter & Graves, 2006; NOAA Fisheries, 2019; Pacheco et al., 2011; Patterson, Hansen, & Larcombe, 2014; Poisson
Gear type	Survival rates of some species vary with gear type (LL, GN, PS, TR).	Direct regulation of authorised gears (e.g., Shark FMP for NW Atlantic and Gulf of Mexico establishes gear restrictions to reduce bycatch mortality).	
Hook type	Circle hooks promote easy removal/reduce severity of injury. Corrodible hooks promote ejection and minimise negative impacts of hooks on released individuals (LL)	Direct regulation of hook type (e.g., Shark FMP for NW Atlantic and Gulf of Mexico stipulates that bottom LL vessels must have non-stainless-steel corrodible hooks)	
Leader material	Nylon monofilament leaders can increase bite-off and escape of pelagic sharks (LL)	-	
Exclusion/ escape devices	Exclusion devices reduce capture of large sharks and rays in TR, escape grates reduce capture of spiny dogfish (<i>Squalus</i>	Direct regulation of gear specifications (e.g., All TR nets in Western Australia required bycatch reduction devices)	

	<i>acanthias</i>) in TR, escape panels may promote release of sharks in PS (PS, TR).		et al., 2010; Serafy, Orbesen, Snodgrass, Beerkircher, & Walter, 2012; Wakefield et al., 2016
Post-capture handling	Reducing time out of the water, cutting the line quickly and close to the hook in LLs, and gentle handling can increase post-capture survival (LL, GN, PS, TR).	Direct regulation of handling procedures or equipment on board to promote safe handling (e.g., Shark FMP for NW Atlantic and Gulf of Mexico stipulates that bottom LL vessels have dehooking device, line-cutters, and dipnet. All TR in Western Australia require onboard in-water sorting systems).	
Retention	Retaining sharks on board for landing and sale causes 100% mortality (LL, GN, PS, TR).	Retention bans, quotas.	
Finning	Removing fins and discarding carcass at sea causes 100% mortality (LL, GN, PS, TR).	Finning bans, fin-to-carcass ratios, or fins naturally attached.	

Compensate: Compensate for residual damage caused through off-site conservation efforts that increase in the probability of another individual in the same stock living to the same age/stage. Equivalent to increases in C_X .

Bycatch tax or fines	Finance off-site conservation efforts within the range of the catch- affected population	International Seafood Sustainability Foundation (ISSF) voluntary bycatch tax to finance sea turtle nesting habitat	Dutton & Squires, 2008; Finkelstein et al., 2008;
Payments in kind	Fisher time, resources and knowledge could contribute to monitoring, management and research within the range of the catch- affected population.	-	Gjertsen et al., 2014; ISSF, 2016; Milner-Gulland et al., 2018; Pascoe, Wilcox, & Donlan, 2011; Squires & Garcia, 2018

Key: LL = Longlines; GN = gill nets, PS = purse seine, TR = trawl. E_X = shark-relevant fishing effort for species X, $CPUE_X$ = catch per unit effort of species X, P_{DOA} = proportion of sharks dead on arrival, P_{DOV} = proportion of sharks dying on vessel, P_{DPR} proportion of sharks dying after release, P_{COL} proportion of sharks dying collaterally, C_X = the positive impact of compensatory conservation measures for species X. FMP = Fisheries Management Plan. FAD = Fish Aggregation Device.)

Assessing effectiveness

Once potential management measures have been explored, the hypothetical effectiveness of measures in achieving the target can be analysed. This should include an assessment of technical, biophysical and socio-economic risks (Table 3.1), and how they can be alleviated.

Technical effectiveness

As illustrated in Table 3.4, different management measures have varying degrees of effectiveness depending on the fishery and species. Assessments of technical effectiveness can be conducted by estimating quantities for the magnitude of avoidance (reduction in E_X), minimisation (reduction in $CPUE_X$), remediation (reduction in $MPUE_X$) and compensation (increase in C_X) that can be achieved for a management measure or combination of measures (Figure 3.3).

For some species-fishery combinations, in which habitat, selectivity and survivability studies have been conducted, data will be available to inform a quantitative assessment. E.g., several studies identify specific geographic areas with higher catch rates for certain species (Oliver et al., 2015; I. Yulianto et al., 2018). These data could help to identify priority areas for avoidance, and quantify hypothetical reductions in E_X . Catch and post-haul survival rates have been quantified for several species caught in longlines and gill nets, as well as the impacts of other operational variables (e.g., soak time, set depth) on these rates (Braccini et al., 2012; Braccini & Waltrick, 2019; Gallagher et al., 2014; Gilman et al., 2008). Studies have also quantified the effectiveness of some minimisation techniques, such as BRTs in prawn trawls (Brewer et al., 2006) and circle-hooks and nylon leader lines in longlines (Gilman et al., 2008; Ward et al., 2008), which could be used to quantify their hypothetical effectiveness in terms of $CPUE_X$ and P_{M_X} .

However, the effectiveness of many existing technical measures is not well quantified. For example, the hypothetical effectiveness of compensation schemes may be difficult to estimate due to a limited understanding of how conservation actions quantitatively influence shark populations, which gives rise to issues related to equivalence, additionality and time lags (Bull et al., 2013). Even for measures that are quantified, observed or tested efficacy may not always be replicated in practice, or may only apply to the conditions in which they were observed or tested (Campbell & Cornwell, 2008). Therefore, quantitative assessments of the hypothetical impact of management measure on a target will be challenging, particularly in small-scale fishery and low-capacity contexts. Rather, it may be necessary to elicit expert opinion or fisher knowledge to explore hypothetical effectiveness. Methods such as the IDEA protocol, Value of Information Analysis and Bayesian belief networks could be adopted in this process (Hemming et al., 2018; Milner-Gulland & Shea, 2017). During recommendations and implementation, precautionary multipliers could be applied to account for uncertainty. For example, large offset areas relative to impacted areas determines successful ecological outcomes in terrestrial biodiversity compensation schemes (zu Ermgassen et al., 2019).

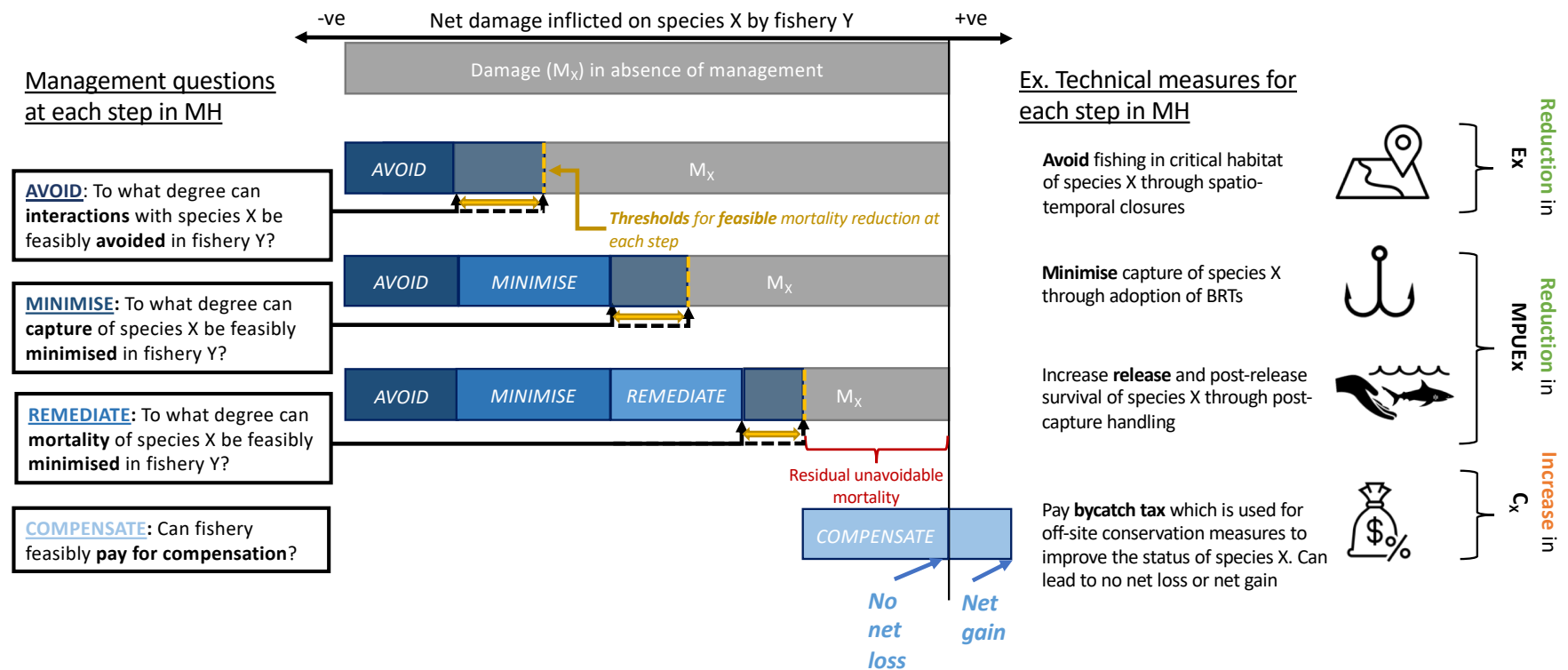


Figure 3.3 A step-wise decision framework for feasible shark management, based on the mitigation hierarchy (after BBOP (2012)).

Feasibility

The conceptual model and management measures I have presented thus far predominantly focus on the technical factors that influence risk of shark mortality. However, given the socio-economic complexities of shark fisheries, shark management is much more than a biological and technical issue: it is a human issue (Chapter 2; Booth, Squires, & Milner-Gulland, 2019b). Risk of post-capture mortality (P_{DOV} and P_{DPR}) and choices about fishing locations and gear deployment will depend on the behaviour and decision-making of fishers and skippers (Figure 3.2). As such, management decisions need to consider the fishery context and constraints, in order to avoid unintended consequences (Baum et al., 2003; Jenkins, 2006; Sarmiento, 2006), unacceptable costs (Campbell & Cornwell, 2008; Gilman et al., 2007; Jaiteh, Loneragan, et al., 2017) and implementation failure (Fulton et al., 2011). Accordingly, potential measures at different steps in the MH need to be assessed in terms of their likely effect on people. Building on previous work on conservation opportunity, conservation likelihood and cost-effective conservation (e.g., Ban, Hansen, Jones, & Vincent, 2009; Dickman, Hinks, Macdonald, Burnham, & Macdonald, 2015; Gjertsen et al., 2014; Knight, Cowling, Difford, & Campbell, 2010) I define these considerations as feasibility (Chapter 2; Booth, Squires, & Milner-Gulland, 2019b). Explicitly considering feasibility can highlight opportunities and barriers to implementation, as well as identify where novel instruments such as financial incentives and intrinsic motivations may be used to overcome implementation gaps (Booth, Squires, & Milner-Gulland, 2019b; Gjertsen et al., 2014; Selinske et al., 2017; Ward-Paige & Worm, 2017).

The proposed approach to feasibility assessments draws on principles from least-cost conservation, which seeks to achieve desired conservation goals at lowest total cost to society (Gjertsen et al., 2014; Squires et al., 2018; Squires & Garcia, 2018). In this approach, the marginal costs of mitigation measures (MC) are traded-off against the marginal benefits of biodiversity gains (MB). In principle, the economically optimal level of conservation occurs when the MC of each additional unit of mitigation reduction is equal to the MB of biodiversity gains (Figure 3.4).

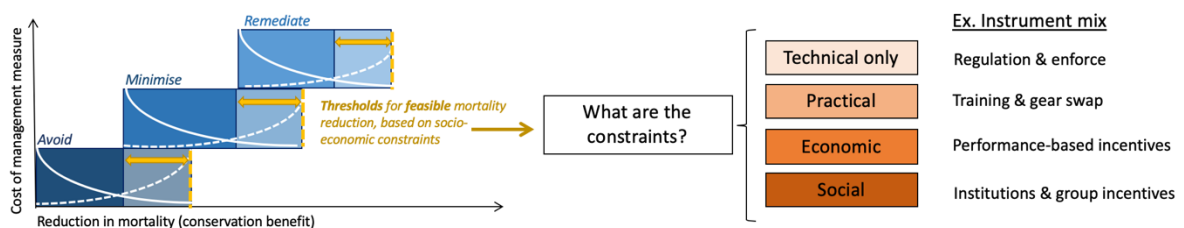


Figure 3.4 Cost and benefit curves for assessing socio-economic feasibility of management measures at each step in the mitigation hierarchy (after Squires and Garcia (2018)). Solid white lines represent the marginal conservation benefit (MB) of management measures at (i.e., reduction in mortality) at a given step. Dotted white lines represent the full marginal cost (MC) to the fishery (i.e., economic and social) of implementing management measures at a given step. Thresholds for feasibility at each step will be determined by socio-economic constraints. These constraints influence the marginal costs of potential management measures, and the instrument mix required to mitigate costs and achieve a desired management target. For least-cost conservation, the optimal management strategy occurs where the desired conservation benefits are achieved at lowest total cost.

However, in practice, the benefits of management measures will be based on physical conservation outcomes as opposed to their economic value. For example, if population models are available MB could be measured in terms of estimated increases in shark population growth rates as a result of mitigation measures, as had been used in cost-effectiveness assessments for sea turtles (Gjertsen et al., 2014). Alternatively, estimated reductions in shark mortality as a result of mitigation, such as estimated change in total catch, catch per unit effort or bycatch ratios, could also be used as a measure of the conservation benefit. Summing and comparing ratios of MBs to MCs for different management measures can help to identify which measures (and combinations of measures) are most cost-effective. The least-cost approach is powerful, as it acknowledges that most real-world conservation actions take place within socio-economic constraints, and explicitly incorporates trade-offs into the management decision-making process (Figure 3.4). In the case of shark fisheries, feasibility can encompass the direct economic costs of implementing a management measure for fishers (e.g., purchasing new gear) and managers (e.g., monitoring, enforcement, compliance management), the opportunity costs of profits foregone (e.g., from lost marketable catch), and the indirect and social costs (e.g., intangible impacts on culture, social networks, livelihood and food security, and well-being). As such, the MC curves illustrated in Figure 3.4 represent this holistic definition of cost (i.e., feasibility).

As with the technical assessment, quantifying feasibility poses several challenges in terms of data availability and uncertainty. I propose a potential approach for assessing and quantifying feasibility in shark fisheries, as proposed in Chapter 2, which could be applied here. This component of the assessment would need to be informed by social research methods, such as socioeconomic surveys, focus group discussions and predictive methods (Travers, Selinske, et al., 2019).

As with goal and target setting, the methods used for assessing feasibility can be adapted to suit different levels of data availability, capacity and budget. For example, costs could be defined quantitatively in economic terms, based on statistically-robust surveys of household income from shark fishing and market prices of shark products, or more qualitatively, based on fisher perceptions of the likely impacts of management measures on their lives (e.g., using scenario interviews or Likert scale questionnaires).

Feasibility assessments could be operationalised through a least-cost approach by considering catch reduction per unit cost (Gjertsen et al., 2014; Squires & Garcia, 2018) or per unit feasibility (Chapter 2; Booth, Squires, & Milner-Gulland, 2019b). The equation for $\Delta\lambda_T$ could be solved quantitatively by expressing E_x , $MPUE_x$ and C_x as functions of cost. For example, if the direct and opportunity costs of management measures can be estimated, in terms of income foregone due to reduced catches, then cost curves could be constructed for each unit of conservation benefit (i.e., mortality reduction (Figure 3.4)). This would also allow for the cost-effectiveness of different management measures to be compared, as conducted for the Pacific Leatherback Turtle (Gjertsen et al., 2014). However, caution should be

exercised with quantitative feasibility assessments. The methods used by Gjertsen et al. (2014) consider the overall economic costs to the fishing industry, yet there may be many intangible costs of shark conservation to small-scale fisher communities, which can be highly heterogeneous across space, time and demographic groups. A holistic approach to social costs and benefits, which captures the multiple facets of human well-being beyond income foregone may be required to ensure that people are no worse off (Booth, Squires, & Milner-Gulland, 2019b; Bull et al., 2018; Milner-Gulland et al., 2014). In principle, these holistic social costs could be calculated using social prices, which are commonly applied in social cost-benefit analyses for development project appraisals, and are calculated on a case-by-case basis to account for economic efficiency as well as equity and distributional concerns (Drèze & Stern, 1990; Little & Mirrlees, 1990; Squires & Vestergaard, 2015). More work is required to apply social prices to a fisheries management context, yet they have been applied to design equitable benefit sharing for deep sea mining, with potential lessons for fisheries management, particularly in high seas fisheries (Lodge et al., 2017).

Determining thresholds

Combining these two types of analyses would help to explicitly acknowledge trade-offs between shark conservation goals and socio-economic fisheries objectives, and thus define thresholds for feasible mortality reduction. These thresholds are illustrated by the yellow arrows and lines in Figure 3.3 and Figure 3.4. Thresholds will be determined by what is technically possible, based on the biology of the species, the operational characteristics of the fishery and available technical measures; and what is feasible, given the socio-economic context and key constraints. Determining thresholds and constraints can identify which management measures are likely to be most impactful and cost effective. In some cases, management measures which are technically possible may be unacceptably costly or unfeasible. These cases may require hard choices or adjusted expectations regarding goals and targets. However, through making socio-economic costs explicit in the planning phase, the MH can help to identify potential causes of implementation failure, and facilitate creative thinking about policies and instruments that could alleviate socio-economic constraints (e.g., training, building institutions or establishing performance-based incentives) (Figure 3.4).

Making decisions

Finally, all information and options need to be drawn together to make management decisions. Acknowledging the inherent complexity and data paucity of shark management, I propose a simple, low-tech approach for using the MH to make robust management decisions (Table 3.5). The approach uses an integrated framework based on informed judgement. A simple high-to-low or traffic light categorization system enables semi-quantitative assessments of effectiveness and feasibility, which can be used flexibly to handle multiple types of information and uncertainty. A semi-quantitative assessment is deemed appropriate here, as such approaches are already widely applied to risk and stock assessments for sharks and other fish species (Arrizabalaga et al., 2011; Braccini et al., 2006; Cortés et al., 2010), and in other

biological risk assessments (e.g., the IUCN Red List Assessment (Mace et al., 2008); the World Organisation for Animal Health risk assessment (Beauvais et al., 2018)). The framework can be used in conjunction with robust stock assessments and quantitative population models under different management scenarios, or informed by expert elicitation and stakeholder consultation where data is lacking. Populating the framework with available data can also help to highlight key uncertainties and data gaps to inform management-relevant research priorities.

The utility of the framework is illustrated in Table 3.5. I offer worked examples from four real-world fishery problems: a commercial purse seine tuna fishery taking pelagic sharks as bycatch in Western and Central Pacific Oceans, a small-scale coastal gillnet fishery taking wedgefish (*Rhinidae* spp.) as valuable secondary catch in Aceh, Indonesia, a small-scale longline fishery taking pelagic sharks as target catch in Lombok, Indonesia and commercial shrimp trawls taking sawfish as bycatch in the Gulf of Mexico, USA. This diversity of examples show how the MH can be used for a range of species and fisheries, in complex socio-economic contexts, and with varying degrees of data availability. For each fishery problem, management options at different levels of the MH are listed sequentially, and assessed in terms of their technical effectiveness and feasibility, based on existing knowledge. For some species-gear combinations the technical effectiveness of different measures can be quantified. For example, for silky sharks caught in tuna purse seines, studies have shown that avoiding purse seine setting on schools of tuna less than 10 tons can reduce amount of silky shark catch by 21%-41%, that at least 21% of silky shark bycatch can be fished out of purse seine nets and released, and that post-release survival of silky sharks in can increase by 20% with good handling (Restrepo et al., 2017). This can be used to quantify or categorise to what degree a given measure could contribute towards achieving the target (Table 3.5). In addition, the sequential impact of these measures can be summed to estimate an overall technically achievable level of mortality reduction, and how this would contribute towards achieving the management goal. Where information is limited, it may be possible to make informed judgements based on studies for similar species. For example, while I am not aware of any studies on the effectiveness of BRTs for sawfish in trawls, Brewer et al. (2006) showed that turtle exclusion devices (TEDs) can be effective at reducing catch rate of large sharks and rays, which could be used as a reasonable proxy of effectiveness sawfish. If appropriate proxies are uncertain or unavailable research priorities can be highlighted (Table 3.5).

Socio-economic context and practical constraints are explicitly considered through feasibility. This can highlight areas where there are mismatches between what is technically possible and socio-economically feasible. It can also highlight opportunities where incentives or new institutions could be used, such as bycatch taxes in commercial fisheries or payments for ecosystem services in small-scale fisheries (e.g., Gjertsen et al., 2014; Selinske et al., 2017), to address these mismatches. For example, rhinidae species exhibit fairly high post-capture survival rates (Ellis et al., 2017; Fennessy, 1994). This suggests that remediation through post-capture release is technically achievable for wedgefish captured in gillnets.

However, in small-scale gillnet fisheries in Indonesia, wedgefish represent high value secondary catch, and play an important role in income and food security. As such, release protocols may represent an unacceptable cost to fishers (Table 3.5). In this case incentives such as payments for ecosystem services and collaborative research could better align conservation objectives with fishers' socio-economic needs. Feasibility can also help to highlight management measures that should not be pursued, since they are ineffective or non-implementable. For example, captured hammerhead sharks (*Sphyrna* spp.) exhibit high at-vessel mortality and low post-release survival rates. In addition, in many fisheries, particularly those targeting sharks, there are strong socio-economic incentives to retain them on board due to their high value. As such, post-capture remediation strategies for hammerhead sharks are unlikely to yield meaningful impacts on fishing mortality. Management efforts should instead focus on avoiding and minimising capture as far as possible (Table 3.5). For targeted shark fisheries this may require measures which shift fishing effort away from hammerhead aggregation sites while allowing for sustainable increases in exploitation of less threatened species such as milk sharks (*Rhizoprionodon acutus*) and blue sharks (*Prionace glauca*).

These various pieces of information can then be drawn together to make an overall assessment and management recommendation, which can include technical measures, policy design and research needs (Table 3.5).

Table 3.5 A simple framework for using the MH to assess the effectiveness of potential measures and make management decisions, with real-world example case study fisheries.

Example fishery	Species of concern, management goal and target	MH Step	Potential measure	Technical assessment		Feasibility assessment		Overall assessment/ management recommendation
Commercial purse seine tuna fishery taking pelagic sharks as bycatch in Western and Central Pacific Oceans	Silky sharks (<i>Carcharhinus falciformis</i>) Net gain (Total fishing mortality < MSY)	A	Spatio-temporal closures	✓ ✓	Parts of range could be closed to fishing, but species is wide-ranging, circum-global. Critical habitat and impact of closures on mortality unclear.	✓	Direct overlap with target species, closure of large areas of fishing ground not economically viable. Off-shore monitoring and enforcement is costly.	A: Spatio-temporal closures where feasible. M: Species-specific fishing restrictions or low quota, with FAD regulations. R: Best practice live release protocols. Trade interventions to reduce incentives to retain. C: Mandatory on-vessel monitoring. Bycatch tax for mortality over and above quota to incentivize good performance and compensate for unavoidable mortality. Needs: Research on effectiveness of escape panel, attractants and post-release survivability, and conservation measures.
		M	Escape panel	?	Effectiveness varies – measure needs to be explored.	✓ ✓	Tested in some fisheries, commonly adopted to reduce dolphin bycatch.	
			Make fewer sets on FADs, especially with low tuna abundance	✓ ✓	Sets on tuna schools >10 tons can reduce SS catch by 21%-41%	✓ ✓	May lead to loss of target catch by 3-10%.	
			Attract sharks away from FADs	?	50% of sharks can be lured away with bait, though not tested on sets.	✓ ✓	Luring requires time and resources.	
		R	Fish and release sharks	✓ ✓	100% of fished and released sharks survive, though only 21% of those encircled could be fished.	✓ ✓	SS not target species, though are marketable catch. Some incentives to retain. On-vessel monitoring and enforcement is costly.	
			Use best handling and release protocols	✓ ✓	High at-vessel mortality, post-release survival can increase by 20% with good handling.	✓ ✓		
		C	Bycatch tax	?	Off-site conservation measures to be assessed.	✓ ✓	Commercial fishery has business risk and resources to pay, but requires costly monitoring. [\$]	
Small-scale coastal gillnet fishery taking wedgefish as valuable secondary catch in	Wedgefish (<i>Rhynchobatus</i> spp.)	A	Spatio-temporal closures	✓ ✓ ✓	Known critical habitat could be closed to fishing.	✓ ✓	WF co-occur with target species, degree of overlap needs to be confirmed.	A: Managed GN use in areas with highest bycatch ratios. M: Restrictions or low quota. R: Live release protocols and improved handling C: Compensation in kind
	Minimise mortality	M	Mesh size and tension to reduce entanglement, electro-sensory deterrents	?	Species-specific effectiveness to be explored.	✓ ✓	Limited capacity to purchase new gear. Potential impacts on target species need to be understood. [\$]	

Aceh, Indonesia		R	Live release protocols and improved handling	✓ ✓ ✓	WF robust to capture in GN, high survivability.	✓	WF high value marketable catch. On-board monitoring of SSFs is challenging and costly [\$].	Needs: Performance-based incentives – training and transitional payments to promote safe handling and release. Participatory research in to survivability. Free gear swaps with participatory testing. Trade interventions to reduce value.
		C	Payments in kind	?	Fishers could contribute time and knowledge to conservation efforts.	✓ ✓	Fishers have limited resources, but may be able to pay in kind.	
Small-scale longline fishery taking pelagic sharks as target catch in Lombok, Indonesia	Scalloped hammerhead sharks (<i>Sphyrna lewini</i>) Population stability	A	Spatio-temporal closures	✓ ✓	HH co-occur with other target species, though exhibit schooling. Closures may be possible for aggregations.	✓ ✓	HHs high value target species, though other species available. Off-shore monitoring and enforcement is costly.	A: Spatio-temporal closures at aggregation sites M: Vessel permits and species-specific catch quotas to reduce E _{HH} . Gear restrictions/modifications to minimize CPUE _{HH} . C: Compensation in kind
		M	Hook number and setting depth	✓ ✓	Hook number and setting depth influences CPUE.	✓ ✓	HHs are high value target species, some cultural attachment to fishing gear. [\$]	
		R	Live release protocols and improved handling	✓	High at-vessel mortality and low post-release survival.	✓	HHs are high value, incentives to retain once on board. On-board monitoring and enforcement is costly.	
		C	Payments in kind	?	Fishers could contribute time and resources to protecting pupping grounds.	✓ ✓	Fishers have limited resources, but may be able to pay in kind	Needs: Performance-based group incentives to reduce mortality, individual awards for exceptional fishers/vessels. Gear swap. Trade interventions to reduce value.
		A	Spatio-temporal closures	✓ ✓ ✓	Critical habitat could be closed to fishing.	✓ ✓	Co-occurrence with target species, complete avoidance would close fishery. Enforcement is costly.	
Commercial shrimp trawls taking sawfish as bycatch in Gulf of Mexico, USA	Smalltooth sawfish (<i>Pristis microdon</i>) Net gain	M	Bycatch reduction devices – TED	✓ ✓	TEDs can reduce capture of large sharks and rays by >60%. SW specific effect unclear.	✓ ✓	Reduces capture of prawns by 2-12%.	A: Spatio-temporal closures where feasible. M: Species-specific prohibitions and gear-based regulations. R: Best practice live release protocols C: Bycatch tax or fines for failure to comply.
		M	Bycatch reduction devices – tickler chain removal	✓ ✓	Removal of tickler chain can increase escape of demersal sharks and rays by ~30%. SW specific effect unclear.	✓	Reduces capture of other commercially valuable/marketable species.	
		R	Use best handling and release protocols	?	Post-release survival rates of SW unclear.	✓ ✓ ✓	Prohibited species/non-marketable in USA.	
		C	Fine or bycatch tax	✓ ✓	Funds for critical habitat protection, enforcement and abundance surveys.	✓ ✓	Industry have resources to pay, requires monitoring and enforcement.	Needs: Mandatory on-board monitoring. Research on effectiveness of TEDs, post-release survivability, and potential conservation measures.
		A	Spatio-temporal closures	✓ ✓ ✓	Critical habitat could be closed to fishing.	✓ ✓	Co-occurrence with target species, complete avoidance would close fishery. Enforcement is costly.	

Key: A= Avoid, M= Minimise, R= Remediate, C= Compensate. [✓] = low, [✓✓] = moderate, [✓✓✓] = high. [\$] = potential for incentives. LL = longline, GN = gillnet, TR = trawl, PS = purse seine, FAD = fish aggregation device, TED = turtle exclusion device. SS = silky sharks, HH = hammerhead sharks, WF = wedgefish, , SW = sawfish.

Implement, monitor and adapt

Once a management decision has been made, measures need to be implemented. This will likely entail a combination of technical measures, with appropriate policies and instruments to facilitate uptake. Alongside this, research and monitoring can fill data gaps and assess progress towards goals. Monitoring will enable continuous updating of models and assessments to verify assumptions and uncertainties and respond to dynamic changes in the socio-ecological system. This can inform changes in management strategies based on updated information (i.e., adaptive management) and progress towards more aspirational and quantifiable targets over time. On-going stakeholder engagement will be crucial throughout to understand the socio-economic impacts of management actions. This can help to ensure people are no worse off as a result of management, and drive change and commitment towards bolder actions (Bull et al., 2018; M. A. Hall et al., 2007). In more intractable cases, where trade-offs between social and ecological objectives are acute, the MH approach can support incremental change, with goals becoming more ambitious over time.

3.3 Discussion

Many shark species and populations are threatened by overfishing (Dulvy et al., 2008; Dulvy, Fowler, et al., 2014). Precautionary approaches for mitigating shark fishing mortality are required throughout global fisheries. Yet robust science-based management is hindered by the inherent complexity, uncertainty and data paucity of shark fisheries (Dulvy et al., 2017). A key source of complexity and uncertainty in fisheries management stems from humans (Fulton et al., 2011), with a need to think more explicitly about the human dimensions of shark fisheries, and the trade-offs between conservation objectives and socio-economic objectives, during management decision-making (Chapter 2; Booth, Squires, & Milner-Gulland, 2019b)

I have presented a novel process and framework for holistic risk-based shark management which can help to address this gap. It builds on efforts by Milner-Gulland et al. (2018) and Squires and Garcia (2018) to apply the MH to marine fisheries management and bycatch mitigation, as well as previous work by Hall (M. A. Hall, 1996; M. A. Hall et al., 2000) and BBOP (2012). The framework draws from existing concepts of risk-based management for sharks (Arrizabalaga et al., 2011; Cortés et al., 2010; S. Griffiths et al., 2019; S. Zhou & Griffiths, 2008) and extinction risk assessments (Dulvy et al., 2014), but offers several novel advantages. In particular, the MH encourages thinking in net terms, and summation of different actions across multiple sites and scales to meet high-level aspirational goals. This can facilitate a move away from one-size-fits all policies for shark conservation, towards context-specific fisheries management, with diverse pathways towards united outcome goals. The MH also provides a structured framework to bring together a range of potential management measures. The process I propose enables evaluation of each potential measure, in the context of the whole suite of measures, in terms of their likely combined effectiveness in achieving a management goal. The framework can highlight which

measures could have the greatest conservation impact (e.g., Milner-Gulland et al., 2018; Shiode, Hu, Shiga, Yokota, & Tokai, 2005) and the lowest cost (e.g., Gjertsen et al., 2014), thus facilitating practical science-based decision making. With quantitative targets and metrics, the actual effectiveness of management actions can then be monitored to enable adaptive management. The framework is also flexible and user-friendly. It can handle multiple types of information, and can be adapted to different levels of data availability and capacity. Further, by explicitly acknowledging uncertainty, the framework can highlight data gaps and research priorities. Finally, by integrating socio-economic feasibility, the framework explicitly considers trade-offs and constraints. This can facilitate creative thinking about least-cost shark conservation, and identify novel instruments to improve implementation. As for any fisheries management issue, poor regulation, limited capacity for monitoring and enforcement, and limited compliance could hamper implementation. Yet I hope that taking constraints into account during management planning can better align shark conservation objectives with the socio-economic needs and constraints of fishers, and minimise implementation failure (Fulton et al., 2011; M. A. Hall et al., 2007; Squires & Garcia, 2018).

Moving forwards, it will be important to provide a proof of concept for this framework by empirically demonstrating its utility in real-world fisheries, particularly in data-poor situations. This will require an inter-disciplinary approach, which incorporates fisheries science with social science, and considers shark fisheries as integrated socio-ecological systems (Ostrom, 2009). As well as filling data gaps on fundamental biological and fisheries factors to answer management questions, there is a need to better understand the broader socio-economic factors that drive shark fishing behaviour and fisher decisions. This holistic understanding will be crucial for designing management measures that are tailored to context and create better outcomes for sharks and people.

4. Impact and lessons learned from an integrated conservation intervention for manta rays

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Photos: A majestic manta ray (top) and a struggling ex-manta fisherman (bottom).

“If the government prohibit us to catch manta, I just want to ask them, what is the trade off? What are their solutions for the livelihood?” – a manta fisher from Lamakera.

“Regulation will not always, and not alone, solve conservation problems” – Rosie Cooney.

4.1 Introduction

Mobulid rays (*Mobula* spp.) are a family of large, slow-growing cartilaginous fish (Class Chondrichthyes, subclass elasmobranch), which consist of nine extant species of devil ray and two species of manta ray (reef: *Mobula alfredi* and oceanic: *M. birostris*) (Dulvy, Pardo, et al., 2014; Lawson et al., 2017). Mobulids are captured and retained in fisheries throughout their range, as target and bycatch species, and sold to meet demand for their gills in Chinese medicine markets (Hau et al., 2016; O'Malley et al., 2017; Ward-Paige et al., 2013). Manta ray gills are the most highly-valued mobulid commodity in commercial trade, and this high value combined with their slow life-history makes manta rays particularly vulnerable to trade-driven extinction (Hau et al., 2016; McClenachan et al., 2016). With growing international concern regarding these threats, manta rays were listed on Appendix II of the Convention on the International Trade of Endangered Species (CITES) in 2013, which requires that Parties to CITES ensure any international trade in manta products is non-detrimental to their survival in the wild.

Within this international context, Indonesia is a global priority for manta ray conservation. Indonesia is the world's largest elasmobranch fishing nation, and has historically has been a major catcher and supplier of manta gills to consumer markets (Dent & Clarke, 2014; O'Malley et al., 2017). Indonesia is also home to the world's largest documented manta ray fishery – Lamakera in East Flores, East Nusa Tenggara Province (Lewis et al., 2015). However, CITES obligations, alongside concerns regarding the negative impacts of population declines on Indonesia's multi-million-dollar manta tourism (O'Malley et al., 2013), motivated the Government of Indonesia (GoI) to declare manta rays a protected species in 2014 (Booth, Pooley, et al., 2020). This was implemented through a Ministerial Decree by the Ministry of Marine Affairs and Fisheries (MMAF), which prohibits capture, retention and trade of manta rays throughout Indonesia's entire exclusive economic zone (MMAF No.4/KEPMEN-KP/2014, herein 'the manta decree').

In Lamakera, villagers have hunted marine megafauna for centuries (R. H. Barnes, 1996; Dewar, 2002). Historically, mobulid hunting was conducted in un-motorised artisanal vessels with harpoons, to provide meat for local subsistence. However, expansion and modernisation of fleets and the emergence of a commercialised industry for gills drove intensification of mobulid hunting. It is estimated that 1,050 – 2,400 manta rays were killed annually in Lamakera in the early 2000's, and these levels of exploitation have led to declines and extirpations of manta population (Dewar, 2002; Lewis et al., 2015). As a result, Lamakera began to attract international attention (e.g., Heinrichs, 2014, and the 2015 documentary 'Racing Extinction').

During 2014, GoI and a coalition of stakeholders (government agencies, non-governmental organisations (NGOs) and community groups), implemented a series of activities to address unsustainable manta hunting, and thus implement the manta decree, in Lamakera. Agencies involved include MMAF, the water police (PolAir), the Directorate General of Environmental and Forestry Law Enforcement (Gakkum) (under the Ministry of Environment and Forestry (MoEF)), and the local and provincial government of East Flores (DKP Flotim) and East Nusa Tenggara (DKP NTT), respectively. These agencies were supported by a partnership between Misool

Foundation and the Wildlife Conservation Society Indonesia Program (WCS-IP). Henceforth I refer to this group of governmental and non-governmental organisations as the project partners.

The ultimate goal of this collaborative effort was to reduce manta ray mortality, by changing human behaviour (i.e., stopping manta hunting) through effective compliance management. Compliance to conservation regulations can be strengthened through enforcement; however enforcement also tends to be challenging, costly and can result in perverse consequences (Arias, 2015; Borrión et al., 2019; Challender et al., 2015b; Keane et al., 2008). Context-specific social factors often determine the long-term success of marine conservation, with local institutions and positive incentives consistently recognised as important components of effective natural resource management (Arias et al., 2016; Brooks et al., 2012; Gutiérrez et al., 2011; Waylen et al., 2010). Efforts to instil pro-conservation behaviour also need to consider the multiple and diverse drivers of human behaviour, including extrinsic motivations (i.e., positive and negative incentives), intrinsic motivations (i.e., beliefs, social norms and cognitive biases) and complex interactions between the two (Ajzen, 1991; Bowles & Polanía-Reyes, 2012; Cinner, 2018; St John et al., 2010; Wright et al., 2016). As such, a cross-disciplinary approach to compliance management was necessary to reduce manta hunting in Lamakera, combining lessons from criminology and social psychology (Borrión et al., 2019; Oyanedel, Gelcich, & Milner-Gulland, 2020; St John et al., 2010). Given the complex situation, a non-experimental mixed-methods research design was required to assess impact.

In this study I use a theory of change, supported by five-years of empirical data, to evaluate the impact of efforts to reduce manta ray hunting in Lamakera from 2014-2018. In doing so I:

- 1) Outline a case study of an integrated cross-disciplinary conservation intervention that has had a measurable impact on illegal and unsustainable exploitation of marine megafauna,
- 2) Provide an example of a rigorous, mixed-methods approach to assessing the impact of a conservation intervention in a complex, real-world situation, where an experimental research design was unfeasible,
- 3) Identify several management recommendations, since this case has broad application to other conservation interventions seeking to mitigate trade-driven exploitation of megafauna. This is particularly relevant in the context of increasing regulation of international shark and ray (elasmobranch) trade under CITES, and the need to provide models of successful implementation at national- and local-levels.

4.2 Methods

4.2.1 Site

Lamakera is the collective name for two adjacent villages (Motonwutun and Wotobuku) in East Flores, East Nusa Tenggara (Figure 4.1). As of August 2014, approximately 2,500 people inhabited Lamakera in 661 households (Jaiteh, 2014a). In 2013, prior to the manta decree, the mobulid hunting fleet consisted of 40 boats, crewed by 5-8 people (approximately 350 people directly involved in hunting, representing ~50% of all households). Mobulids are targeted using harpoons, with ‘top hunters’ leaping off the bow of the boat to capture mobulids aggregating at the surface (Lewis et al., 2015). Most manta mortality occurs within a spatio-temporal hotspot (Figure 4.1), during

an eight-month season (March to October, peaking in July), and during 2-4 days per month, with *en masse* hunting events triggered by mantas aggregating at the sea surface around cleaning stations (Lewis et al., 2015). Aggregations typically take place during the new moon, related to manta feeding and cleaning behaviour, driven by oceanographic and environmental variables (Herwata, 2018; Herwata & Lewis, 2018). As well as hunters, four major gill traders were operating in Lamakera in 2013, acting as middlemen between hunters and exporters in Surabaya (Jaiteh, 2014b). Local women also made a living as *Papalele*: local processors and traders of mobulid meat and gills. Gross annual revenue from manta hunting and trade was estimated at IDR 1 billion (~US\$ 90,000) in 2014, providing an estimated monthly income of IDR 36,000 (\$32) per household per month (Jaiteh, 2014b; Lewis et al., 2015).

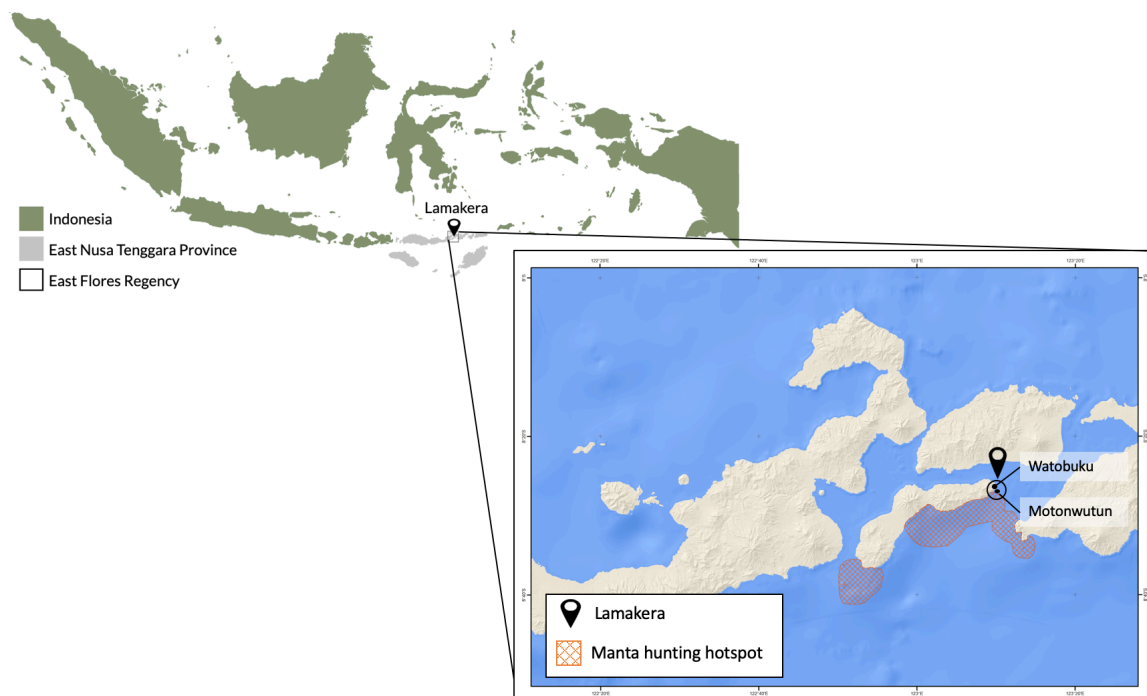


Figure 4.1 The location of Lamakera and the manta hunting hotspot (top left shows Indonesia and East Nusa Tenggara Province; bottom right shows East Flores Regency, and the specific locations of Watobuku and Motonwutun, which collectively comprise Lamakera)

4.2.2 Intervention design

Key events relating to manta ray conservation in Lamakera first began in 2013, when mantas were listed on CITES Appendix II (Booth, Pooley, et al., 2020) (Figure 4.2). From 2016, an adaptive management cycle was adopted for planning, implementing, analysing, adapting and sharing, based the Conservation Measures Partnership (CMP) guidelines (CMP, 2020). In 2016, a robust theory of change (ToC) was developed through a joint planning process by the project partners, which guided strategies and activities, and provided a hypothesis for theory-based impact evaluation (Figure 4.3).

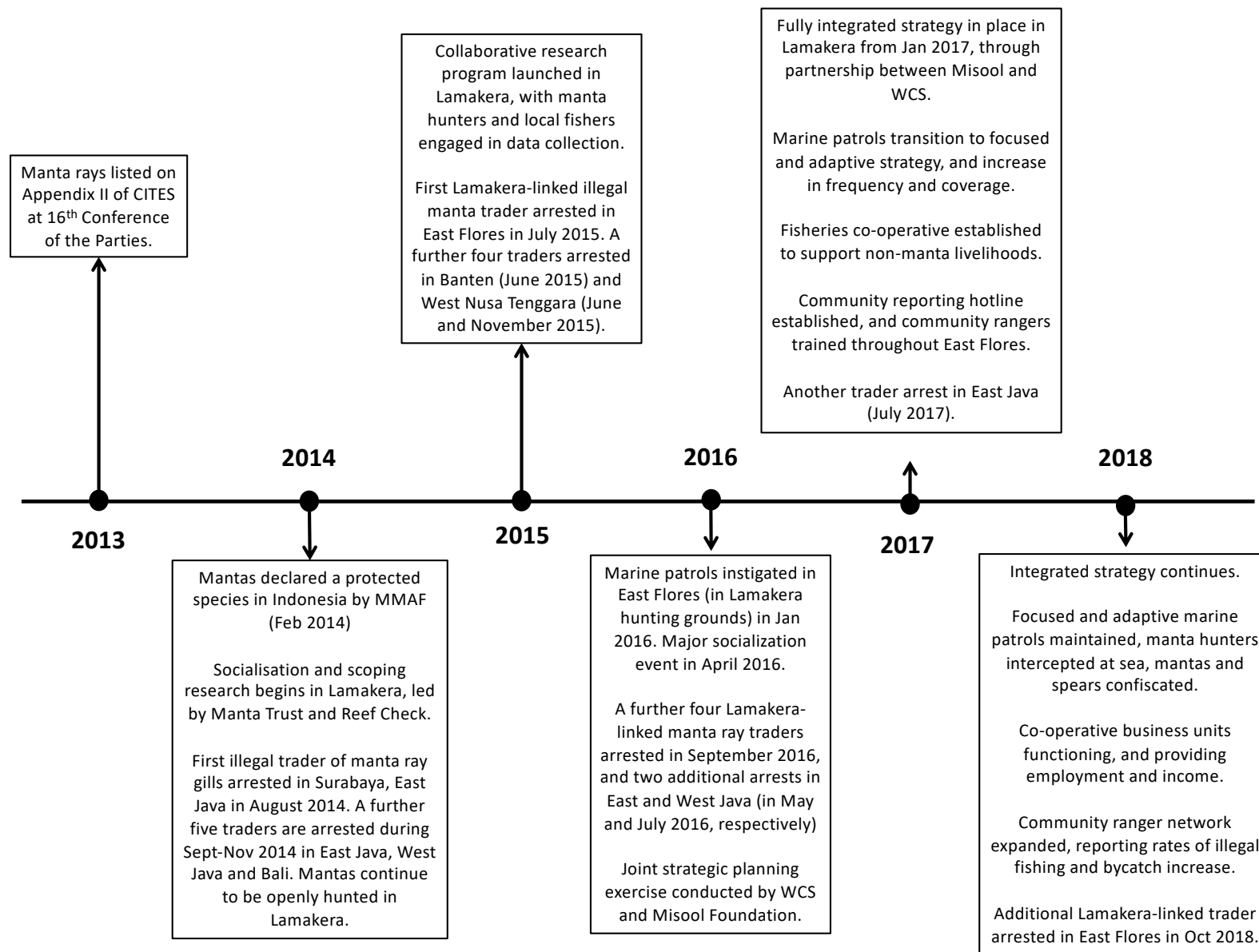


Figure 4.2 Timeline of key events relating to implementation of the manta decree in Lamakera

Intervention planning was grounded in criminology, social psychology and conservation science literature, and informed by empirical data on the ecological and socio-economic context in Lamakera. The ToC aimed to address several intrinsic and extrinsic motivators of hunter behaviour, including: a lack of deterrents, a lack of alternative income sources, a high reward, and community norms and perceived legitimacy (Figure 4.3). To do so, two broad strategies were developed:

- 1) A community outreach and livelihood-focused incentives strategy, which aimed to reduce barriers and create positive intrinsic and extrinsic motivations for compliance. Strategy design was informed by social psychology, behavioural economics and community-based conservation (Ajzen, 1991; Brooks et al., 2012; Cinner, 2018; St John et al., 2010; Waylen et al., 2010). Activities included awareness raising, collaborative research, community-based monitoring, and development of conditional non-manta livelihoods (Figure 4.3).
- 2) A species protection strategy, which aimed to create perceived net negative incentives for non-compliance. Strategy design drew from problem-oriented wildlife protection and situational crime prevention (Borrion et al., 2019; L. E. Cohen & Felson, 1979; Cornish & Clarke, 2003; Goldstein, 1979; Lemieux & Pickles, 2018; Petrossian, 2015), plus other examples of compliance management in conservation (Arias, 2015; Arias et al., 2015, 2016; Keane et al., 2008; Travers, Archer, et al., 2019). Activities included overt site-based marine patrolling, community-based monitoring, covert monitoring of illegal trade, and arrest and prosecution of major illegal traders. Importantly, implementation was based on adaptive crime analysis, which drew on a range of data sources (mobulid habitat use models, observed hunting patterns and community-based monitoring), to target enforcement towards hotspots, peak times and priority offenders. These activities worked synergistically to: reduce the perceived rewards of manta hunting, increase the perceived risks, increase the perceived effort, remove excuses and reduce provocations (Appendix 1, S4.1) (Cornish & Clarke, 2003).

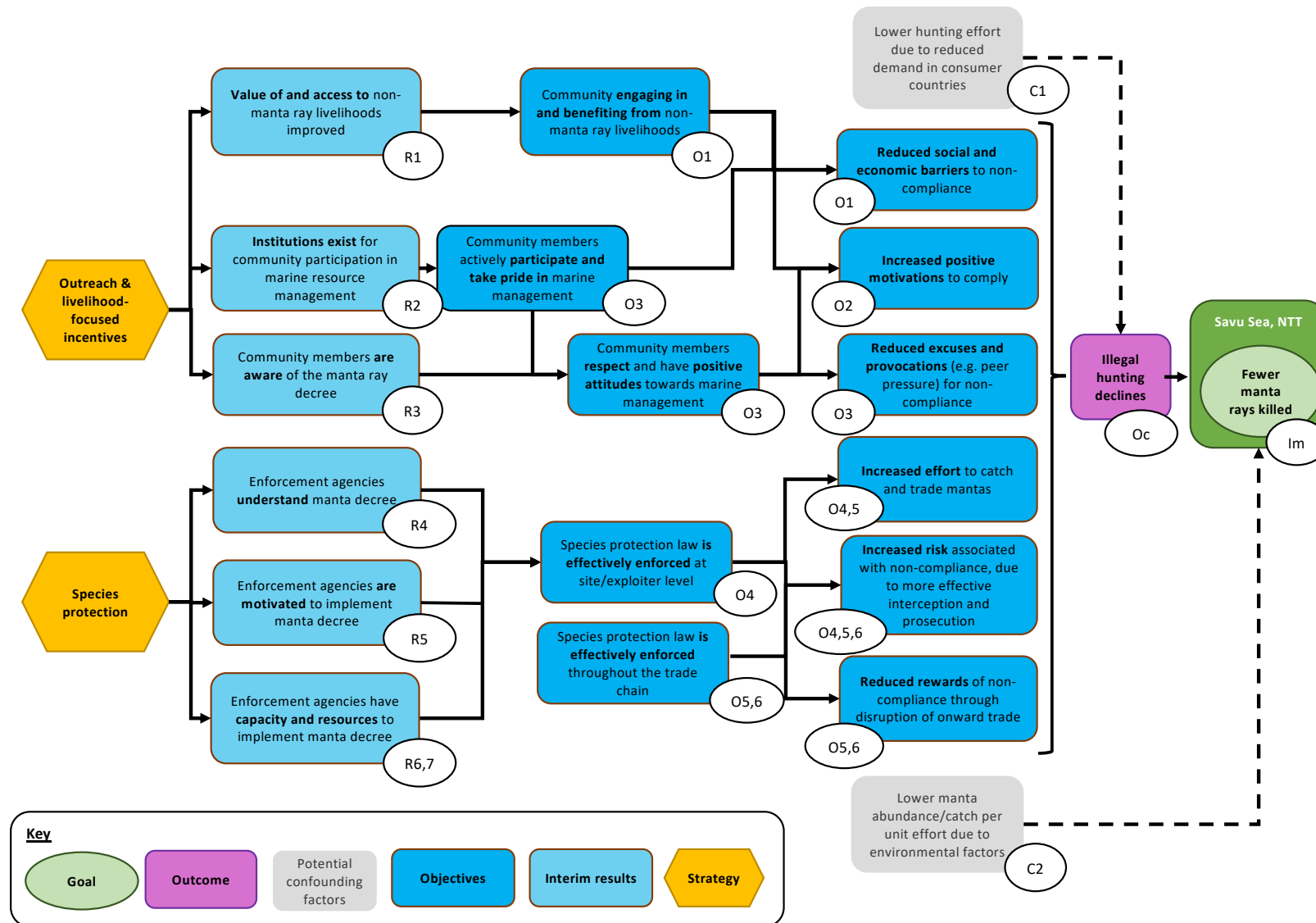


Figure 4.3 A simplified Theory of Change (ToC) for the Lamakera manta conservation intervention, demonstrating the hypothesised causal link between strategies, results, outcome and impact. This is based on a conceptual model of the key threats driving illegal hunting of manta rays in Lamakera. White circles indicate sources of evidence for verifying the ToC, which are described in Table 4.1 (R1-7 = results, O1-6 = objectives, Oc = outcome, Im = Impact)

4.2.3 Impact assessment

Research design

Given the context and complexity of this conservation intervention, it was not feasible to adopt an experimental research design. Rather, I adopted a theory-based method to assess impact, which is particularly useful when baseline data and sample sizes are limited, and can provide an in-depth understanding of how and why an intervention works (Margoluis et al., 2009; Salazar et al., 2019; H. White & Phillips, 2012; Woodhouse et al., 2016). The ToC (Figure 4.3) provides a testable conservation hypothesis to explain observed trends, with a null hypothesis that intervention activities did not cause any change in manta ray hunting and mortality. I verify the ToC and reject the null hypothesis through a mixed-methods approach, by analysing and triangulating empirical evidence for each step of the hypothesised causal chain (Table 4.1) and testing three key assumptions: A1) The hypothesised ToC is an accurate representation of the world, A2) Observed trends in empirical data are reliable. A3) observed trends are caused by the conservation intervention, as opposed to external confounding factors (Appendix 1, S4.2).

Data and analysis

Mixed-methods and triangulation are particularly important for this study, given the many potential biases (e.g., illegality and sensitivity) and confounding factors (e.g., environmental and market fluctuations) (Booth, Pooley, et al., 2020; Gavin et al., 2010).

Empirical evidence for the ToC was collected from a variety of existing data sources (Table 4.1; Appendix 1, S4.3). For data on interim results and objectives, I used NGO project reports and records from Misool and WCS-IP (evidenced by training records and sign-in sheets, signed community agreements, photographs and cooperative business records, see Appendix 1, S4.4); SMART patrol data; and government intelligence and law enforcement records (Table 4.1, Appendix 1, S4.3). For outcomes and impact (i.e., manta ray hunting effort and landings) I used SMART patrol data and landings monitoring data collected by WCS-IP and Misool Foundation enumerators (following survey methods from White *et al.* (2006), as well as historic published landings data from Dewar (2002) and Lewis et al. (2015) (Table 4.1, Appendix 1, S4.3).

I used several techniques to establish causal inference and attribution, including statistical regression, pre/post comparison, counterfactual modelling, natural experiments, and process tracing to understand and explain observed trends (A. Bennett, 2010; Booth, Pooley, et al., 2020; Margoluis et al., 2009; Scriven, 2008). I conducted statistical analyses of trends using linear modelling, and tested for correlations between objectives, outcomes and impact to substantiate proposed causal links (where changes in numbers of manta rays killed is the impact indicator). More specifically: For law enforcement trainings (R4) and number of traders arrested and prosecuted (O5), I used a paired t-test to compare average fines and prosecutions before and after the onset of trainings (i.e., pre/post comparison). For hunting effort (outcome) and numbers of manta ray killed (impact), I used regression analysis, fitting negative-binomial models (Zuur, et al. 2009) to assess the influence of daily and monthly species

protection activities (i.e., patrols) on manta hunting and manta catch. Pre-decree daily landings data were not available, as such the models only included post-decree/pre-intervention data (May 2015-Dec 2015) and post-integrated intervention data (Jan 2016-Dec 2018) (See Appendix 1, S4.5 for detailed methods). I used counterfactual thinking to support inference and attribution, based on two types of counterfactuals: 1) devil rays as a natural experiment or quasi-counterfactual and, 2) a modelled counterfactual of predicted manta ray mortality in the absence of the intervention. I first conducted an events-based analysis of manta mortality, using linear modelling of daily landings (May 2015-Dec 2019) to assess whether there had been a significant decline in manta ray hunting events and total mortality over time, and if this decline was temporally associated with the intervention (i.e., pre/post comparison). Since devil rays are exploited for the same market as manta rays but are not a protected species, I used devil ray landings as a natural experiment to indicate external mobulid market fluctuations (C1) (Booth, Pooley, et al., 2020). I considered all meaningful models and used minimum AIC to determine the best-fit (Burnham & Anderson, 2003) (Appendix 1, S4.5). For models with multiple variable contrasts I applied a post-hoc generalised linear hypothesis test (GLHT), which enables multiple comparisons of factor means and confidence intervals simultaneously, to confirm which specific factor (or factor combinations) is significant (Appendix 1, S4.5). This ensures that the contrast relates to differences between the of interest (i.e., comparisons between manta pre- vs. manta post- and devil pre- vs. devil post-), and not the overall level of the response variable (Meier, 2020). Secondly, I used a General Additive Model (GAM) developed by Herwata et al. (2018) as a modelled counterfactual for estimated levels of targeted manta mortality in the absence of the intervention (C2). This model used pre-intervention data to predict trends for targeted manta catch per unit effort (CPUE), based on environmental variables (Appendix 1, S4.5). I then compared these predicted trends with empirical observations of actual post-intervention CPUE. I used Rstudio software for all statistical analyses (RStudio Team, 2020), see Appendix 1, S4.5 for the list of packages. Finally, for comparison with pre-decree data, which could not be included in the statistical analysis due to different sampling methods (2002-2014, Dewar, 2002; Lewis et al., 2015) I provided a descriptive overview of trends in total annual landings over time, which I mapped against a timeline of events.

Quantitative data were supplemented with qualitative evidence on contextual factors, such as onset of activities, external market trends and the perceived reliability of available data. This enabled triangulation and verification of observed trends, and substantiation of assumptions through critical assessment of the inferential weight of available data and plausible alternative explanations (Appendix 1, S4.2) (A. Bennett, 2010; Booth, Pooley, et al., 2020; Scriven, 2008).

Table 4.1 Summary of available evidence for ToC (in bold) and associated data sources (in italics) used to assess project impact at each stage of the ToC. Numbers correspond to stages in the ToC in Figure 4.2.

Strategy	Interim results	Objectives	Outcome	Impact
Outreach and livelihood-focused interventions	R1. Number of business units developed for non-manta livelihoods (2014-18) <i>from official government cooperative records</i>	O1. Number of community members benefiting from new non-manta livelihoods (2016-2018) <i>from Misool sign in sheets and cooperative records</i>		
	R2. Number of community members engaged in institutions for marine monitoring and management (2014-18) <i>from Misool training records and meeting sign-in sheets</i>	O2. Number of (ex) manta ray hunters committed to complying to regulations (2014-18) <i>from signed community agreements</i>		
	R3. Number of socialisation and training events, and estimated village coverage/attendance (2014-18) <i>from Misool and WCS-IP project records and sign in sheets</i>	O3. Frequency of marine monitoring reports from community (2016 -2018) <i>from reporting hotline records</i>	Illegal hunting effort (2016-18) <i>from WCS-IP SMART patrol data and Misool enumerators</i>	Number of manta rays killed (2013-18) <i>from overt and covert observation of catch collected through WCS-IP and Misool enumerators</i>
Species protection	R4. Number of law enforcement trainings conducted (2014-18) <i>from WCS-IP project reports and sign-in sheets</i>	O4. Number of illegal fishers intercepted and punished (2016-18) <i>from WCS-IP SMART patrol data</i>		
	R5. Number of media articles published (2014-18) <i>from WCS-IP online media monitoring records</i>	O5. Number of illegal traders arrested and prosecuted (2014-18) <i>from law enforcement records</i>		
	R6. Days of marine patrols conducted (2016-18) <i>from WCS-IP SMART patrol data</i>	O6. Amount of fines and prosecutions levied against illegal traders (2014-18) <i>from law enforcement records</i>		
	R7. Number of traders investigated (2014-18) <i>from WCS-IP intelligence database</i>			

4.3 Results

4.3.1 Activities and objectives

Outreach, consultation and awareness raising

Between 2014-2018, 11 official socialisation events relating to marine conservation and the manta decree were conducted in East Flores, based on WCS-IP and Misool Foundation project records. Of these, five were conducted in Lamakera. Notably, a high-level government event in April 2016 was jointly organised by MMAF, Misool Foundation, WCS-IP, the Islamic Council Indonesia (MUI) and the water police (PolAir), to reinforce and

legitimise manta ray protection (Appendix 1, S4.4). This event was attended by roughly 1,000 villagers, representing almost all Lamakera's households. Informal meetings were regularly conducted across 50 villages in East Flores from 2016 onwards. These meetings were held 1-2 times per month in the 20 most actively-engaged villages, and once every 2-3 months in the 30 remaining villages. Based on attendance records and other informal community engagement activities, Misool staff estimate that 100% of Lamakeran villagers were aware of the manta decree by 2016. This substantiates R3 in the ToC: community members are aware of the manta ray decree (Figure 4.3).

To raises awareness beyond Lamakera, over 1,000 media articles were published in the national and international media during April 2015-December 2018, highlighting GoI's efforts to combat marine wildlife crime (Appendix 1, S4.4). Several of these publicised high-level cases of illegal manta trade, which emphasised GoI's commitment to protect manta rays to, made information on prosecutions and punishments widely available, and gave national and international recognition to the officials involved in the cases. This provides evidence for R3 and R4 in the ToC, that both community members and government agencies are aware of and understand the manta ray decree (Figure 4.3).

Community surveillance

During 2017-2018 a network of community surveillance groups (referred to in Indonesia as *Kelompok Masyarakat Pengawas* or pokmaswas) was established throughout East Flores to conduct community-based monitoring. A system was established for pokmaswas to report incidents of illegal fishing and megafauna bycatch, via a reporting hotline. This was coupled with a rapid response team, led by MMAF and PolAir, to respond to reports. Based on Misool Foundation project records, the number of people engaged in pokmaswas groups grew from 30 across 5 villages in 2017 to 470 across 50 villages in 2018. Pokmaswas reports increased from an average of 2.5 per month in 2017 to 20 per month by 2018, leading to the successful live release of 42 vulnerable marine animals during 2016-2018, including manta rays, whale sharks, sunfish and turtles (Appendix 1, S4.4). Participating community members received non-monetary awards (certificates) for reporting to the hotline, and recognition in a local newsletter every time a bycatch incident was reported. This substantiates R2 and O3 in the ToC, that institutions exist for community participation in marine resource management, and that community members actively participate and take pride in marine management, with positive attitudes towards marine conservation, and reduced provocations for non-compliance (Figure 4.3).

Livelihood-focused incentives

Following market research and community consultation, a sustainable fisheries cooperative was established in Lamakera in 2017 (Appendix 1, S4.4). Members benefited from access to resources and capital for developing non-manta livelihoods, which focus around five business units: a mini purse seine fishing vessel, ice supply, micro-finance loans, fish drying and a mini-market offering basic provisions at a discounted rate. Cooperative membership was contingent on signing a 'no megafauna hunting' agreement, ratified by the local government,

which links benefits to pro-conservation behaviour (Appendix 1, S4.4). Based on Misool Foundation project records, the cooperative recruited 107 members during 2017-2019, 28 of which were ex-manta ray hunters. Of this, the mini purse-seine crew comprised 14 ex-manta hunters, who each received earnings of IDR 500-750,000 (\$35-\$53) per month. This salary is equivalent to the estimated monthly income from manta hunting during peak seasons, and was more stable by comparison. A further five people were employed as full-time staff within the cooperative's business units. Microfinance loans were distributed to 59 people, including 23 *Papalele*. The loans supported development of non-manta small enterprises. This substantiates R1 and O1 in the ToC: the value of and access to non-manta livelihoods is improved, and communities engage in and benefit from non-manta livelihoods, with reduced barriers and positive incentives to comply to the manta decree (Figure 4.3).

Marine patrols and interception of illegal fishers

Marine patrols began in 2016, with an average of 2.9 patrol days per month. Initially, patrols were ad-hoc, but analysis of multisource crime data supported a new patrolling strategy for 2017. Based on WCS-IP SMART patrol data, patrol effort increased significantly in 2017 and 2018 (Table 4.1; Figure 4.4; Appendix 1, S4.4), aided by the deployment of two new fast patrol boats, which reduced travel and response times, increased patrolling area per unit time, and boosted morale of patrol staff. Patrol focus also improved, with patrols directed towards known spatio-temporal hotspots (Figure 4.4). Patrol units operated using a dual strategy, with preventative routine patrols direct towards hotspots, and rapid-response patrols based on real-time reports of hunting or bycatch from project partners and pokmaswas. This dual strategy, with continuous data collection and crime analysis, enabled patrols that were focused and adaptive. A total of 483 boat inspections were carried out from 2016-2018, leading to interceptions of 10 manta hunting vessels, as inferred by evidence of harpoons or manta body parts. Harpoons and manta parts were confiscated and administrative sanctions (i.e., warnings or confiscation letters) were given (Table 4.1). This substantiates R5, R6 and R7 in the ToC, as well as O4 and O5: enforcement agencies have capacity, resources and motivation to implement the manta decree, and species protection is effectively enforced at the site level, thus increasing the effort and risk to catch mantas (Figure 4.3).

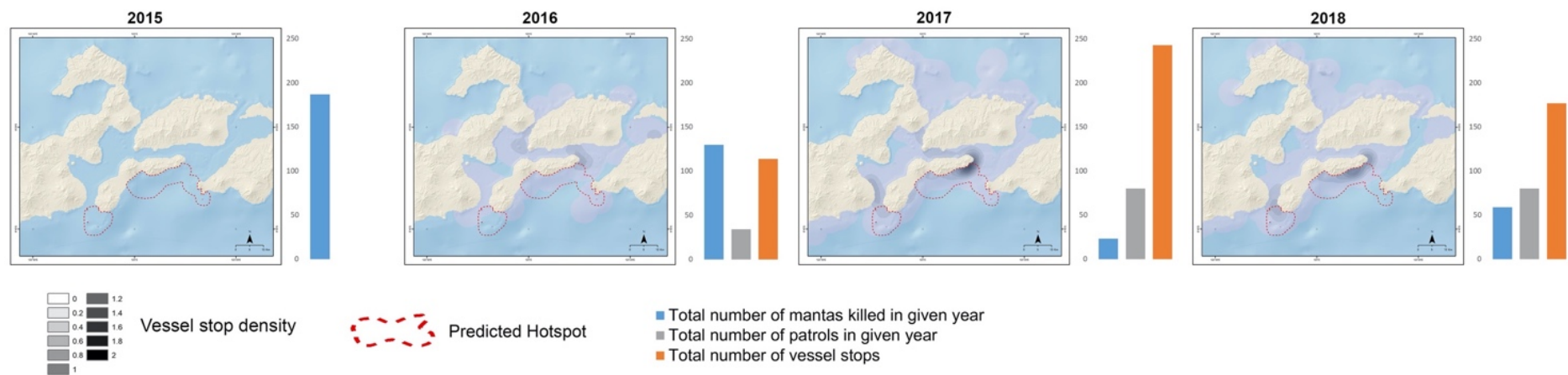


Figure 4.4 Changes in patrol intensity and focus from 2015 (no patrols) to 2018 (high vessel stop density in predicted hotspot locations)

Trader investigations and arrests

From April 2014 to December 2018 project partners collaboratively mapped data on illegal traders. 124 suspected illegal traders were identified, linked to nine geographically-clustered syndicates operating across 14 provinces (Table 4.1; Appendix 1, S4.4). This informed targeted enforcement actions by GoI, leading to 69 marine wildlife crime cases involving 86 offenders during 2014-2018, according to official enforcement records. Of these, 23 cases involved seizures of manta products, with the arrest of 25 suspects and an estimated 4,500kg of manta products seized. 11 of these cases have led to successful fines and prosecutions, resulting in more than US\$22,500 levied in fines and 24 months of jail time (Table 4.1; Appendix 1, S4.4). Six cases (in July 2015, September 2016 and October 2018) involved traders with known links to Lamakera, who were facilitating onward trade and trafficking to international markets. This substantiates R4-R7 in the ToC: enforcement agencies understand, and have capacity, resources and motivation to implement the manta decree; and O4-O6: the manta decree is effectively enforced throughout the trade chain, thus increasing the effort and risk of catching and trading mantas, and reducing the rewards of illegal trade, through enforcement disincentives (Figure 4.3).

Enforcement training and sanctions

From August 2015-December 2018 a series of 18 training workshops were conducted on marine wildlife law enforcement, and in-house training modules for public prosecutors and environmental judges were established (Appendix 1, S4.4). A total of 645 fisheries and law enforcement officers including MMAF, police, customs, aviation security, the National Agency of Drug and Food Control (NADFC), judges and public prosecutors were trained (Table 4.1). Onset of the training series coincides with an observed increase in average punishments for traders of protected marine species, with a statistically significant increase in average fines before vs. after the onset of trainings (paired t-test, $p < 0.05$ (Appendix 1, S4.4)). This provides evidence for causal links between R4-R7 and O5-O6, indicating that training improved capacity and motivation of enforcement agencies, leading to increasing punishments (and thus reduced rewards) for illegal trade (Figure 4.3).

4.3.2 Outcomes and impact

Observed trends

Overall, targeted manta ray mortality (i.e., mantas killed using spears) reduced by over 90%, and total mortality (i.e., including mantas killed using spears, and mantas captured in gill nets) reduced by 86% by 2018, in comparison to the 2013 (pre-decree) baseline (Figure 4.5). This equates to an estimated 400 manta rays saved since 2014, assuming constant fishing mortality would have occurred in the absence of the decree and intervention.

Observed trends in daily landings data (2015-2018) indicated that *en masse* hunting events declined significantly following onset of the integrated intervention. There was a significant negative relationship between year and likelihood of a hunting event (coefficient = -0.26, $p < 0.01$), and significant declines in hunting events in 2017 (coefficient = -1.17, $p < 0.001$) and 2018 (coefficient = -0.53, $p < 0.05$) vs. 2015 (Table 4.1; Appendix 1, S4.4, S4.5). *En masse* hunting events appeared to halt by 2017 and 2018, and hunting patterns shifted to become less

frequent, less severe (in terms of number of mantas killed) and generally more one-off or opportunistic (Appendix 1, S4.4, S4.5). Linear models of recorded mortality over time indicated a significant negative relationship between year and number of manta rays killed (coefficient = -0.72, $p < 0.001$; Appendix 1, S4.4). Mortality in all post-intervention years (2016, 2017 and 2018) was significantly lower than 2015 (Table 4.1; Figure 4.5; Appendix 1, S4.4, S4.5). This substantiates the outcome (illegal hunting declines) and impact (fewer manta rays killed) in the ToC (Figure 4.3).

Attribution and assumptions

Negative binomial models of manta landings indicate significant negative relationships between the number of mantas killed and implementation of site-based patrolling activities. In particular, the introduction of patrols by government enforcement agencies, improved patrol focus (i.e., a data-driven patrol strategy, targeting patrols towards hotspots) and the interaction between the two are all significant in explaining declines in landings ($p < 0.05$) (Appendix 1, S4.5).

Linear models indicate a step-change in the number of mantas killed occurred in 2017 (Figure 4.5, Appendix 1, S4.5). This step-change temporally coincides with the onset of the integrated strategy, with focused patrols and arrests of Lamakera-linked traders. A similar decline was not observed for devil rays, rather, 2017 and 2018 saw increases in devil ray catches (Figure 4.5, Appendix 1, S4.5). This suggests the declines in manta landings cannot be attributed to fishers moving out of the mobulid fishery due to external factors (e.g., declines in demand for mobulid products). This is also supported by market surveys showing continued sales of mobulid gills in China and Hong Kong during this period (Hau et al., 2016; O'Malley et al., 2017). Modelled predictions of CPUE based on environmental variables shows there is a significant difference between predicted and empirically observed CPUE (Figure 4.5, Appendix 1, S4.5). This suggests trends cannot be attributed to environmental variables alone.

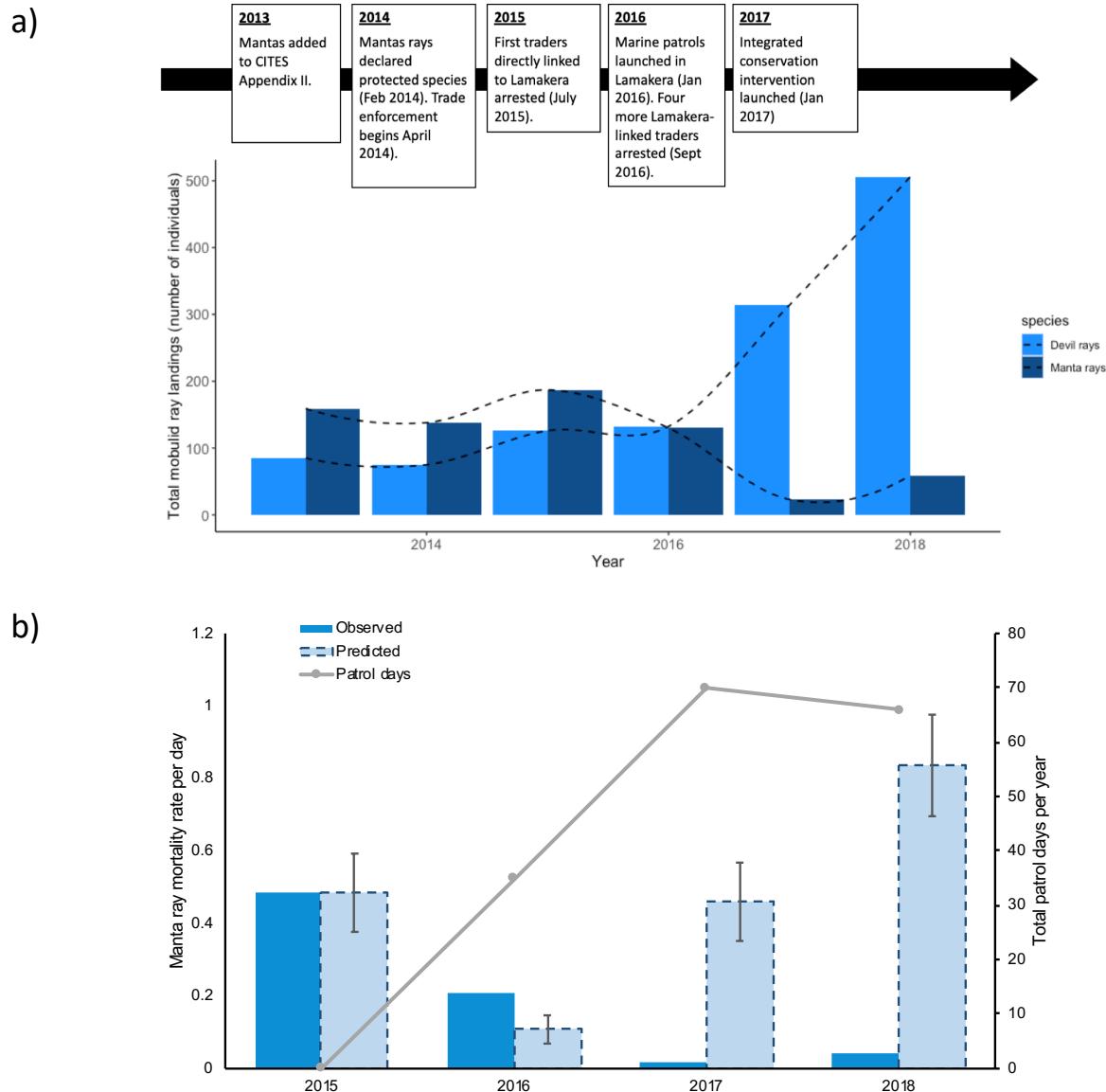


Figure 4.5 Observed trends in number of manta rays killed, in comparison to counterfactuals, and including contextual considerations, such as the timing of intervention activities. A) Illustrates an events-based analysis of total landings of devil and manta rays over time. A step change decline in manta ray landings occurred in 2017 ($P < 0.01$, $z = -3.21$), with increase in devil ray landings from 2016. B) Illustrates modelled prediction vs. actual observations of manta catch rates, alongside trends in number of patrol days per year. Mean observed manta ray catch rates per year are lower than predicted based on pre-intervention data.

Data on contextual considerations indicates several sources of bias and external factors that could have influenced observed trends in manta hunting and mortality: 1. Manta hunting was displaced to other locations with no monitoring or enforcement; 2. Manta hunting and landings became clandestine, due adaption of offenders in response to enforcement; 3. Manta catch declined due to environmental variables, natural population stochasticity or a population collapse; 4. Manta hunting effort declined due to reduced consumer demand. However, I can largely discount these alternative explanations. Manta hunters are part of a small and highly localised community. Hunting occurs in near-shore waters, using small boats which are ill-equipped to travel long distances to other waters. Manta hunting also uses specialised equipment (harpoons) and occurs *en masse* during manta aggregations, which makes the behaviour relatively conspicuous and predictable. News of illegal hunting is usually relayed to

pokmaswas and other reliable sources. Although there is no robust data on the status of the manta population, there are still regular sightings of mantas, as well as accidental bycatch of mantas in gill nets, which is evidence that the population is extant (though may well be reduced). Finally, the increased targeting of devil rays, and reported rises in consumption of mobulid gill plates (Hau et al., 2016; O'Malley et al., 2017) indicate that the international market persisted at this time.

4.4. Discussion

This study presents an example of an integrated conservation intervention, designed to combine law enforcement and livelihood-focused incentives to deliver measurable conservation impact. The planning process and management recommendations offer insights for the design, implementation and evaluation of other conservation interventions, particularly those focusing on reducing illegal or unsustainable behaviour, or exploitation of marine megafauna. The impact assessment also provides an example of a non-experimental research design, using a theory-based mixed-methods approach, which could be applied to similar interventions where experimental designs are unfeasible.

4.4.1 Conservation impact

The results show that there were significant observed declines in the frequency and severity of manta ray hunting from 2013 to 2018, with up to 400 manta rays saved by 2018 compared to the 2013 baseline. Given that an individual manta ray is estimated to be worth up to US\$ 1-million to marine tourism over its lifetime, this also equates to up to US\$ 400-million in global tourism value (O'Malley et al., 2013). Evidence substantiating the steps in the ToC and associated assumptions suggests that these observed declines are primarily due to a real reduction in hunting and manta mortality, as opposed to monitoring bias and external factors. Significant differences between observed trends and counterfactuals (i.e., modelled predictions and devil ray landings) and temporal coincidence and correlation of observed trends with intervention activities (i.e., trainings and marine patrols) show it is reasonable to attribute these trends to the conservation intervention. This suggests I can reject the null hypothesis, that intervention activities did not cause any change in manta ray hunting and mortality. However, the extent to which these declines can be attributed to the intervention is difficult to quantify, particularly in the absence of robust data on the status of the manta ray population, hunting in un-monitored seascapes, and comprehensive market demand data, which could all have influenced observed trends to some degree.

I also acknowledge that these findings focus on a specific site (Lamakera in East Flores, Indonesia). While this does provide an example of a successful site-based conservation intervention in Indonesia's largest known manta fishery, it cannot be assumed that the manta ray regulation has led to effective conservation outcomes throughout Indonesia. There could be displacement of hunting and trading effort to other sites that lack monitoring and implementation effort. In addition, the observed increase in manta ray mortality in 2018, and the significant increase in devil ray mortality in 2017 and 2018, highlight the complex and evolving nature of threats facing mobulids. Enforcing natural resource regulations can lead to 'arms races' with offenders continually responding

to enforcement tactics, and the need for enforcement to continually adapt (S. C. Anderson et al., 2011; Jenkins, 2006). The 2018 increase in manta mortality is predominantly attributed to 'bycatch' in gill nets as opposed to targeted spear hunting, which may represent perceived loopholes or adapted tactics, or may be the result of genuine increases in accidental catch due a small recovery in the manta ray population. Preliminary information from project partners in 2019 suggested this trend is continuing, as offenders adapt their tactics in response to enforcement. This highlights the need for long-term investments, with continuous learning and adaptation to emerging challenges.

Increases in devil ray catch is also cause for concern, since these species are currently unprotected in Indonesia, and may now be the focus of unsustainable targeted fisheries. This also highlights how protection of some species or sites can lead to displacement of threats towards other vulnerable species or sites (Agardy et al., 2011; Suuronen et al., 2010). These substitution effects may be particularly acute in the absence of other legal, sustainable, comparable alternatives to manta hunting. In Lamakera it is unsurprising that effort has shifted towards devil rays, since this represents a familiar activity with a similar social and economic function. The transition is therefore much less frictionless than turning to entirely new fishing gears or target species. These findings reinforce previous work on the importance of understanding the function of environmentally-damaging behaviour, and ensuring that livelihood-focused interventions are suited to people's capacities and aspirations, and have real market appeal (IMM, 2008; Wright et al., 2016). In this case, devil rays hunting may represent a natural substitute activity, with unintended consequences for these species if management is not strengthened in the future.

4.4.2 Management implications

This case study highlights the importance of adopting multi-faceted, adaptive interventions when attempting to reduce illegal and unsustainable behaviour. Significant declines in illegal hunting and landings were only observed when focused marine patrols were launched and a fully integrated strategy was adopted. Prior to this, national regulations and trade enforcement alone did not appear to significantly reduce the number of manta rays killed at the site-level in Lamakera (Figure 4.5) (Booth, Pooley, et al., 2020). It is well known that a diversity of drivers and barriers influence individual and group compliance to conservation regulations, and that these are often heterogeneous within a given community (Cinner & McClanahan, 2006; Keane et al., 2016; Oldekop et al., 2010; Wright et al., 2016). In Lamakera, most community members were willing to comply with the manta decree, while a small number of highly-invested individuals were less willing to comply and continued to offend. As such, it was important to acknowledge heterogeneity in attitudes and behaviours, and address different drivers and barriers to compliance. By combining efforts to promote positive attitudes and norms for pro-conservation behaviour, motivate compliance through incentives, and reduce barriers to compliance, these different strategies worked synergistically to deliver impact at scale. This also supports the utility of combining individual and situational approaches to wildlife crime prevention, in which multiple techniques are adopted to simultaneously address individual motivations, as well as the immediate environment/situation (Oyanedel, Gelcich, & Milner-Gulland, 2020).

In terms of enforcement, the significance of focused marine patrols in influencing the number of manta rays killed, and the interaction between patrol frequency and focus (Figure 4.4; Figure 4.5; Appendix 1, S4.4, S4.5), highlights that both patrol quality and quantity are needed to effectively enforce species protection laws. This is consistent with previous studies showing that efficient distribution of patrol effort in space and time is important for optimising enforcement and compliance in a marine conservation context (Arias et al., 2016), particularly when patrols areas are large and detection probabilities can be low. This also supports the utility of problem-oriented approaches to wildlife protection (Borrion et al., 2019; Lemieux & Pickles, 2018) in which enforcement resources target specific spatio-temporal hotspots and key offenders, in order to address a specific problem or cluster of incidents (Goldstein, 1990). Adoption of overt and covert approaches also enabled disruption of illegal activity at multiple levels of the trade chain, from choking supply at the site-level to blocking onward trade and export within the broader trade network. Coupling this with support of judicial processes to strengthen criminal sanctions created a strong, credible deterrent against non-compliance. Crucially, success was facilitated by a clear pre-existing legal framework for manta ray protection, and a national-level mandate to act. Species protection actions would not have been possible without commitment and buy-in from key agencies and institutions at multiple levels within GoI.

Alongside enforcement, the role of community outreach and engagement should not be overlooked. There is both a moral and practical impetus for paying attention to the socioeconomic dimensions of illegal behaviour (Arias, 2015; Keane et al., 2008; St John et al., 2010). In Lamakera, community engagement played a role in gathering information, building awareness and goodwill, improving relationships between different stakeholders, and incentivising compliance. These activities can also help to ‘capture hearts and minds’ for longer-term intrinsic motivations to take shape (Blomley et al., 2010; Bowles & Polanía-Reyes, 2012; Gneezy et al., 2011).

The amount of information available to direct species protection activities was also important, with scoping activities prior to 2016 helping to build a robust evidence-base for designing intervention actions. As demonstrated, this enabled data-driven conservation planning, as well as timely feedback on the success or otherwise of those actions, and continuous adaptive management. By investing in many forms of monitoring project partners had an on-going ‘pulse check’ via indicators which were sensitive to change and meaningful, and could therefore quickly react to emerging threats to ensure the strategy remained preventative and resilient. This also provides benefits for impact assessment, as it creates multiple datasets which can be triangulated. This is particularly important for illicit, sensitive behaviour such as wildlife trade, which may be clandestine, such that monitoring data is subject to multiple forms of bias. Combining multiple methods and datasets can help to identify and circumvent the biases associated with monitoring illegal behaviour (Booth, Pooley, et al., 2020; Gavin et al., 2010; Ana Nuno & St. John, 2014).

Though this project benefitted from a wealth of monitoring data, regular analysis, learning and adaption is required for data to be useful for adaptive management. The seasonality of manta hunting gave project partners a period of inaction and review each year, creating time for the ‘Analyse and Adapt’, and the ‘Share’ steps of the project management cycle (CMP, 2020), and for new operations and tactics to be planned for the upcoming season. Not many conservation sites benefit from seasonality, but learnings from this case study underlines the value of regular review and learning.

Finally, from an ethics perspective, when implementing an integrated intervention, it is also important to maintain separation between enforcement teams, outreach and livelihood teams, and research teams. This helps to maintain the safety, integrity and reputation of project team members; ensure that trust and relationships within the community are not abused and undermined; and guarantee that any social research remains confidential, ethical, and does not cause harm to human subjects (Brittain et al., 2020).

4.4.3 Research implications

This study also provides lessons for assessing the impact of conservation regulations in cases where experimental research designs are unfeasible. Measuring the impact of complex interventions can benefit from a comprehensive theory of change, with collection and analysis of empirical data at each step to demonstrate change, and clear causal links between project activities and desired conservation outcomes (Salazar et al., 2019; Woodhouse et al., 2016). Where factors influencing compliance to regulations are multiple and dynamic, it is important to gain a comprehensive understanding of the broader socio-ecological system beyond the site of project implementation, and trends in distant drivers. Modelled counterfactuals or natural experiments can aid this process. Where behaviour is illicit or sensitive, it can be useful to triangulate multiple datasets, and supplement empirical observations with qualitative information on data collection processes and physical and social context (Salazar et al., 2019; Woodhouse et al., 2016). Rather than taking observed trends at face value, this mixed-methods approach led to improved understanding of the systems and contexts that created observed trends, and thus acknowledgment of uncertainties, bias and confounding variables; and strengthen causal inference and attribution (Booth, Pooley, et al., 2020; Burn et al., 2011).

4.4.4 Future directions

Overall, this study has shown that data-driven and cross-disciplinary approaches to conservation can reduce illegal and unsustainable exploitation of megafauna, through addressing the multiple and diverse drivers of human behaviour. This supports previous research on the value of: understanding context, robust project design, working with and building capacity of local institutions, and on-going adaptive management (Brooks et al., 2012; Lejano & Ingram, 2010; Travers, Selinske, et al., 2019; Waylen et al., 2010). However, with new emerging challenges, I also emphasise the need for long-term commitments, strong partnerships with diverse local stakeholders, and commitments to capturing and sharing learning. Together, these approaches can build resilient and adaptive institutions for enduring impact.

5. A socio-psychological approach for understanding and managing bycatch in small-scale fisheries



Photo: Me interviewing a fisher in Lhok Rigaih.

“Dia bawa rezeki” (she brings the gifts) – a fisher in Lhok Rigaih, on whale sharks.

5.1 Introduction

Bycatch of marine megafauna is common and particularly problematic in small-scale multi-species fisheries, where unselective gears are used to opportunistically catch a variety of fish (Gupta et al., 2020; Shester & Micheli, 2011). In general, but in these types of fisheries in particular, bycatch is poorly defined, because perceptions of target and non-target vary widely (Davies et al., 2009). In practice, marine megafauna bycatch occurs on a spectrum (Figure 5.1), from undesirable incidental catch, which can be costly to fishers, to valuable retained secondary catch, which fishers may secondarily target. When seeking to manage bycatch in a particular context, it is important to diagnose where along this spectrum a given species falls, to design interventions that can effectively change bycatch-relevant behaviour for that species. For example, if bycatch is undesirable, a low-cost technical fix may be easily adopted (M. A. Hall, 1996; Piovano et al., 2012). However, if bycatch mitigation has opportunity costs, incentives or compensation may be required to promote uptake of bycatch mitigation measures (Matwal et al., 2014; Wosnick et al., 2020) (Figure 5.1). In SSFs the latter is often the case, as incidentally-caught species are sold as a part of overall livelihood strategies, or consumed for subsistence purposes (Glaus et al., 2018; Gupta et al., 2020). This issue is ubiquitous throughout coastal areas, and represents a cross-disciplinary challenge for biodiversity conservation, food security and livelihoods.

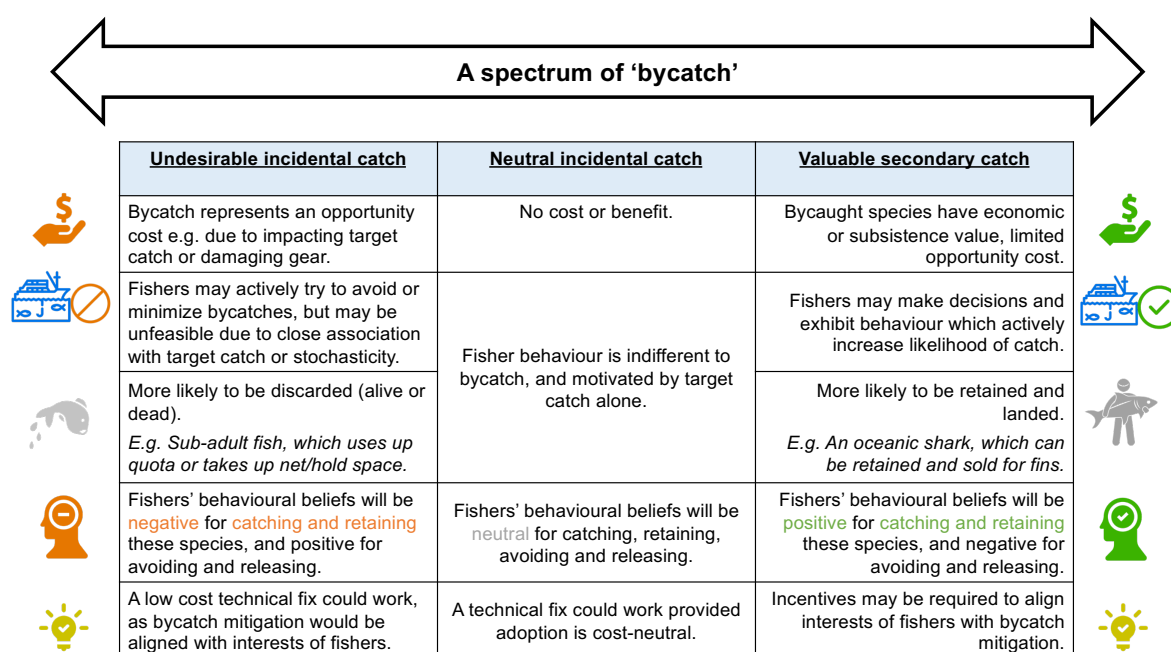


Figure 5.1 A spectrum of bycatch, from undesirable incidental catch which has opportunity costs to useful secondary catch, which may have economic or subsistence value. Fishers' underlying beliefs, and appropriate interventions to change fisher behaviour, will be

At its core, diagnosing and managing bycatch requires understanding and changing fisher behaviour, yet the integration of behavioural sciences into marine conservation and fisheries management remains limited. For example, in a recent systematic review, Andrews et al. (2021) noted that socio-psychological approaches for understanding and explaining fisher behaviour are under-used, with a need for theoretical models and novel insights to inform governance and policy. To address this gap, I offer a simple socio-psychological approach for

understanding and managing bycatch, drawing on a well-established theory from behavioural sciences: the Theory of Planned Behaviour (TPB) (Ajzen, 1991, 2011). I illustrate its utility with empirical data from a multi-species multi-gear SSF in Indonesia, which is representative of other tropical SSFs in the Global South, and show how the results can be used to diagnose and manage bycatch. Finally, I outline implications and ways forward for adopting socio-psychological approaches to design more effective and socially-just management interventions for bycatch in the future.

5.2 Methods

5.2.1 A socio-psychological approach to bycatch

Bycatch mitigation ultimately seeks to change human behaviour, by altering fishers' strategic and tactical decisions to, for example, avoid hotspots, adopt bycatch-reducing technologies (BRTs), or release threatened species (Campbell & Cornwell, 2008; M. A. Hall, 1996). Interventions to change behaviour may be more effective if grounded in an appropriate theory (Davis et al., 2015). While there are many theories of behaviour, the Theory of Planned Behaviour (TPB) (Ajzen, 1991, 2011) is one of the most widely-applied and empirically-tested, including for understanding and encouraging pro-environmental behaviour, but with limited application to marine conservation and fisheries management (Andrews et al., 2021; Davis et al., 2015; St John et al., 2010). TPB states that behaviour is driven by three factors: 1) behavioural beliefs, 2) normative beliefs and 3) control beliefs towards a behaviour (Ajzen, 1991, 2011) (Figure 5.2). Behavioural beliefs result in an individual positive or negative evaluation of the behaviour, based on the outcome expectations of performing the behaviour. As per the theory and its empirical applications to other pro-social behaviours, these expectations relate to extrinsic or intrinsic rewards, such as material profit and social recognition, or individual moral attitudes (Ajzen, 1991; Arvola et al., 2008; Tonglet et al., 2004). Normative beliefs result in subjective norms, of which there are two types; descriptive – perceptions of peers' actual behaviour, and injunctive – perceptions of whether peers approve or disprove of a behaviour. Control beliefs result in perceived behavioural control, and are based on perceptions of the ease or difficulty of performing the behaviour depending on the context (Ajzen, 1991, 2011). All things being equal, the more positive a person's behavioural, normative, and control beliefs, the greater their behavioural intention and, thus, the higher the likelihood that they perform a behaviour (Vallerand et al., 1992).

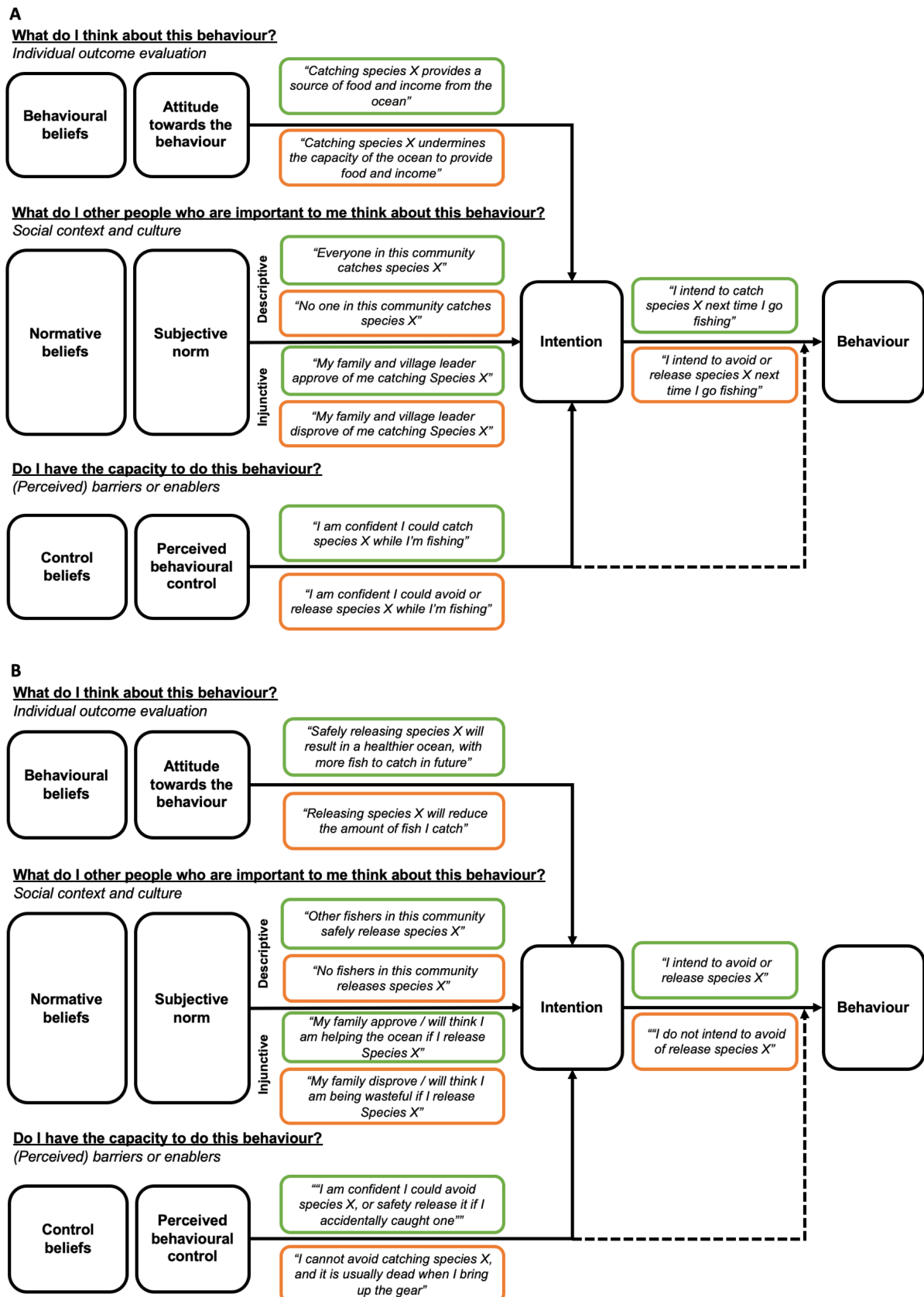


Figure 5.2 The theory of planned behaviour, with example beliefs relevant to bycatch. Panel A relates to catching species X while Panel B relates to avoiding or releasing species X. Green boxes represent positive beliefs towards the behaviour and orange boxes represent negative beliefs towards the behaviour. (adapted from St. John et al. 2010).

TPB is particularly appropriate for fisher behaviour, since it acknowledges that behaviour is multi-faceted, multi-levelled and multi-scaled (Andrews et al., 2021). Importantly, it provides space for different types of outcome expectations that can influence an individual's evaluation of the behaviour (e.g., behavioural beliefs may depend on perceived external rewards, while normative beliefs may depend on social recognition) and different levels of influence (i.e., behavioural beliefs at the individual-level, normative beliefs at the societal-level, control beliefs relating to environment/context). Unpacking these different influences on fisher behaviour is important, since different beliefs can act synergistically or antagonistically, and may align or conflict with conservation outcomes. For example, outcome expectations which are extrinsically rewarding (e.g., food, income) can crowd in or crowd out those which are intrinsically rewarding (Cinner et al., 2021; Grillos et al., 2019). Similarly, different scales of influence can interact, such as conflicts and synergies between individual beliefs and societal norms, or between formal laws and local customs (Bicchieri, 2017; Booth, Mardhiah, et al., 2020; Oyanedel, Gelcich, & Milner-Gulland, 2020). As such, a socio-psychological approach to bycatch could help to disentangle its underlying drivers, diagnose its typology along the bycatch spectrum (Figure 5.1), and support the design of management interventions. For example, if attitudes and norms towards catching threatened species are negative, but perceived behavioural control towards avoiding them is also negative, interventions which improve behavioural control (such as cost-effective BRTs) could be appropriate. Conversely, if attitudes and norms towards catching threatened species are positive, underlying socio-economic motivations may hinder uptake of BRTs.

5.2.2 Case study application

Based on this, I used a mixed methods approach, structured around the TPB, to understand the socio-psychological drivers of catching and potentially releasing endangered elasmobranchs in a coastal gill net fishery in Aceh Jaya, Aceh Province, Indonesia, which frequently interacts with three endangered marine megafauna taxa: hammerhead sharks (*Sphyrna* spp.), wedgefish (*Rhynchobatus* spp.) and whale sharks (*Rhincodon typus*) (Simeon et al., 2020). This is a pertinent case study, since Indonesia is a global priority for aligning SSF management, marine conservation and human well-being (Golden et al., 2016; Selig et al., 2014, 2018); the fishery type is representative of coastal SSFs in the Global South in terms of gears, habitat types and bycatch-affected species; and the socio-economic context in Aceh represents an interesting case for testing a socio-psychological approach, where socio-cultural beliefs and economic motivations interact within the realm of fisheries management and conservation (Quimby, 2015; Wilson & Linkie, 2012).

Within this study location, I aimed to answer the following questions:

1. What are fishers' motivations and salient beliefs regarding a) (by)catching and retaining, b) not catching/avoiding, and c) releasing endangered species?
2. Based on these motivations and beliefs, where does fisher behaviour sit along the spectrum of bycatch?
3. What do these findings tell us about the barriers and opportunities for designing interventions which can effectively and ethically change fisher behaviour for bycatch mitigation?

Study Site

This study focused on Lhok Rigaih in Aceh Jaya regency, Aceh Province, Indonesia. In Aceh, fisheries management is primarily implemented under the *Panglima Laot*, a unique customary fisheries management organisation, which enforces traditional fishing regulations and access, resolves disputes over marine resources, and coordinates sea rescues (Wilson & Linkie, 2012). Under the *Panglima Laot* fisheries and marine management is primarily conducted at the ‘*Lhok*’ level, with *Lhoks* as the smallest unit of customary management, according to customary law (Qanun Aceh Nomor 9 Tahun 2008 Tentang Pembinaan Adat Dan Adat Istiadat, 2008). *Lhok* essentially translates to ‘bay’, and it is a spatial area encompassing a portion of the coast and associated marine habitat. Aceh Jaya regency is on the south coast of Aceh Province, encompassing a 2,095km² coastline and 54 islands fringing the Indian Ocean. This coastline represents important marine habitat, with extensive mangroves, turtle nesting beaches and shallow coastal waters with muddy substrate and coral reefs, which provides nursery ground habitat for a range of species (DKP Aceh, 2018). The regency has a total population of around 93,000 people, and those living in coastal areas typically adopt mixed subsistence livelihoods combining fishing and farming (E. S. Yulianto et al., 2018). As such, the coastal waters of Aceh Jaya also represent an important marine resource, characterised by small-scale multi-gear mixed-species fisheries, which support local food security and livelihoods.

There are 8 officially-registered Lhoks in Aceh Jaya, and Lhok Rigaih is home to the largest harbour in the regency, which serves six villages across two districts, and approximately 200 full- and part-time fishers (E. S. Yulianto et al., 2018). Lhok Rigaih is thought to be the largest fishery landing site in Aceh Jaya (i.e., in terms of number of active vessels), while also being representative of the other Lhoks in Aceh Jaya in terms of fishery characteristics (i.e., gears uses, species caught and habitat). There are three main gear types in use in Lhok Rigaih – gill nets, longlines and handlines. Vessels are typically specialised to use either gill nets (*jaring*) or longlines (*rawai*) as the primary gear, while fishers often also use individual handlines (*pancing*) as secondary gears while the primary gear is soaking. Gill nets are further divided in to ‘*jaring tancap*’ and ‘*jaring lobster*’. *Jaring tancap* is a bottom gillnet that is stuck into the ground, typically in muddy substrate, and used to catch reef and demersal fishes; *jaring lobster* is a smaller and thinner gillnet, used closer to coral reef habitat, for catching lobsters. Government and NGO landings records and indicate that *jaring tancap*, are by far the highest-risk gear in terms of bycatch of endangered species, accounting for roughly 80% of total hammerhead sharks and wedgefish bycatch (Simeon et al., 2020).

Study species

I examined fisher behaviour in Lhok Rigaih regarding bycatch of three different endangered species: hammerhead sharks (*Sphyrna* spp.), wedgefish (*Rhynchobatus* spp.) and whale sharks (*Rhincodon typus*). These three species were chosen because they represent relevant but somewhat contrasting case types. That is, both hammerhead sharks and wedgefish are Critically Endangered species which are frequently landed in Lhok Rigaih, reportedly as incidental catch. The landed hammerheads are typically juveniles, and therefore represent limited economic value,

but are consumed locally (Simeon et al., 2020). In contrast, large wedgefish have high value in the international fin trade (Hau et al., 2018). In contrast again, while whale sharks are anecdotally commonly seen by fishers, and can become entangled in gill nets, they appear to be rarely caught and landed in Lhok Rigaih. In part this is likely due to relatively rarer interactions with the species and national laws which protect whale sharks in Indonesia, but also may be because whale sharks are considered important species in Acehnese culture, which should not be caught and captured.

Data collection

I conducted 16 in-depth interviews (Appendix 2, S5.1) to gather data on the general socio-ecological context of the fisher; and to understand fishers' beliefs, intentions and behaviours for three bycatch-relevant behaviours (catching, avoiding and releasing) and for each study taxa (hammerhead sharks, wedgefish and whale sharks), based on the TPB.

Since this was a descriptive, exploratory study, I conducted opportunistic, snowball sampling, with no *a priori* assumptions regarding sample sizes, and continued collecting data until saturation (D. Cohen & Crabtree, 2006; Newing et al., 2010). During the study, ten bycatch-intensive gill net vessels were operating, with each vessel taking 2-3 crew, and I focused on interviewing vessel captains (10 out of 16 interviewees), so this sample captured most of the relevant population. Interviews were complemented with a focus group discussion (FGD) with eight local marine managers from the Aceh Jaya regency *Panglima Laot* and regency fisheries agency, who provided independent corroboration for the findings from the interviews, and informal discussions and observation at Rigaih harbour and a local coffee shop.

Analysis

I conducted simple descriptive and thematic analysis of the results using Rstudio base code and GGPlot2 (RStudio Team, 2020; Wickham, 2016), and word clouds were prepared using World Cloud Generator.

5.3 Results

5.3.1 Socio-ecological context

Demographics

The fishers I interviewed ranged from 30 to 59 years old, with collectively over 400 years of fishing experience (average 25 years per person). All but one fisher was married, with an average of 3.3 children per person. The majority (81%, N = 13) were born in the Lhok Rigaih area, though three people had immigrated from elsewhere in Aceh. The majority (63%, N = 10) had only completed primary education (up to age 11), while four had completed middle school (up to age 14) and two had completed secondary school (up to age 18). Six of the interviewees were owner-captains, while the rest were crew or using a boat from another private owner or *bantuan pemerintah* (government support). Every fisher reported that they fished primarily to sell the catch and get money to cover their '*kehidupan sehari-hari*' (daily needs or livelihood), all fishers also reported taking a small amount of

catch home for subsistence purposes. Fishers reported a range of target species, primarily reef and demersal fish, as well as some small pelagic species (trevally and mackerel tuna). Most fishers initially responded to the question about what species they targeted with '*ikan*' (fish) or '*apa pun*'/'*ikan apa saja*' (any fish), reflecting the mixed-species nature of the fishery. Almost half of the fishers reported obtaining some additional seasonal income from farming activities, reflecting the fluid and somewhat inter-changeable safety net functions of fisheries and farming for subsistence/cash-based livelihoods.

Well-being

In terms of economic welfare, fishers reported household incomes of IDR 1-3 million per month, averaging at around 2.3 million (US\$ 160). In terms of subjective well-being, 50% (N = 8) of fishers reported being satisfied or very satisfied with their lives. Most of the remainder were 'neutral', while one individual reported dissatisfaction. Many reported being grateful for having enough to get by, and for *rezeki*, which is an Arabic word rooted in Islam, meaning a gift from God/sustenance from God. However almost every fisher reported wanting their own boat, a bigger boat or better fishing gear to make them more satisfied. All interviewed fishers unanimously reported that they enjoy fishing, with nine people explicitly stating intrinsic motivations such as 'it's a hobby' and 'I enjoy the challenge when the fish pull'. In contrast, they also unanimously stated that they don't want their children to become fishers. Rather, they want their children to continue their education and become '*orang yang baik*' (good people) or '*orang sukses*' (successful people).

Ocean stewardship

The concept of *rezeki* was also commonly reported when fishers were asked about their positive associations with the ocean, e.g.: "*ada rezeki di laut*" (there are gifts from God in the ocean) (Figure 5.3); while big waves and bad weather were reported as negative associations with the ocean. In general, fishers report a strong sense of ocean stewardship, with most people agreeing or strongly agreeing with concepts of ocean protection "to keep the fish in order", "so the fish can grow" or "to protect the current livelihoods, and for *akan cucu* (future generations)" (Figure 5.3). Others also reported that "it's not good to catch the babies" and that they needed to be "*pelapor*" (whistle-blowers) for use of destructive fishing gears such as bombs and cyanide. There was also a general appreciation of the connectedness of the ocean, and the need to protect some aspects of marine ecosystems so that others can be harvested. For example, one fisher stated "*laut adalah ranti, hilang satu, hilang semu*" (the ocean is a chain, lose one [link], lose all). All fishers were aware of both government and *adat* (customary) regulations regarding fisheries management and protected species, these include laws protecting manta rays, sawfish and whale sharks, as well as some spatial regulations and regulations regarding not fishing on Fridays (primarily for religious purposes). All fishers reported positive perceptions of these regulations, to 'keep the ocean safe', 'protect the babies' or because some species 'cannot be eaten'. The role of the *Panglima Laut* in terms of organising fishers, coordinating customary law and resolving conflicts was also frequently stated.

Social relations

Almost all fishers agreed or strongly agreed that they get along well with other fishers in their community (Figure 5.3), and that they talk with other fishers about their fishing practices and help each other e.g., “*di laut semua saudara*” (at sea we are all brothers). Most fishers reported that they would be likely to adopt fishing practices from others, provided they were easy or more profitable, though some stated that they preferred to go their own way (Figure 5.3).

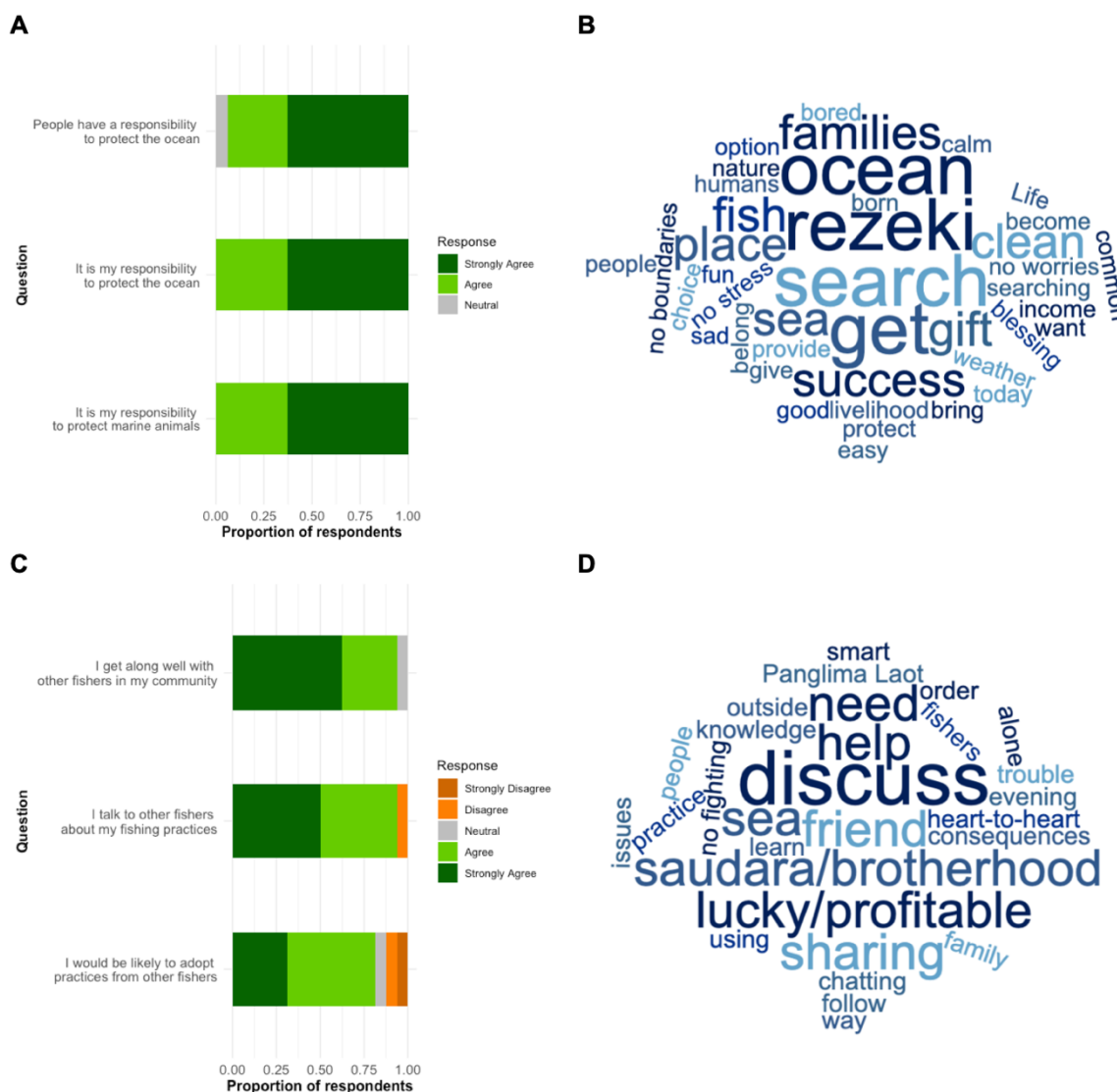


Figure 5.3 Fishers' perceptions of ocean stewardship (A, B) and social relations with other fishers (C, D), where A and C represent quantitative responses to Likert-scale questions, while B and D summarise qualitative answers (Rezeki means 'gift', and in this context 'gift from God' or 'gift from the ocean').

5.3.2 Catching and releasing endangered elasmobranchs: fishers' beliefs, intentions and behaviour

Beliefs

Fishers reported strong positive behavioural and normative beliefs regarding catching wedgefish and hammerheads (Figure 5.4), with neutral to negative beliefs regarding avoiding and releasing them (Figure 5.5). In

contrast, they held strong negative behavioural and normative beliefs regarding catching whale sharks (Figure 5.4), and positive beliefs regarding avoiding and releasing them (Figure 5.5).

Positive behavioural beliefs were strongest for catching wedgefish (Figure 5.4), with all fishers stating the main advantage as financial gain from sales. Behavioural beliefs for catching hammerheads were also consistently positive, with food more commonly mentioned as an advantage; they are consumed in shark curry and thought to be *enak* (delicious). The term “*rezeki*” (gift from God) was commonly used to describe these catches (Figure 5.4). These attitudes were reinforced by positive normative beliefs, with almost all fishers agreeing or strongly agreeing that most people catch wedgefish and hammerheads, and approve of their capture (Figure 5.4). Control beliefs for wedgefish and hammerheads were positive on average, but less consistent, with fishers reporting stochasticity as a barrier, e.g., it depends on “luck”, “the ocean” and/or “God’s will” (Figure 5.4). Behavioural beliefs regarding not catching wedgefish and hammerheads were neutral-to-negative, with many stating “*tidak apa-apa*” (no problem) or “*Insh’Allah*” (if God wills), while control beliefs for avoidance were more strongly negative, e.g., with fishers stating “they just come to the net” (for wedgefish) or “they are everywhere” (for hammerheads). Behavioural beliefs regarding releasing wedgefish and hammerheads were generally negative, with disadvantages including less income (e.g., “it brings more money even though it’s not the target, if it’s dead I want to bring it home”) and ‘*mubazir*’. *Mubazir* literally translates as wasteful, but also has religious connotations, and implies that “God will be displeased” i.e., catching a fish is *rezeki*, and throwing it back in to the ocean is disrespectful to God (*mubazir*). E.g., one fisher stated “it is *rezeki* (a gift from God), it cannot be released” (Figure 5.5). However, some fishers reported advantages, e.g., “if it’s small and alive ... it can make more fish”. Control beliefs regarding releasing wedgefish and hammerheads were heterogenous (Figure 5.5). For wedgefish, control beliefs were neutral-to-slightly positive, with several fishers reporting that they are sometimes alive and typically “stronger than hammerheads”. For hammerheads, four fishers reported strong positive control beliefs and four reported strong negative control beliefs. Those that reported negative control beliefs stated that hammerheads are “usually” or “always” dead when the gear is brought up, while those that reported positive control beliefs conditioned their answer with “if it’s small and alive” (Figure 5.5).

In contrast, all but one fisher (N = 15) consistently reported strong negative beliefs regarding catching whale sharks and strong positive beliefs regarding releasing them, which were backed up by statements regarding customary beliefs and norms (Figure 5.4). E.g., most fishers stated “*dia bawa rezeki*” (she/he brings the gifts from God) or “*dia bawa ikan kecil*” (she/he brings the small fish), and reported that whale sharks cannot be consumed or sold under government and customary regulations. All but one fisher also reported positive control beliefs regarding releasing whale sharks, stating that they are “strong”, “calm” and “not dangerous” so that it’s easy to release them (Figure 5.5). The only reported disadvantage was they sometimes have to cut their net to release whale sharks, though some fishers also said that cutting the net wasn’t necessary.

Intentions

For both wedgefish and hammerheads, there was a disconnect between beliefs and intentions. When fishers were asked if they intend to catch these taxa, answers were inconsistent and neutral on average; some fishers stated they wanted to catch them, others stated that it wasn't their target and the other fish are more valuable (Figure 5.4).

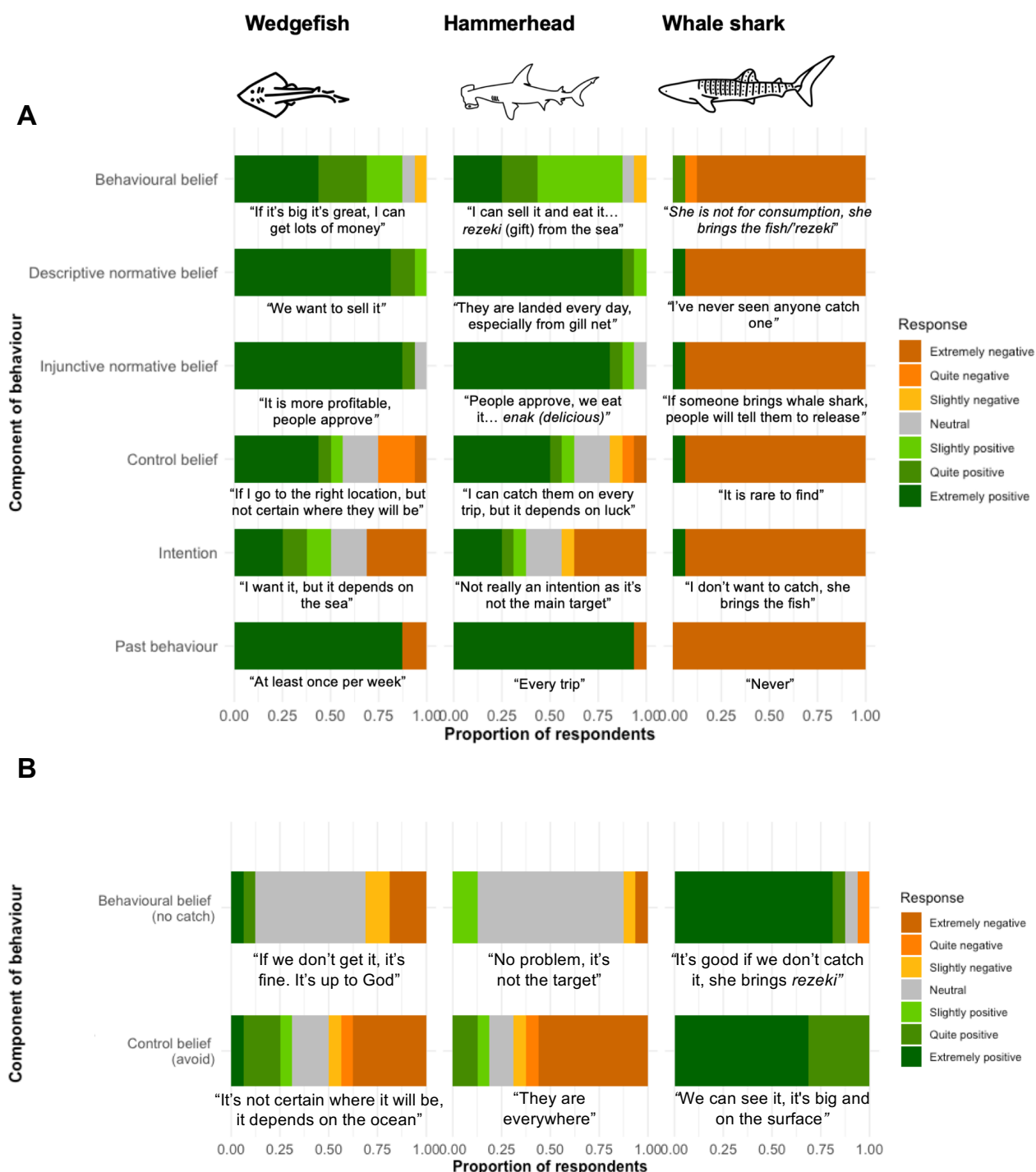


Figure 5.4 Summary of the positivity/negativity of fishers' beliefs regarding (A) catching and (B) avoiding wedgefish, hammerheads and whale sharks, where negative pertains to all responses that were 'bad', 'disagree' or 'false', while positive pertains to all responses that were 'good', 'agree' or 'true' (see Appendix 2 for details on question framing). The bars represent quantitative results from Likert-scale questions, while the quotes below each bar represent an illustrative qualitative explanation from one or more fisher.

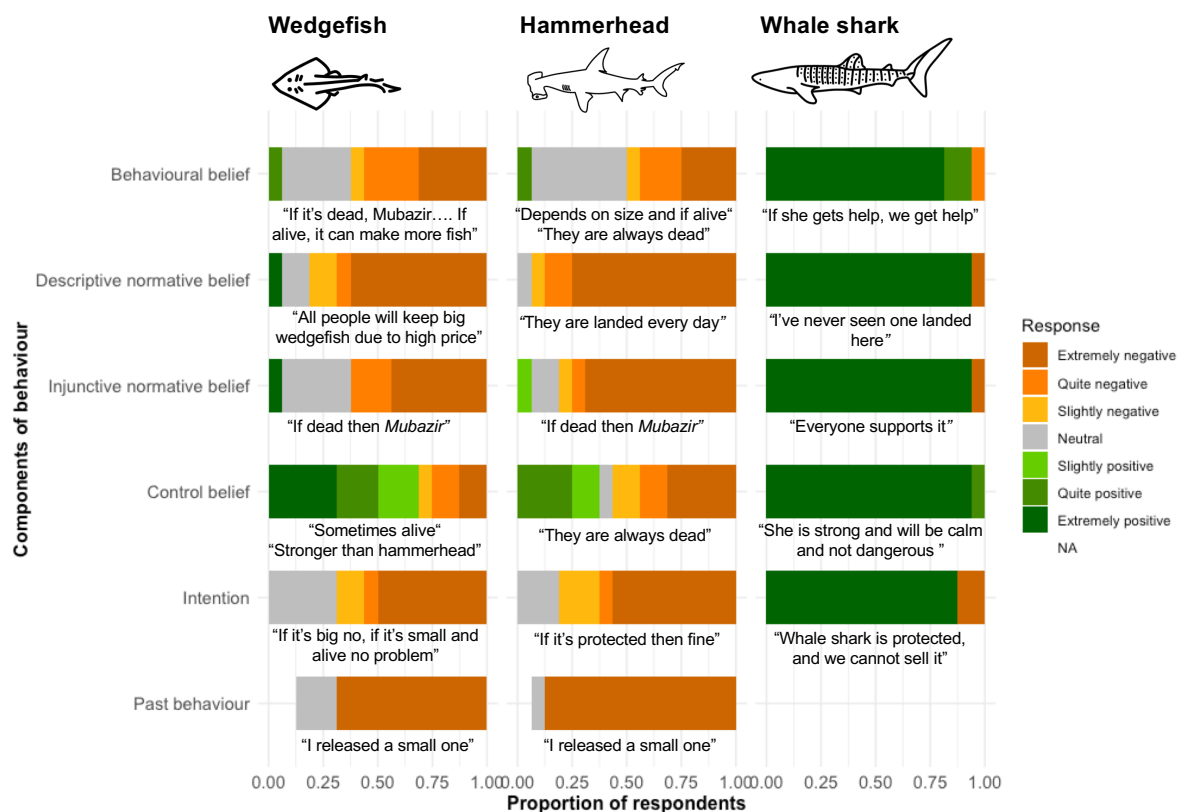


Figure 5.5 Summary of the positivity/negativity fishers' beliefs regarding releasing wedgefish, hammerheads and whale sharks, where negative pertains to all responses that were 'bad', 'disagree' or 'false', while positive pertains to all responses that were 'good', 'agree' or 'true' (see Appendix 2 for details on question framing). The bars represent quantitative results from Likert-scale questions, while the quotes below each bar represent an illustrative qualitative explanation from one or more fisher.

Intentions for releasing hammerheads and wedgefish were consistent with beliefs and predominantly negative (Figure 5.5). For fishers who were neutral ($N = 4$), they explained that if it was small and alive they would release it, but if it was big and/or already dead then they would not. One fisher stated that if they were protected by the *Panglima Laot* (a local traditional fisheries management institution in Aceh) he would be willing to release them. For whale sharks, reported intentions to catch, avoid and release them were consistent with beliefs; strongly negative for catching them and strongly positive for avoiding and releasing (Figure 5.4, Figure 5.5).

Behaviour

Regarding past behaviour, most fishers reported having caught wedgefish or hammerheads in the past three months, though wedgefish catches were less frequent and numerous, while none reported catching a whale shark (Figure 5.4). Most fishers stated that they have not released a wedgefish or hammerhead shark in the past three months, however three fishers reported releasing a small wedgefish and one fisher reported releasing a small hammerhead. Since no fishers reported catching a whale shark, none could answer the question on release (Figure 5.5).

5.3.3 Managers' perspectives

During the FGD, local marine managers also corroborated that food and profit are the main drivers of bycatch and retention of hammerhead sharks and wedgefish, noting that profit is relatively more important for wedgefish, while food and 'culinary culture' is relatively more important for hammerhead sharks. Potential solutions were also discussed, including compensation schemes for cutting nets, and compensation schemes for release of individuals. The *Panglima Laot* of Aceh Jaya regency also noted recent efforts to encourage fishers to release juveniles (though this had not been formally incorporated in to any government or customary rules at the time of the study), which may explain why some fishers reported positive attitudes towards releasing small wedgefish and hammerheads, and positive past behaviour.

5.4 Discussion

I have presented a socio-psychological approach for diagnosing drivers of bycatch in SSFs, and applied this approach to gain a better understanding of fishers' beliefs, intentions and behaviours regarding three endangered elasmobranch species in a case study SSF in Indonesia. The findings have implications for marine conservation and fisheries management in Aceh Jaya, and offer broader lessons for bycatch mitigation in small-scale mixed-species fisheries, and for disentangling intrinsic and extrinsic motivations of fishers, which may align or conflict with conservation objectives.

5.4.1 Diagnosing bycatch: interpreting beliefs and behaviour within the spectrum of bycatch

The results of the case study suggest that (by)catch of wedgefish and hammerhead sharks in Lhok Rigaih can be diagnosed as valuable secondary catch, while catches of whale sharks are undesirable incidental bycatch (Figure 5.6). This interpretation is consistent with what might be expected given the current regulatory and economic context of elasmobranch trade: it remains legal to catch and domestically trade hammerhead sharks and wedgefish in most of the world's fisheries, and wedgefish are one of the highest value species in international markets (Hau et al., 2018), while hammerheads are used for subsistence purposes in Aceh (Simeon et al., 2020). In contrast, whale sharks are legally protected in Indonesia, and also protected via customary law in Aceh. Fisher's control beliefs are also largely consistent with other case studies on catchability and survivability of hammerhead sharks and wedgefish. For example, fishers reported relatively low control beliefs for avoiding and releasing hammerheads, which is consistent with other case studies that suggest hammerheads have high vulnerability to capture in gill nets and low post-release survivability (Ellis et al., 2017; Gallagher et al., 2014; Harry et al., 2011). In contrast, wedgefish are relatively less abundant, and therefore catch and avoidance may be harder to predict; however they are reportedly relatively more robust to capture, which is consistent with past studies on survivability of other shovelnose rays (*Rhinidae* spp.) in other fisheries (Fennessy, 1994; Stobutzki et al., 2002).

This diagnosis sheds light on the types of interventions and policy-mixes that might help to reduce threats to hammerhead sharks and wedgefish. For example, the consistency of fisher's control beliefs with past ecological studies builds the evidence base for live-release as a feasible bycatch-reduction option for wedgefish and other shovelnose rays, yet indicates that avoidance measures (such as spatio-temporal closures) are more important for hammerhead sharks. However, in both cases, incentives or compensation may be required to align interests of fishers with bycatch mitigation. For example, in the case of wedgefish, this could be performance-based payments for live release (Wosnick et al., 2020), in the case of hammerhead sharks, this could involve marine conservation agreements which buy-out fishing rights in areas of important habitat (Sykes et al., 2018). More broadly, the consistency between fisher perceptions and other studies supports the validity of using fishers' ecological knowledge to inform management of the Lhok Rigaih fishery and other data-poor small-scale fisheries (Silvano & Valbo-Jørgensen, 2008), which can also build engagement, legitimacy and compliance with rules (Oyanedel, Gelcich, & Milner-Gulland, 2020).

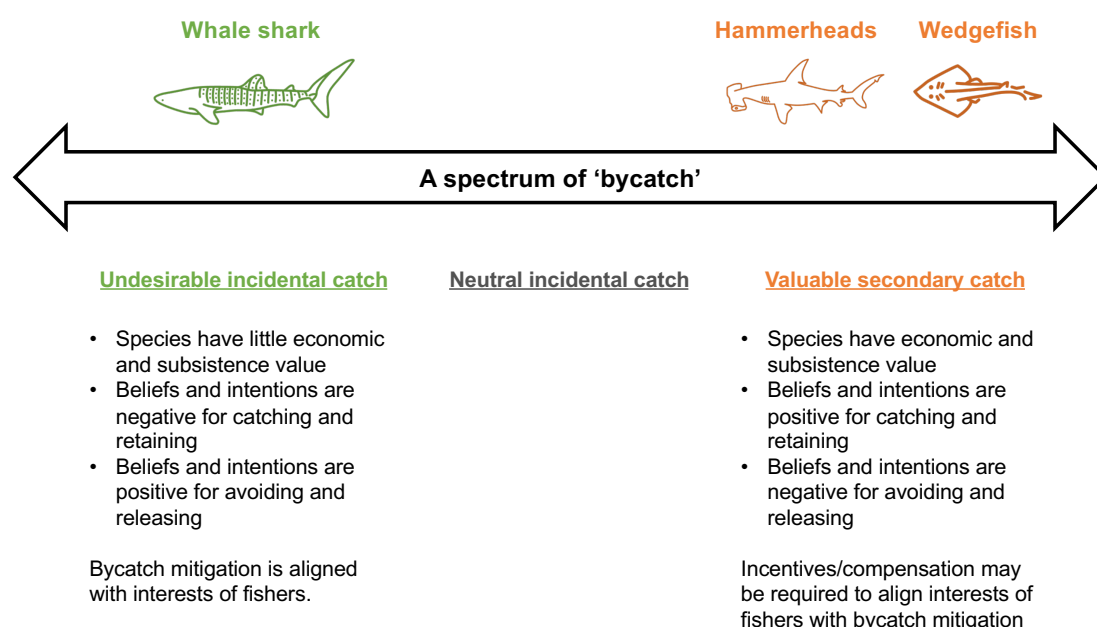


Figure 5.6 Results for each study species mapped onto the bycatch spectrum

5.4.2 Attitudes and norms: interactions between customs, income and food security

This study also indicates how culture and customary institutions, in this case in form of Islam and the *Panglima Laot*, can both help and hinder conservation of marine species. In general, the *Panglima Laot* plays an important role in establishing rules and norms for fishers in Aceh, as well as supporting social capital and resolving conflicts (Quimby, 2015; Wilson & Linkie, 2012). It therefore acts as a highly influential institution regarding shaping fisher behaviour through rules from a legitimate authority, and the transmission of information and norms. Similarly, religious beliefs, particularly concepts of *rezeki* (gifts from God) and *mubazir* (wasteful, God will be displeased) shape fishers' relationships with marine animals. For example, the concept that whale sharks bring *rezeki*, alongside a customary rule of protecting whale sharks (which was more recently reinforced with government

regulations) mean that fishers almost unanimously support avoidance and release of whale sharks. In contrast, the idea that other species which are caught are *rezeki*, and that to throw them back in to the ocean, particularly if they are already dead, would be *mubazir*, means that fishers are disinclined to avoid or release other species. Relatedly, the inherent biophysical stochasticity of fish catches – a fundamental component of the control factors of behaviour – are generally interpreted in a religious context, such that if fishers do catch wedgefish or hammerheads it is positively interpreted as *rezeki*, whereas if they don't catch them it is more neutrally interpreted as God's will. This may also explain the mismatch between beliefs and reported intentions regarding wedgefish and hammerhead shark catches.

These intrinsic and normative motivations also interact with desires for material well-being, including income and food, which are common drivers of elasmobranch catches in small-scale fisheries (Glaus et al., 2018; Jaiteh, Loneragan, et al., 2017). In the case of wedgefish and hammerhead sharks, intrinsic and extrinsic motivators act synergistically to drive exploitation, since fishers also want to gain the monetary and subsistence values from these catches, which are driven by both local and international markets. In the case of whale sharks, they are a protected species in Indonesia, and reportedly “not for consumption” locally, therefore social norms and economic drivers act synergistically to drive behaviours in favour of conservation.

5.4.3 Future directions

The approach and results I have presented also have implications for bycatch management and marine conservation more broadly, which can help to advance progress towards effective and equitable conservation beyond the case study fishery.

By identifying the salient beliefs which drive overexploitation or conservation, a socio-psychological approach can build understanding between ocean stakeholders, helping to identify common interests where they occur, and create opportunities for honest dialogue about conflicts of interest to work towards negotiated solutions (Newing & Perram, 2019; Redpath et al., 2013). In the case of Aceh Jaya, income is an extrinsic motivator for catching and retaining wedgefish. This means that technical and regulatory fixes alone may be insufficient for changing fisher behaviour. Rather, collaboratively designing an incentive-based scheme, whereby fishers are paid to protect and release wedgefish (Bladon et al., 2016; Sykes et al., 2018; Wosnick et al., 2020) or compensated if their nets are cut or damaged (Matwal et al., 2014) could improve uptake of bycatch-reduction practices as well as being socially just. In addition, there are opportunities to build on existing intrinsic motivations and institutions for ocean stewardship to develop legitimacy-based and bio-cultural approaches to conservation. In Aceh Jaya, this could involve working with respected customary and religious institutions, such as the *Panglima Laot* and/or the Eco-Islam movement (Abdelzaher et al., 2019; Oyanedel, Gelcich, & Milner-Gulland, 2020). Such efforts could be further supported by voluntary and information-based approaches, including social-marketing (Segerson, 2013; Veríssimo, 2019), which build on existing social relations, sense of ocean stewardship, and legitimacy of customary rules. This may be particularly effective in Aceh Jaya, where stewardship, institutions and social networks appear

to be strong (Figure 5.3), and in other locations where these conditions are also met (De Lange et al., 2019; Green et al., 2019). Overall, combining incentive-based mechanisms with cultural and norms-based approaches could help to change intrinsic and extrinsic motivations regarding catching and releasing wedgefish and hammerheads in Aceh Jaya, while also respecting the needs and rights of local fishers, and ensuring they are ‘no worse off’ in terms of income and food security.

More broadly, developing interventions which can change human behaviour at scale are critical for ‘bending the curve’ on marine biodiversity loss, while also meeting the socio-economic needs of society (N. J. Bennett et al., 2019; Díaz et al., 2019; Giron-Nava et al., 2021; Travers et al., 2021). Achieving this requires that marine conservation scientists move beyond understanding and addressing the technical and operational mechanisms that cause bycatch, towards identifying the underlying drivers of conservation issues (Booth, Squires, & Milner-Gulland, 2019b; D. R. Williams et al., 2020). Socio-psychological approaches to understanding bycatch can support this, by diagnosing local and individual drivers of behaviour, and their links to broader structural issues such as markets, policies and institutions, which in turn can inform the design of behavioural and structural interventions, for driving pro-environmental transformative change at scale (Naito et al., 2021). This will not only make bycatch management more effective – through promoting uptake of management measures, but also more equitable – through seeking to align conservation outcomes with the needs of fishers, or working towards negotiated agreements, which respect the rights and values of coastal communities. This could respond to calls for advancing equity in marine conservation (N. J. Bennett et al., 2019, 2021), and I encourage researchers, managers and decision-makers to think more holistically about bycatch, and adopt social and behavioural methods to understand and manage bycatch in the future.

6. Exploring cost-effective management measures for reducing risks to threatened sharks in a problematic longline fishery



Photo: a woman helps to offload shark catches in Tanjung Luar harbour.

“The world would be a much simpler place if one could bring about social change merely by making a logically consistent moral argument” – Peter Singer.

6.1 Introduction

In an attempt to address the ongoing decline of shark populations (Dulvy et al., 2021; IUCN, 2021; Pacoureau et al., 2021), international policy mechanisms for sharks have been strengthened in the past decade. For example, under various Regional Fisheries Management Organisations (RFMOs), the Convention on Migratory Species (CMS), and the Convention on International Trade of Endangered Species (CITES), there are provisions relating to catch, retention and/or international trade of over 40 of the most threatened and commercially-important shark species (UNEP-WCMC, 2020). However, conservation outcomes in terms of population recoveries remain elusive for most species and places, hampered by policy complexity, socio-economic trade-offs and inadequate fisheries management (Booth, Squires, & Milner-Gulland, 2019b; Dulvy et al., 2017; Lack & Sant, 2011). In particular, science-based fishery management plans, which can translate high-level policies into sustainability outcomes, and the effective implementation thereof, remain limited for most of the world's shark population (Simpfendorfer & Dulvy, 2017). As a result, population declines and species extirpations continue (Kyne et al., 2019; MacNeil et al., 2020; Pacoureau et al., 2021; Yan et al., 2021).

Indonesia is the world's largest shark fishing nation, and therefore a global priority for tackling unsustainable shark fishing (Dent & Clarke, 2014; Dulvy et al., 2017). While large-scale closures (e.g., 'shark sanctuaries' in several island states) appear to be effective for maintaining healthy shark populations (MacNeil et al., 2020; Ward-Paige, 2017; Ward-Paige et al., 2013), these measures are unfeasible for fishery-dependent nations like Indonesia, where coastal communities are highly dependent on marine resources, including sharks, for income, livelihoods and food security (Jaiteh, Loneragan, et al., 2017; Lestari et al., 2017; Selig et al., 2018), and shark bycatch is ubiquitous in small-scale unselective mixed-species fisheries (e.g. Chapter 5). Rather, there is a need to carefully manage trade-offs between shark conservation objectives and the important role of fisheries for human well-being. Moreover, sharks are a highly diverse species group, which are not uniformly vulnerable to overexploitation; some shark fisheries can be sustainable if they operate within science-based management plans (Simpfendorfer & Dulvy, 2017). As such, nuanced approaches to managing shark catches are needed, which can protect the most at-risk species from overexploitation, whilst allowing for sustainable use of more resilient and faster-growing species (Dulvy et al., 2017; Simpfendorfer & Dulvy, 2017). The Indonesian government have already shown strong commitment to implementing CITES, for example through protecting manta rays and developing a robust Non-Detriment Finding (NDF) study for silky sharks (Booth, Mardhiah, et al., 2020; Booth, Pooley, et al., 2020; LIPI, 2018). However, implementing adequate fisheries management for other threatened and CITES-listing species, whilst minimising negative socio-economic impacts of such regulations on vulnerable coastal communities, remains challenging.

Acknowledging these needs and challenges, I explore the cost-effectiveness of potential management measures for mitigating mortality of threatened and protected species in a mixed-species targeted shark fishery (Tanjung Luar in East Lombok, West Nusa Tenggara). More specifically I aim to answer the following research questions:

1. Which variables related to fishing operations create the greatest risk to priority (i.e., threatened and CITES-listed) shark and ray species?
2. Based on these variables, which technical management measures (e.g., spatial closures, temporal closures, gear restrictions and effort restrictions) could have the largest relative conservation benefits for priority species, in terms of reduced risk of capture?
3. What is the relative socio-economic feasibility of these mitigation measures, based on estimated economic opportunity costs to fishers?
4. Overall, which fisheries management measures might be most cost-effective, in terms of minimising risks to priority species with minimal economic welfare losses to fishers?

To my knowledge, this represents the first attempt to operationalise a least-cost approach to shark conservation. The findings and lessons learned can be used to improve the sustainability of problematic mixed-species longline fisheries with multiple stakeholders and management objectives, and cross-taxa conflicts, including the case study fishery and beyond (Gilman et al., 2019; Smart et al., 2020).

6.2 Methods

6.2.1 Study site and taxa

I focus on Tanjung Luar, a small-scale semi-commercial shark fishery in East Lombok, West Nusa Tenggara Province. There are roughly 52 specialised shark vessels operating from Tanjung Luar, all of which are <10GT capacity, which is classified as small-scale according to the Indonesian Ministry of Marine Affairs and Fisheries. All vessels use longlines to target a range of shark species, though some vessels use bottom longlines and others use surface longlines (I. Yulianto et al., 2018). The vessels typically spend 10-20 days at sea, and return with a mixed-species catch of 20-30 individual sharks, which are landed whole and sold at auction as an entire aggregated catch.

Over 70 species of sharks and rays have been recorded at Tanjung Luar (I. Yulianto et al., 2018). For the purposes of this study I focus on ‘priority taxa’, which I define as species which are 1. subject to regulated use under domestic or international policy frameworks (e.g., Indonesian law and/or listed on one of the CITES Appendices) and/or 2. classified as endangered or Critically Endangered according to the IUCN Red list of threatened species. Based on preliminary exploration of available landings data (WCS-IP, 2019), there are 20 species captured in Tanjung Luar which fit this definition. Where ecologically-meaningful I grouped some species into higher taxonomic groups, and removed any taxa which made up less than 1% of total catch, to ensure sufficient statistical power for further analysis. Based on these groupings, seven taxa were used for the main analysis: thresher sharks (*Alopias pelagicus* and *A. superciliosus*); silky shark (*Carcharhinus falciformis*); mako sharks (*Isurus oxyrinchus* and *I. paucus*); mobula rays (*Mobula* spp.); bottlenose wedgefish (*Rhynchobatus australiae*); hammerhead sharks (*Sphyrna lewini* and *S. mokarran*); and dusky shark (*Carcharhinus obscurus*, Table 6.1).

Table 6.1 Summary of total catches of priority taxa in Tanjung Luar shark fishery, 2014-2018

Latin name	Common name	IUCN Red List	CITES	Bottom longlines		Surface longlines		Grand Total	
				N	%	N	%	N	%
<i>Carcharhinus falciformis</i>	Silky shark	VU	II	775	7%	12363	56%	13138	40%
<i>Sphyrna lewini</i> & <i>S. mokarran</i>	Hammerhead sharks	CR	II	1178	11%	1761	8%	2939	8%
<i>Carcharhinus obscurus</i>	Dusky shark	EN		792	7%	223	1%	1015	3%
<i>Rhynchobatus australiae</i>	Bottlenose wedgefish	CR	II	651	6%	39	0%	690	2%
<i>Isurus oxyrinchus</i> & <i>I. paucus</i>	Mako sharks	EN	II	61	1%	764	3%	685	2%
<i>Alopias pelagicus</i> & <i>A. superciliosus</i>	Thresher sharks	VU/EN	II	59	1%	611	3%	465	1%
<i>Mobula spp.</i>	Mobula rays	VU/EN	II	7	0%	160	1%	167	1%
Other species				7230	67%	6303	28%	13533	41%
Total				10,753		22,224		33,244	
Priority %				34%		72%		59%	
Other %				66%		28%		41%	

6.2.2 Exploring management measures

I considered all management-relevant predictors of catch (i.e., operational variables which could be feasibly regulated), based on available data and known correlates from other studies (I. Yulianto et al., 2018). To explore the technical effectiveness and socio-economic feasibility of management measures I used five years of available landings data (WCS-IP, 2019; I. Yulianto et al., 2018), and applied a Boosted Regression Tree (BRT) method (De'ath, 2007) to explore the effects of different types and ranges of fishing practices on: 1) the likelihood of catching each priority taxon during a fishing trip (as an indicator of hypothetical mortality risk that could be avoided); and 2) trip profit (as indicator of socio-economic feasibility, in terms of which types of operational fishing practices tend to lead to higher/lower profits overall).

Data

I used landings data collected by the Wildlife Conservation Society Indonesia Program during 2014-2018. Data were collected by three experienced enumerators, following SEAFDEC protocols (SEAFDEC, 2016; I. Yulianto et al., 2018). The enumerators were trained, assessed and mentored in species identification, with records verified using photo-ID in cases of uncertainty or ambiguity. Landings were recorded every morning at the Tanjung Luar shark auction facility at 5am – 10am from January 2014 to December 2018. Data were recorded on catch composition and fishing operations, as well as information on the final auction price of total catches from each vessel. These data were collected under a Memorandum of Understanding and Technical Cooperation Agreement between the Wildlife Conservation Society and the Ministry of Environment and Forestry, Ministry Marine Affairs and Fisheries and the Marine and Fisheries Agency of West Nusa Tenggara Province.

Analysis

I used boosted regression trees (BRTs) – a machine-learning method for data exploration and analysis (De’ath, 2007; Elith et al., 2008) – to analyse the relative influence of different operational measures on the risk of capture of each priority taxon and total trip profit.

Since this study specifically focused on exploring management measures, rather than building a predictive model of shark catches based on all possible environmental, operational and socio-economic predictors, BRTs were well suited to the research aims. BRTs are non-parametric, and therefore require no prior distributional assumptions or data transformations; rather, they fit complex, non-linear relationships using algorithms, which learn the relationship between the response and its predictors, and are trained iteratively on random partitions of the data (Breiman 2001). As such, BRTs can handle skewed and multi-modal data, and different types of predictor variables and response types, as well as interaction effects between predictors. They have also been proven to handle non-linear relationships, unbalanced data, missing values and higher-order interactions among many variables (all of which are characteristics of the available fisheries data for Tanjung Luar) more robustly than many other parametric methods (Pennino et al., 2013; Soykan et al., 2014). Accordingly, BRTs have been increasingly used in ecology, conservation biology and fisheries science during the past decade (De’ath, 2007; Elith et al., 2008; Froeschke & Froeschke, 2011), and provided a suitably flexible method for analysing the dataset. Unlike conventional models, BRTs do not produce confidence intervals or p-values. Rather, they use relative influence (RI) to characterize the contribution of each explanatory variable to the model, and mean receiver operating characteristic (ROC) from 0 to 1 as a measure of a model’s explanatory power, where values of more than 0.7 indicate acceptable performance of the model and values of >0.8 indicate excellent performance (De’ath, 2007; Elith et al., 2008).

For BRT models with risk of capture as the response variable, I used a Bernoulli distribution (i.e., a binary 0/1 response variable) to model the influence of the different operational predictor variables on the presence/absence of priority species within each aggregated catch per trip. For the BRT model for profit, I used a Gaussian distribution to model the influence of the different operational predictor variables on total profit per trip. The predictor variables for all models were: month, fishing zone, hook number, trip length, fishing depth, and gear type. For all models I used a learning rate of 0.001, tree complexity 5, and bag fraction 0.5, which were selected to minimize holdout residual deviance (a measure of the error remaining in the tree after construction) (De’ath, 2007; Elith et al., 2008). To interpret the model outputs, fitted models were used to generate RI plots and partial dependence plots (PDPs) with 90% confidence intervals. RI indicates the influence of a given predictor variable on the response variable, with the RIs for each variable in a model scaled so that the sum adds to 100, such that higher numbers indicate stronger influence relative to other variables included in the model. PDPs show the effect of a predictor variable on the response variable after accounting for the average effects of all other variables in the model. Together, these two outputs indicate how and to what degree a given predictor

variable affects a response variable. The mean of the predicted effects of these models were used to assess the conservation benefits and cost of different measures.

6.2.3 Assessing the hypothetical cost-effectiveness of management measures

Based on the outputs of the BRT models I developed prediction datasets for likelihood of capture for all priority taxa and total trip profit under a range of imputed plausible values for each operational variable, and across all possible combinations of those values. The values included: each of the twelve months (Jan-Dec), which could be managed via temporal closures; five fishing depths (10m, 20m, 50m, 100m and 200m), which could be managed via depth restrictions; each of the two gear types (bottom and surface longline), which could be managed via gear restrictions; five representative values of hook numbers (50, 100, 150, 300 and 500), which could be managed via hook limits; two fishing zones (West Nusa Tenggara (NTB) and East Nusa Tenggara (NTT)), which could be managed via spatial closures; and three representative values of fishing efforts (trip lengths of 10, 15 and 20 days) per trip, which could be managed via trip limits. This gave a total of 3,601 combinations of operational variables, each with a predicted catch risk (0-1) for each taxon and a predicted total profit. This prediction dataset then allowed us to assess the conservation benefits and socioeconomic costs of different ranges of manageable variables, as follows: Conservation benefits were assumed to be proportional to the reduction in capture risk for each taxon, if that range of operational variables were restricted. Similarly, socio-economic cost was assumed to be proportional to the reduction in total trip profit, if that range of operational variables were restricted.

I then used the prediction data to inform a semi-quantitative approach for assessing the overall cost-effectiveness of each type of management measure across all priority taxa, whilst also considering differences in conservation need for each taxon. Where the risk of capture for a given taxon and a given set of operational values was predicted to be above average, these values were classified as a potential set of operational restrictions (i.e., management measures) which could have conservation benefits for that taxon. Within this set of management measures, I identified the single set of operational values associated with the highest overall conservation benefit for each taxon, which I gave a score of 3. The next 4 operational values were scored 2, and the rest of the above-average values were scored 1. The scores for each management measure for all taxa were then combined into an assessment of overall conservation benefit, with weighted scores for each taxon based on their threat status according to the IUCN Red List of Threatened Species. For weighting, I followed conventions from previous studies which used IUCN Red List categories for species conservation priority scoring, where each threat category represents a doubling of conservation need (Dickman et al., 2015). In this study, all taxa considered as management priorities were Vulnerable (VU) threat category or above, therefore VU species = 1, Endangered (EN) species = 2 and Critically Endangered (CR) species = 4. Where a taxon included multiple species with different threat categorisations (i.e., for *Alopias* spp. And *Mobula* spp.) I used the highest threat level in order to take a precautionary approach. We recognize this scoring system to be inherently subjective and values-based, and emphasize that other weights could be transparently applied for other studies and sites, to represent the conservation values of different taxa in different contexts.

Based on the total conservation benefit score, management measure was then categorised in to very high, high, moderate or low conservation benefit categories, based on which quartile they fell in to. For socio-economic costs I used two categories due to the relatively low predictive accuracy of the profit model (see Section 3.1). Where profits at a given set of operational values were predicted to be above the average, these were classified as management measures with 'higher cost', while those which were below average were categorised as 'lower cost'. The overall conservation benefit scores were then compared with the socio-economic cost categorisation, to identify management measures which might have higher conservation benefits for lower socio-economic cost.

I emphasise that these scores do not account for potential displacement effects of restrictions, wherein, for example, restricting bottom longlines could have a negative impact on taxa that are more frequently caught with surface longlines, due to bottom longlines fishers switching to that gear type. These scores also assume that each operational variable is fully independent, as opposed to being combined as part of an overall fishing strategy, however that is not necessarily the case. For example, on average, fishers who use surface longlines gears also typically set their gears in shallower waters and use more hooks, while fishers who use bottom longlines gears typically set in deeper waters and use fewer hooks (I. Yulianto et al., 2018). Nonetheless, there are overlaps in the distribution of all the operational variables I use. I acknowledge these nuances and draw on additional sources of information about the fishery in the management recommendations section.

6.2.4 Making a management recommendation

Finally, I draw together the results of the assessment with additional sources of knowledge from published literature, to make overall management recommendations.

6.3 Results

6.3.1 Exploring management measures

For identifying management measures, the BRT analysis produced models with excellent ($ROC > 0.8$) or acceptable ($ROC > 0.7$) performance for most of the priority taxa. However, the model for profit performed quite poorly ($ROC = 0.4$), and the management measures model for hammerheads was just below acceptable levels ($ROC = 0.6$; Figure 6.1). This implies that operational fishing variables alone are not reliable predictors of trip profit, which is unsurprising given that profits typically depend on catch outputs rather than fishing inputs (Booth, Squires, et al., 2021). Nonetheless, the model results remain fit for purpose for the aims of this study (i.e., to explore the relative cost-effectiveness of management levers, as opposed to building a predictive model of catch and profit).

The model outputs indicated that the most influential operational predictors for the relative likelihood of catching priority taxa are: month, fishing depth, gear type and hook number; while depth, gear type and month are the most influential predictors of profit (Figure 6.1; Appendix 3, S6.1, S6.2).

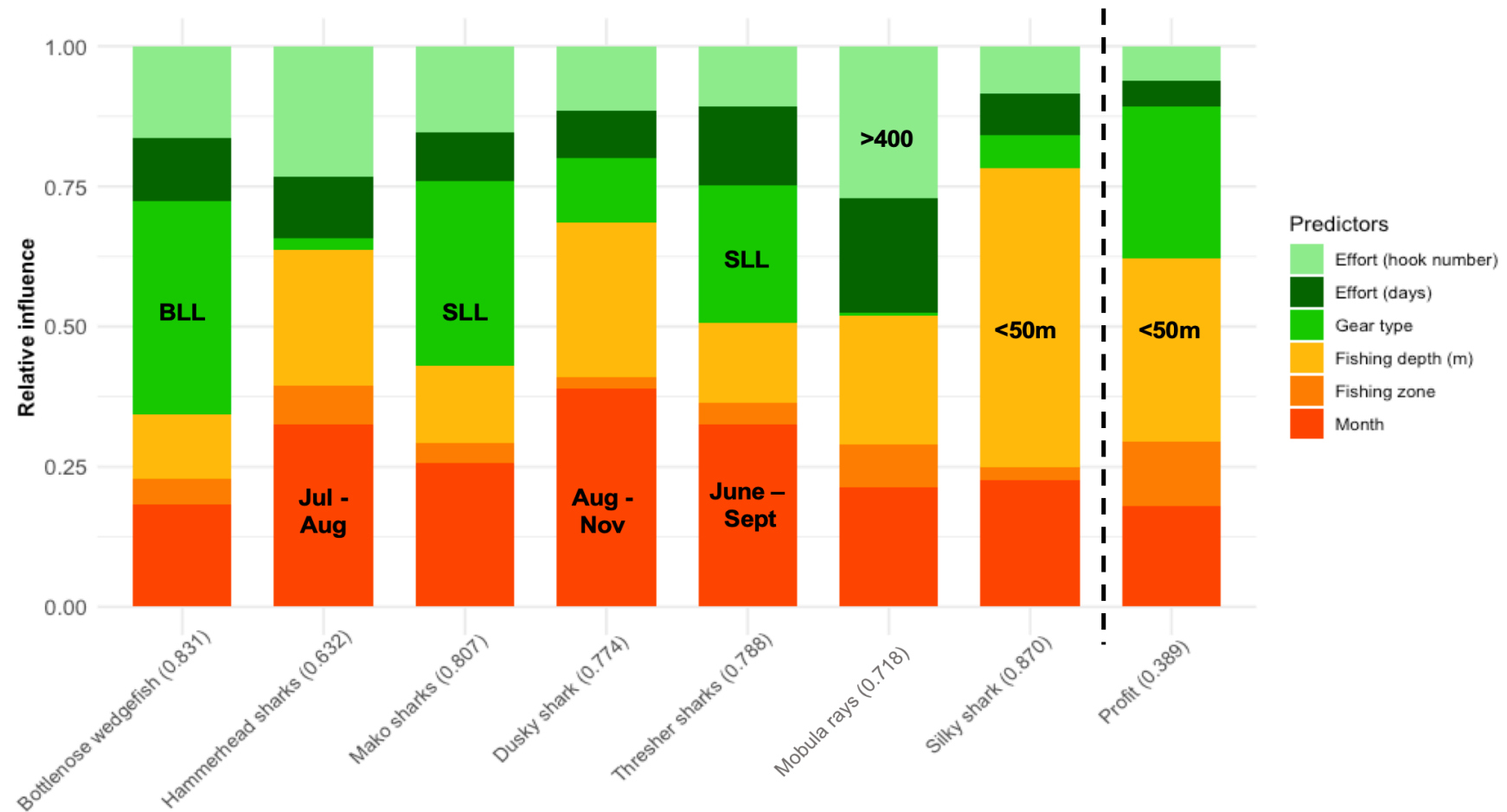


Figure 6.1 Summary of the relative influence of each management-relevant predictor on the likelihood of catching each priority taxa and total trip profit, based on BRT models. The labels in bold text indicate the highest-risk value for the most influential predictor(s) of risk for a given species group (e.g. for wedgefish, the gear is the most influential predictor and the highest-risk gear is bottom longline; see Table S1 for details). The numbers in brackets next to each response variable are the mean ROC values for each model, where values above 0.7 indicate an acceptable performance. BLL = bottom longline, SLL = surface longline.

6.3.2 The month in which temporal closures occur

Month is particularly influential for thresher sharks (33%), dusky sharks (39%) and hammerhead sharks (33%). The likelihood of thresher sharks and hammerhead sharks being caught increases during the middle of the year (June – September and July – August, respectively), while dusky sharks are more likely to be caught during the mid-to-end of the year (Aug to Nov) (Figure 6.2). This suggests that a temporal closure in around July or August could reduce capture of these taxa. This could also be beneficial for mako sharks and mobula rays, though it may have no or negative impacts on wedgefish, since wedgefish are at higher risk in Jan-March (Figure 6.2). Month is also moderately influential for profit (18.1%), with profits peaking in June and July (Figure 6.3).

6.3.3 Fishing depth restrictions

Fishing depth is most influential for silky sharks (53%), and has moderate influence for dusky sharks (28%), mobula rays (23%) and hammerheads (24%; Figure 6.1). The relative likelihood of catch occurrence declines with increasing depth for silky sharks, mobula rays, thresher sharks and mako sharks, and increases with increasing depth for dusky sharks, bottlenose wedgefish, hammerhead sharks (Figure 6.2). Fishing at depths of 0-50m leads to a higher likelihood of catching silky sharks and mobula ray, while fishing at depths greater than 60m and 100m leads to a higher likelihood of catching of dusky sharks and hammerheads respectively (Figure 6.2). Depth is also the most influential variable for profit (33%), with higher profits associated with fishing at <50m depth (Figure 6.3).

6.3.4 Gear restrictions

Gear type has a large influence on capture of wedgefish (38%) and mako sharks (33%), and a moderate influence on thresher sharks (25%). Surface longlines are highest risk for threshers and makos, while bottom longlines are highest risk for wedgefish (Figure 6.2). Gear type is also the second most influential variable for profit (27%) with surface longlines associated with higher profit (Figure 6.3).

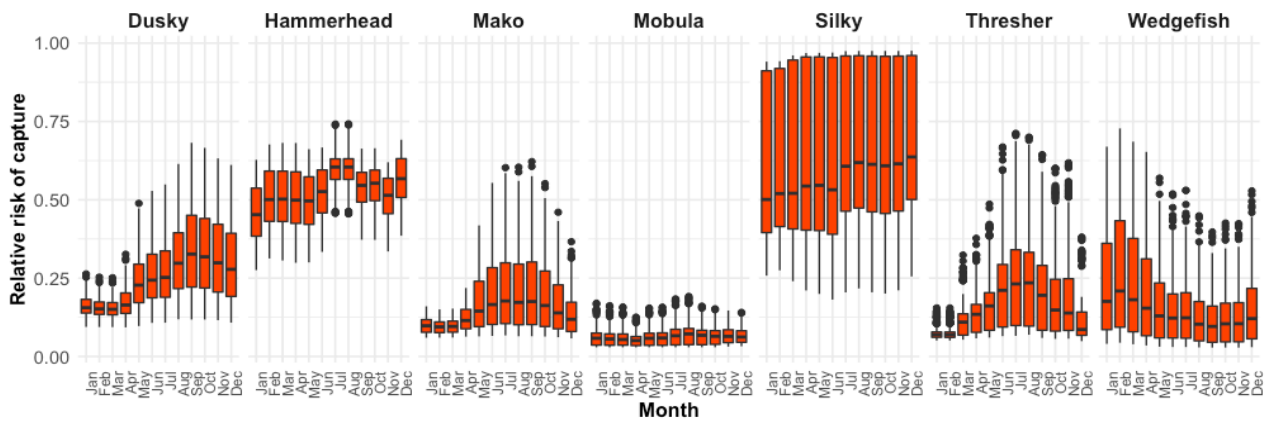
6.3.5 Effort restrictions

Hook number is the most influential variable for capture of mobula rays (27%; Figure 1). Higher hook numbers are associated with higher catches of mobula rays, particularly above 400 hooks (Figure 6.2). Hook number has limited influence on profit (6%) (Figure 6.3).

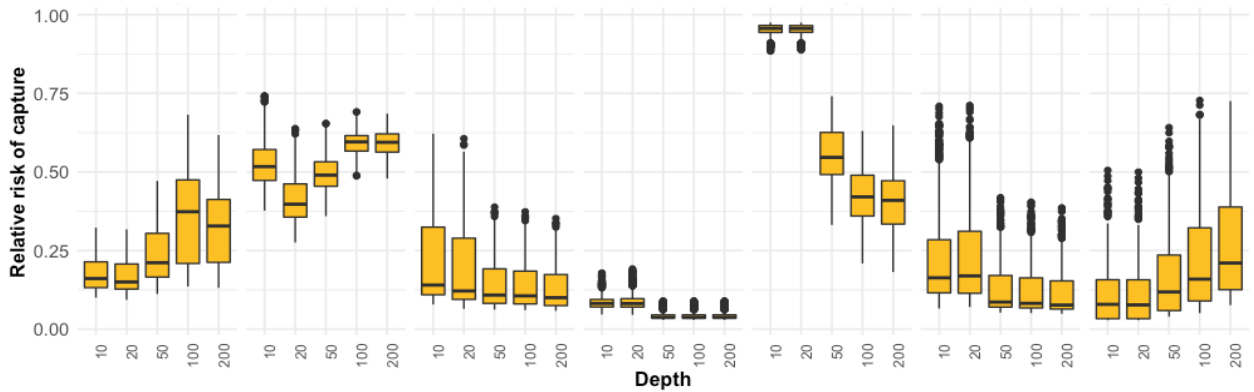
6.3.6 Profit

Depth (33%), gear type (27%) and month (18%) are the most influential predictors of profit (Figure 1). Fishing depths of shallower than 50m and surface longlines are associated with higher profits, while the months of June-July and August-September are also associated with higher profits (Figure 6.3).

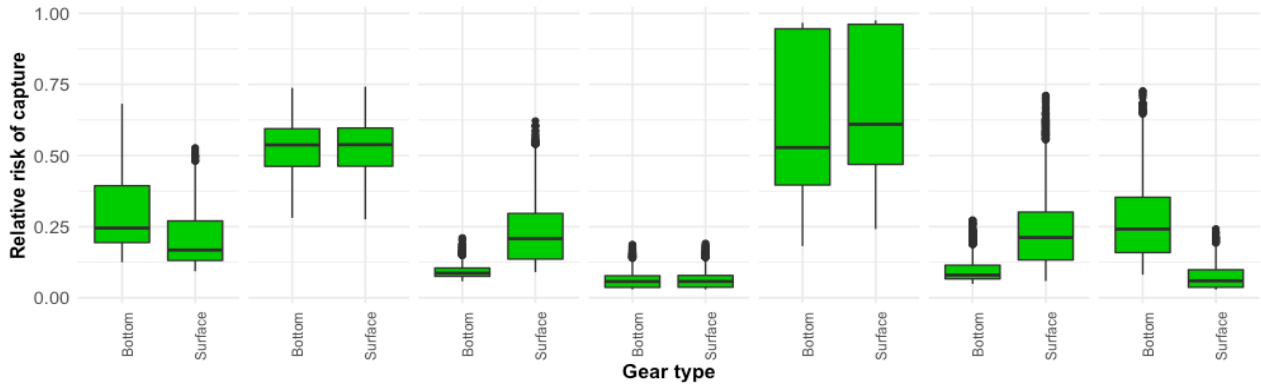
a



b



c



d

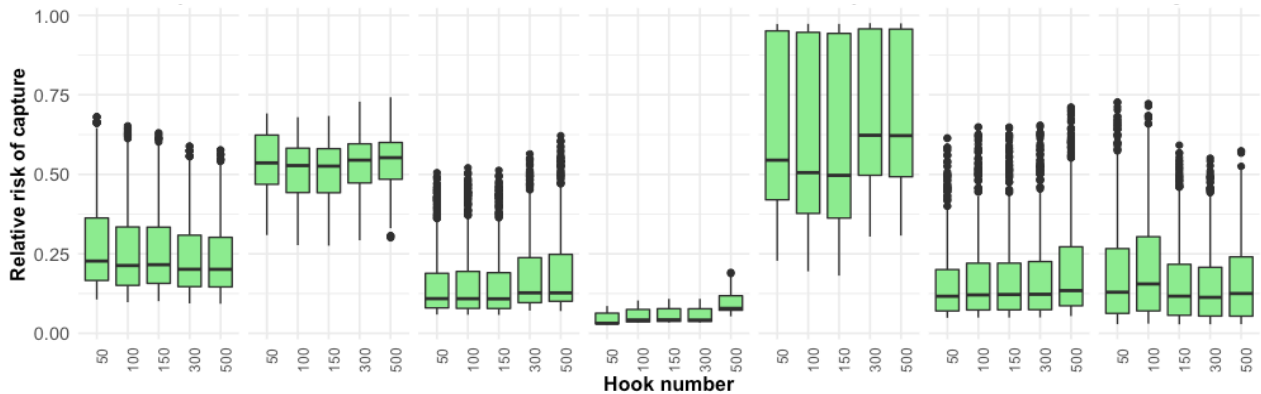


Figure 6.2 The relationship between the relative risk of catching each priority taxon and the most influential operational predictors (a. month, b. depth, c. gear type and d. hook number)

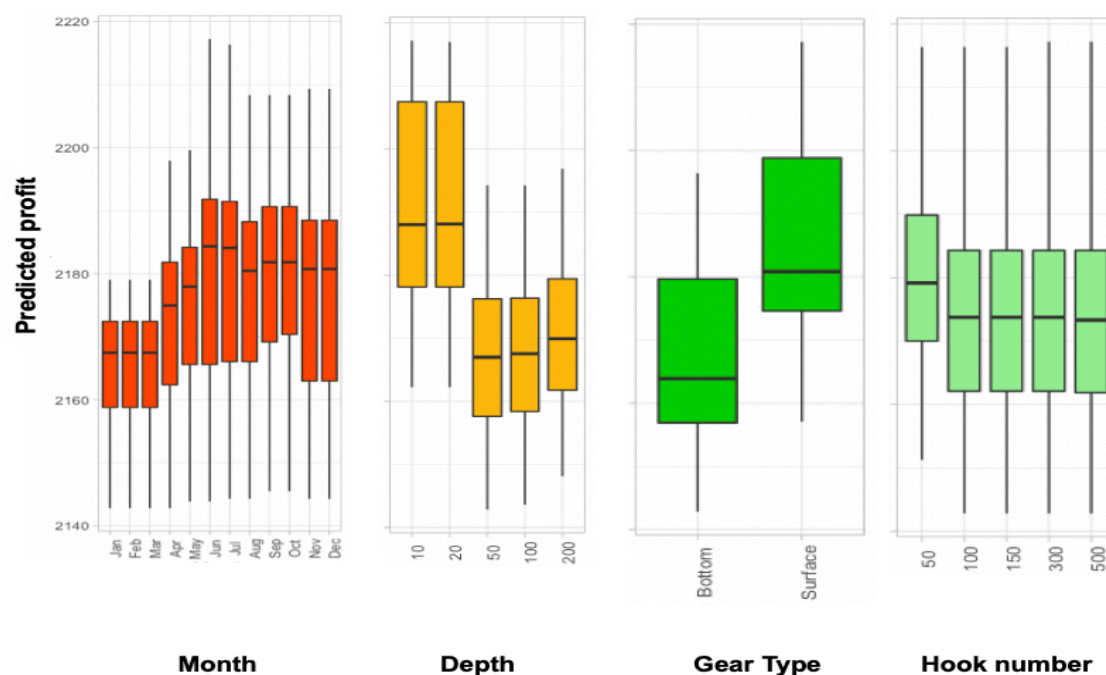


Figure 6.3 The relationships between trip profit and the most influential operational predictors

6.3.7 Hypothetical cost-effectiveness

Based on the assessment, the most cost-effective management measures – in terms of maximising overall conservation benefits while limiting costs – could be seasonal closures in January to March (moderate benefit for lower cost), depth restrictions at >100m (high benefit for lower cost), gear restrictions for bottom longlines (high benefit for lower cost), and hook restrictions at >300 hooks (moderate-to-high benefit for lower cost) (Table 6.3). These measures could be particularly beneficial for hammerhead sharks, dusky sharks, wedgetfish and mobula rays, which are the most threatened species captured in the fishery (Table 6.3). These measures may, however, provide limited benefits for silky and mako sharks, since capture of these species is more frequent in shallower waters using surface longlines (Figure 6.2), which are associated with higher trip profit (Figure 6.3).

Table 6.2 Summary of semi-quantitative cost-effectiveness estimates for each management measure analysed in the BRT models, and their potential benefits for each priority taxa

Mgmt. measure	Benefit to priority taxa							Overall conservation benefit	Relative economic cost
	TH (EN)	SY (VU)	DU (EN)	MK (EN)	MB (EN)	WF (CR)	HH (CR)		
Seasonal closure in...									
Jan						✓✓		Moderate	Lower
Feb						✓✓		Moderate	Lower
Mar						✓✓		Moderate	Lower
Apr						✓		Low	Lower
May	✓			✓				Low	Higher
Jun	✓✓		✓	✓				Moderate	Higher
Jul	✓✓✓	✓	✓	✓✓	✓		✓✓	Very High	Higher
Aug	✓✓	✓	✓	✓✓	✓✓		✓✓✓	Very High	Higher
Sep	✓	✓	✓✓	✓✓	✓✓		✓	High	Higher
Oct	✓	✓	✓✓	✓	✓		✓	High	Higher
Nov	✓	✓✓	✓✓	✓	✓			Moderate	Higher
Dec		✓✓	✓	✓	✓		✓✓	High	Higher
Depth restrictions at...									
10	✓	✓✓		✓✓	✓✓		✓	Very High	Higher
20	✓✓	✓✓✓		✓	✓✓			Moderate	Higher
50								Low	Lower
100			✓✓✓			✓	✓✓	High	Lower
200			✓✓			✓✓	✓✓	High	Lower
Gear restrictions for...									
BLL			✓			✓✓✓		High	Lower
SLL	✓✓	✓		✓✓✓	✓			Moderate	Higher
Hook restrictions at...									
50			✓			✓	✓	Moderate	Higher
100			✓			✓		Low	Lower
150			✓					Low	Lower
300		✓		✓			✓	Moderate	Lower
500	✓	✓✓		✓	✓✓✓		✓	High	Lower

TH = thresher, SY = silky, DU = dusky, MK = mako, DV = mobula ray, WF = wedgefish, HH = hammerhead;

VU = Vulnerable, EN = Endangered, CR = Critically Endangered;

(✓) = predicted conservation benefit above average for that taxon, (✓✓) = top five predicted benefit for that taxon, (✓✓✓) = absolute highest predicted benefit for that taxon

6.3.8 Management recommendations

Combining and interpreting the results of the BRT analysis with additional sources of knowledge on the fishery and taxa (Table 6.3) indicates that depth restrictions and effort limits could provide cost-effective management options for reducing threats to Critically Endangered species, while allowing fishers to continue to benefit from shark fishing. Temporal closures could also have conservation benefits, but may be difficult to implement due to the opportunity costs of closing the fishery for an entire 2-3-month period, during which fishers would have limited other sources of income. This could be particularly challenging in July and August, when the conservation benefits could be highest but the socio-economic costs would also be greatest. Closures during January-March may represent a more socio-economically feasible option, which would be particularly beneficial for Critically

Endangered wedgefish, though would still create costs, albeit lower than at other times of year (Table 6.3). Gear restrictions on bottom longlines could also have high conservation benefits for lower overall cost, but may be inequitable and difficult to implement due to limited inter-changeability between surface and bottom longlines. Fishers are typically strongly ‘bought-in’ to their particular gear, both socially and in terms of skill and capital: they usually learn and inherit from their fathers, and the gears are relatively expensive such that the investment in switching to a different gear would be significant (Lestari et al., 2017). As a result, while the overall cost may be relatively lower, there would be an unequal impact on the fishers using bottom longlines (Table 6.3). Nonetheless, it may be possible to implement these measures if agreements and compensation schemes are carefully negotiated with fishers, such as conservation payments for profits foregone during closures, vessel buyouts for bottom longlines fishers or financial support for gear swaps (Gleason et al., 2013; Sykes et al., 2018; Wasserman et al., 2013).

As demonstrated, for most management measures there are clear trade-offs between different taxa. Of course, a complete closure of the shark fishery would address these cross-taxa conflicts and have the largest conservation benefit, but would come at a significant (and arguably unacceptable) cost to the fisher community. If this policy were seriously considered, it would require a negotiated buy-out or compensation scheme, with full free, prior and informed consent of affected fishers. Based on the sum of total profits from all trips in 2017 and 2018, the total economic surplus from the Tanjung Luar fishery is around US\$437,000 – US\$830,000 per year, therefore a negotiated buy-out scheme may need to cover these opportunity costs (discounted in perpetuity).

Table 6.3 Overall assessment of different possible management measures

Potential management measure	Benefit	Feasibility	Overall assessment, and implementation needs
Complete fishery closure	Very High	Very Low	Would have largest conservation benefit for all species, but highly costly to fishers and is unfeasible within current economic and policy context. Could only be implemented through a negotiated buy-out or compensation scheme.
Temporal closure (July – Aug)	High	Low	Could have conservation benefits for all species, since all are caught year-round, and particularly for hammerhead, dusky and thresher sharks. Would result in two months of lost income for fishers, so would require compensation if implemented.
Temporal closure (Jan – March)	Moderate	Moderate	Could have significant conservation benefits for Critically Endangered wedgefish. Would result in lost income for fishers, but during a less-profitable time of year, which may be more socio-economically acceptable. Would require a compensation scheme if implemented.
Spatial closure	Low	Moderate	Found to have limited influence on risk to priority species, could benefit from further analysis using fine-scale spatially-explicit catch data.
Depth restrictions (no fishing >300m)	High	Moderate	Could have conservation benefits for Critically Endangered taxa (wedgefish, hammerheads). Could result in lost income from reduced wedgefish catches which, while caught less frequently than other taxa, are high value per individual (Hau et al., 2018). Could have inequitable cost for bottom longline fishers who typically set gears deeper.

Gear restrictions (on bottom longlines)	High	Low	Could have conservation benefits for Critically Endangered taxa (wedgetfish), however may have inequitable cost for bottom longline fishers who may not easily switch to surface longline due to differences in skill, equipment and social attachments to fishing practices (Lestari et al., 2017). Would require a buyout or gear swap scheme if implemented.
Effort limit (days)	Mode rate	Mode rate	Found to have limited influence on risk to priority species, though an overall effort cap on the fishery could reduce mortality for all species.
Effort limit (hooks <150)	Mod erate	High	Could have conservation benefits for mobula rays. Could be easy to implement at limited costs to fishers, though may reduce total catch and hence profit per trip.

6.4 Discussion

I have applied a BRT analysis of fishery data to explore cost-effective management options for reducing risks to threatened and CITES-listed shark species in a small-scale longline fishery of economic importance in Indonesia. Based on the results, I have outlined a range of potential management measures for the shark fishery, with transparent assessments of their relative cost-effectiveness, and cross-taxa trade-offs. The results can be used within participatory management planning processes to design fisheries management interventions that support endangered species conservation and CITES-implementation, whilst mitigating the negative socio-economic impacts of conservation on small-scale fishers.

Based on these results I highlight some limitations, knowledge gaps and research priorities; and offer some general recommendations regarding managing small-scale shark fishing in challenging socio-economic contexts.

6.4.1 Limitations and knowledge gaps

There are several limitations in this study, which warrant caution and potential future research. Firstly, the low ROC for the profit BRT model indicates that the model has low performance in terms of explaining and predicting profit per trip. There are several possible reasons for this. 1) Profit data were only collected for 2017 and 2018, meaning there was limited data available to train the model. 2) The management-relevant variables used as predictors for the purposes of this study may not be the most relevant predictors for explaining profit. Since shark catches are sold for their meat and fins, catch outputs (such as total catch and species composition) are likely to be more important for explaining economic value than fishery inputs (Booth, Squires, et al., 2021). Profit may also be related to other socio-economic factors, such as skipper skill and capacity, which were not included in this analysis. As such, the profit predictions should be interpreted with caution, with further work needed to gain more accurate measures of the economic opportunity costs of different management measures. Nonetheless, they provide a directional indication of how changes in fisheries operations might influence profit, which remains useful for the aims of this study. In the future, more long-term data on trip profit could help to prepare a better model for a more robust assessment, or opportunity costs could be more accurately estimated based on estimated reduced catches of different taxa, provided such data are available.

Secondly, impact on trip profit represents a somewhat simplified indicator of the socio-economic feasibility of management measures, which will also depend on the beliefs, attitudes, norms and values of fishers, as well as management costs and the practicalities of implementation (Booth, Squires, & Milner-Gulland, 2019b; Fulton et al., 2011; Gupta et al., 2020). In the future it will be important to supplement this narrow assessment with social surveys and participatory techniques, to understand the perceptions of fishers, and any potential non-monetary values associated with certain fishing practices, such as social capital and personal pride. Overall, while I outline basic management recommendations above, any management decisions would need to be made following open discussions and negotiated agreements with fishers, with full respect for their rights and well-being (Gleason et al., 2013; Newing & Perram, 2019). Relatedly, there is a need to understand which types of policies and instruments could help to better align conservation outcomes with socio-economic benefits for fishers in the short and long term. I suggest several options in Table 6.3, such as compensation/incentive schemes, but note that economic and rights-based instruments have received limited attention in small-scale fisheries and even less so in a shark conservation context (Bladon et al., 2016). In the future, it will be important to use interdisciplinary methods to explore the potential role of such instruments in delivering conservation outcomes for sharks, whilst maintaining the well-being of coastal communities (Giron-Nava et al., 2021).

Third, I have quantitatively explored the potential conservation benefits of the different management measures, while only qualitatively acknowledging any potential negative effects due to displacement. In the future, before implementing a management measure, it will be important to understand the likely adaptive responses of fishers so that any unintended consequences – in terms of socio-economic and conservation impacts – can be foreseen and addressed.

Finally, this study only focused on at-sea catch mitigation measures to avoid or minimise capture of the priority species, yet a fully integrated conservation strategy should also consider the potential role of post-capture remediation and compensation strategies (Booth, Squires, & Milner-Gulland, 2019a; Dutton & Squires, 2008). This could be especially important for Critically Endangered species like wedgefish and hammerhead sharks, which are difficult to avoid entirely, and thus require alternative approaches to address residual unavoidable mortality. For example, retention bans with live release protocols could provide further cumulative conservation benefits for species within family Rhinidae (i.e., wedgefish), which exhibit relatively high post-capture survivability (Fennessy, 1994; Stobutzki et al., 2002; Wosnick et al., 2020). However, since wedgefish are not legally protected in Indonesia, and are also amongst the highest value species in domestic and international markets (Booth, Squires, et al., 2021; Hau et al., 2018), this may require a compensation/incentive scheme to encourage uptake amongst fishers. Such programs have proven to be effective for reducing mortality of guitarfishes in small-scale fisheries in Brazil, and these lessons could easily be applied to similar situations in Indonesia (Wosnick et al., 2020). In contrast, live release protocols may be of limited effectiveness for hammerhead sharks, which typically have low survivability (Gallagher et al., 2014). Instead, compensatory approaches, such as protecting and restoring

important hammerhead shark habitat (e.g., nursery and pupping grounds) may be needed to maintain population health.

6.4.2 The future of cost-effective shark conservation

The results show the importance of considering socio-economic costs/feasibility in conservation decision-making, since incorporating this information can result in somewhat different recommendations in comparison to considering absolute effectiveness alone. For example, while seasonal closures in July and August and depth restrictions for <10m might have the highest overall conservation benefits, they are also associated with some of the highest socio-economic costs (Table 6.2). This could make them ineffective or unfeasible to implement, due to ethical dilemmas regarding causing harm to vulnerable coastal communities, and backlash or non-compliance from communities as a result of these costs (Oyanedel, Gelcich, & Milner-Gulland, 2020; Semedi & Schneider, 2021).

This study also highlights the more general lesson that shark conservation entails hard choices and trade-offs (Gilman et al., 2019; McShane et al., 2011). These manifest as trade-offs between biodiversity conservation and human well-being, and as trade-offs between species and different groups of people. These dilemmas warrant further consideration in the move towards a more effective and ethical paradigm for shark conservation, which can contribute to the Convention on Biological Diversity's vision of "*living in harmony with nature*" by 2050.

7. Estimating economic losses to small-scale fishers from shark conservation: a hedonic price analysis

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Photo: Shark fins drying in the sun (left), a shark fisher's modest household (right).

"First, do no harm" – Hippocrates.

7.1 Introduction

Overexploitation for commercial trade threatens wildlife (Broad et al., 2002; IPBES, 2019), with high market prices driving overexploitation and extinction risk for many species of megafauna (McClenachan et al., 2016). However, wildlife trade regulations and conservation interventions often fail to consider the economic realities of wildlife use, which can lead to ineffective interventions with perverse socio-economic consequences (Booth, Clark, et al., 2021; Challender et al., 2015a; Wright et al., 2016). Economic methods can help to address this, and offer important insights for conservation science. Firstly, understanding wildlife market dynamics can reveal important information on trade-driven extinction risk and drivers of exploitation (Challender et al., 2015a; McClenachan et al., 2016; McNamara et al., 2016). Secondly, this information can be used to design cost-effective management interventions, such as: identifying leverage points within supply chains (t Sas-Rolfes et al., 2019; McNamara et al., 2016; A. Nuno et al., 2018); weighing-up costs and benefits of different interventions, and their distributional impacts (Ban & Klein, 2009; Naidoo et al., 2006; Visconti et al., 2015); and identifying least-cost or market-based solutions, such as incentives or compensation (Booth, Arlidge, et al., 2021; Lubchenco et al., 2016; Travers et al., 2016). Finally, market signals can also serve as indicators of market distortions or responses to regulations, to monitor the impacts of interventions (Booth, Pooley, et al., 2020; Challender et al., 2015a). These insights can help to design conservation interventions that are socio-economically feasible and robust to market forces.

I demonstrate this by applying consumer theory to wildlife market transactions. According to consumer theory, the utility of heterogeneous goods (and thus their price) is derived from the utility of their implicit attributes (Lancaster, 1966). For example, in property markets, the utility of a house is determined by number of bedrooms and proximity to amenities (H. Anderson, 2018). Similarly, the value of wildlife goods are determined by attributes such as size, colour, texture, species, quality and sustainability (Hau et al., 2016; Hinsley et al., 2015; Roheim et al., 2018). Hedonic price analysis (HPA) is based on this theory. It is a revealed preference method of economic valuation, which can be used to estimate the implicit value of different attributes of heterogeneous goods (Lancaster, 1966; Rosen, 1974; Taylor, 2003). Most environmental applications of HPA have been based on property markets (Taylor, 2003). To my knowledge, HPA has not yet been applied to understand markets for traded wildlife products, aside from seafood sales (Hammarlund, 2015; Kristofersson & Rickertsen, 2004; Roheim et al., 2018).

Indonesia's shark markets provide a policy-relevant case study for applying HPA to a conservation science and wildlife trade problem. Sharks, rays and their cartilaginous relatives (Class Chondrichthyes, herein sharks) are threatened by over overexploitation (Dulvy et al., 2017; Dulvy, Fowler, et al., 2014; Pacoureau et al., 2021), and for some species fishing mortality is driven by the high economic value of their fins. For example, dusky shark (*Carcharhinus obscurus*) and wedgefish (Rhinidae spp.) fins sell for US\$400-500 per kilogram in China and Hong Kong (Hau et al., 2018; Wu, 2016). As the world's largest shark fishing nation and a major fin exporter Indonesia is a global priority for shark fisheries and trade management (Dent & Clarke, 2014; Dulvy et al., 2017). Moreover, as a Party to the Convention on the International Trade of Endangered Species (CITES), Indonesia is required to

ensure that international trade is sustainable for the 46 species listed on CITES Appendix II (Booth, Pooley, et al., 2020; UNEP-WCMC, 2020).

However, some Indonesian fishers are highly dependent on sharks for food and income, and regulation of shark fishing and trade can result in direct costs (i.e., out-of-pocket expenses) and opportunity costs (i.e., profits foregone) to these fishers, who are often economically vulnerable (Booth, Squires, & Milner-Gulland, 2019b; Jaiteh, Loneragan, et al., 2017; Lestari et al., 2017). This raises practical challenges for shark conservation in Indonesia, since high socio-economic costs may hamper fishers' willingness to comply with regulations, and increase the cost of monitoring and enforcement (Margavio & Forsyth, 1996); and ethical concerns, since conservation should respect the rights of local communities and 'do no harm' to vulnerable people (Balmford & Whitten, 2003; Newing & Perram, 2019; Poudyal et al., 2018). As such, management measures need to be adapted to the socio-economic realities of shark fishing (Booth, Squires, & Milner-Gulland, 2019b; Dulvy et al., 2017).

Estimating the economic costs of shark conservation for fishers requires understanding the supply-side value of conservation-priority species. However, in Indonesia sharks are typically captured and sold as mixed-species bundles, with a single price at the ex-vessel level that does not differentiate by species and their implicit attributes (Fahmi et al., 2013; Ichsan et al., 2019). This makes it difficult to tease apart the values of different species within the catch. I use HPA to tackle this issue. I first show that taxa is implicit in price, then develop a hedonic price function to derive implicit marginal prices for a range of threatened, CITES-listed and commercially-important shark species. I then use the marginal price estimates and hedonic price function to estimate the short-term economic opportunity costs of plausible management scenarios for selected threatened and CITES-listed species, which would be borne by local fishers. Finally, I explore the management implications of the findings, with a focus on how to maximise shark conservation outcomes while minimising cost to local people.

7.2 Methods

7.2.1 Study site

I used data on shark catches and market prices collected from Tanjung Luar in East Lombok, West Nusa Tenggara Province (Figure 7.1). Tanjung Luar is a landing site and auction facility for a small-scale semi-commercial targeted shark fishery, which primarily operates in the Indian Ocean and Makassar Strait (Fisheries Management Area (FMA) 573 and 713), with occasional trips to the Java Sea (FMA 712) (Figure 7.1) (I. Yulianto et al., 2018). At least 1,000 small-scale vessels operate from Tanjung Luar, with around 50 specialised vessels (Figure 7.1) and 132 fishers using longlines to target a mixture of sharks and rays, predominantly large pelagic species (I. Yulianto et al., 2018). Sharks are landed whole, and the entire animal is sold and used. Fins are traded on to international markets, while non-fin products, particularly meat, are consumed locally and domestically. The shark industry is more profitable than non-shark fisheries, and shark fishers report high dependency on shark fishing, limited occupational diversity and low adaptive capacity for shifting into other fisheries (Lestari et al.,

2017). This suggests there are significant socio-economic barriers to regulating shark fishing, with high potential opportunity costs to fishers.

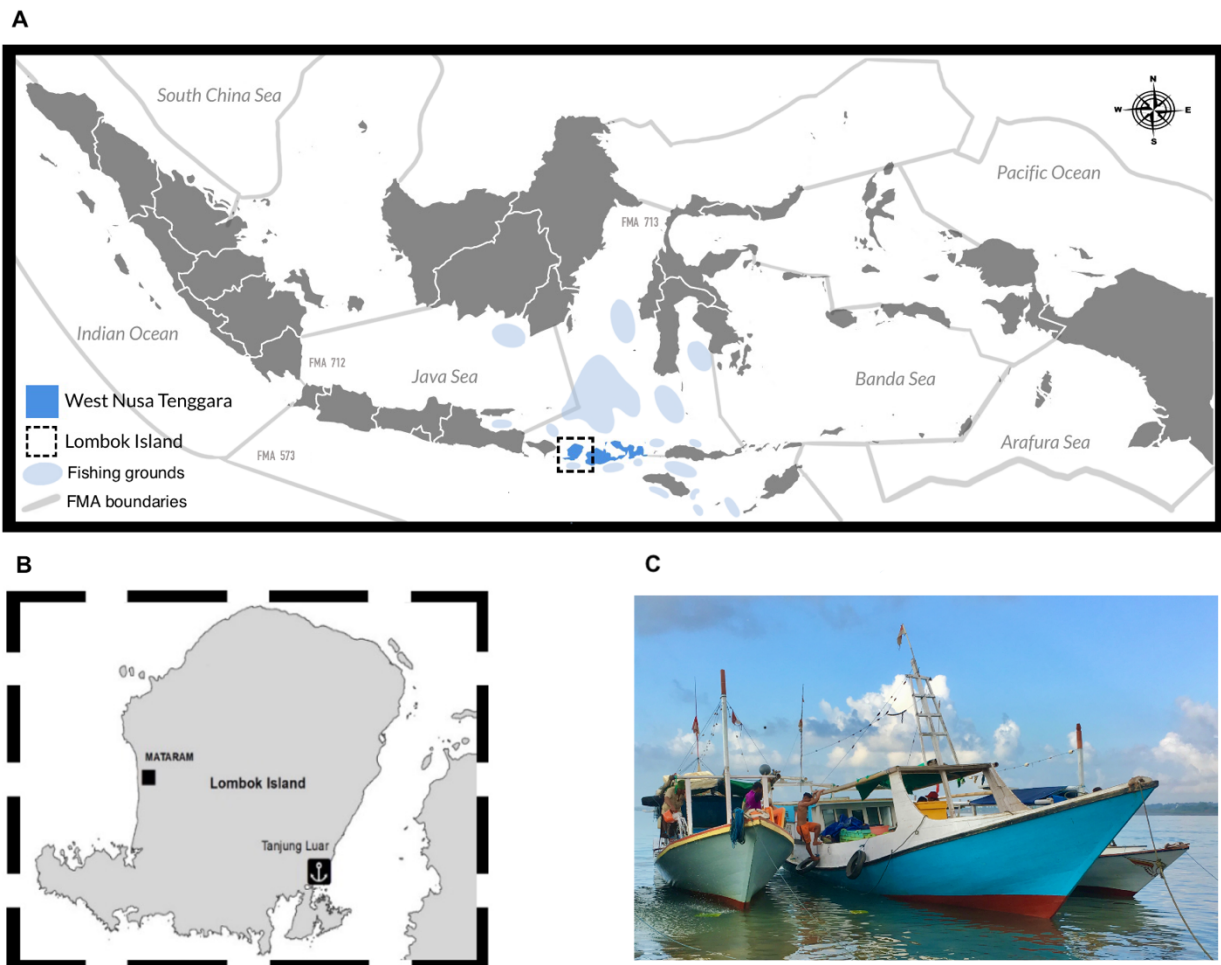


Figure 7.1 Study site: Tanjung Luar shark fishery. A. Location of Lombok Island, West Nusa Tenggara, and Tanjung Luar fishing grounds in Indonesia (FMA = Fisheries Management Area, only those which are relevant to Tanjung Luar are numbered), B. Lombok Island, C. Vessels used to target sharks

7.2.2 Data collection

Shark landings data were collected by three trained enumerators from the Wildlife Conservation Society Indonesia Program (WCS-IP). Landings and auction prices were recorded every morning at the Tanjung Luar shark auction facility between 5am and 10am from January 2017 to December 2018. Data collection methods follow established international protocols (Jaiteh, Hordyk, et al., 2017; SEAFDEC, 2016; I. Yulianto et al., 2018), with data recorded on operational variables, catch composition and the final auction price of total catches from each vessel/trip (Appendix 4, S7.1). This dataset was collected under a Memorandum of Understanding and Technical Cooperation Agreement between the WCS-IP and the Ministry of Environment and Forestry, Ministry Marine Affairs and Fisheries and the Marine and Fisheries Agency of West Nusa Tenggara Province.

7.2.3 Data preparation

I calculated total catch (number of individuals) and estimated total weight (based on empirical total length measurements and published length-weight relationships taken from FishBase (Froese & Pauly, 2021)) per vessel per trip. I also calculated the species composition of the catch, in terms of numbers of individuals per species per trip; and estimated total weight per species per trip. I grouped species into conservation-priority species and other species. I define conservation-priority species as Endangered or Critically Endangered species according to the IUCN Red List of Threatened Species (IUCN, 2020); internationally-regulated species (i.e., those listed on CITES appendices (UNEP-WCMC, 2020)); and commercially-important species (additional species which made up more than 1% of the total catch, or are of particularly high value in international markets (Fields et al., 2018; Hau et al., 2016, 2018; Wu, 2016)) (Table 7.1). This left 31 ‘other’ species, predominantly consisting of small rays (family Dasyatidae) and small requiem sharks (family Carcharhinidae), which together made up just under 6% of total catch for 2017-18 (Table 7.1). The data were not balanced across all species: catch is dominated by silky sharks (*Carcharhinus falciformis*), which make up almost 50% of total catch, followed by tiger sharks (*Galeocerdo cuvier*, 8.3%), scalloped hammerhead sharks (*Sphyrna lewini*, 7.2%) and blue sharks (*Prionace glauca*, 6.3%) (Table 7.1).

Table 7.1 Species groupings and justifications

Family	Species		>1% of total recorded catch	EN or CR	CITES	International commercial value*	Contribution to total catch (2017-18)	
	Common name	Scientific name					n	%
Aetobatidae	Longheaded Eagle Ray	<i>Aetobatus flagellum</i>		EN		None	1	<0.1
Alopiidae	Pelagic thresher	<i>Alopias pelagicus</i>	✓	EN	II	Moderate ²	196	1.5
	Bigeye thresher	<i>Alopias superciliosus</i>			II	Moderate ²	90	0.7
Carcharhinidae	Silvertip shark	<i>Carcharhinus albimarginatus</i>	✓			Moderate ²	185	1.4
	Grey reef shark	<i>Carcharhinus amblyrhynchos</i>	✓			Moderate ²	305	2.3
	Spinner shark	<i>Carcharhinus brevipinna</i>	✓			Moderate ²	775	5.9
	Silky shark	<i>Carcharhinus falciformis</i>	✓		II	Moderate ²	6,282	47.5
	Blacktip shark	<i>Carcharhinus limbatus</i>	✓			Moderate ²	275	2.1
	Oceanic whitetip shark	<i>Carcharhinus longimanus</i>		EN	II	High ¹	5	<0.1
	Dusky shark	<i>Carcharhinus obscurus</i>	✓	EN		High ¹	392	3.0
	Sandbar shark	<i>Carcharhinus plumbeus</i>	✓			High ¹	135	1.0
	Spot-tail shark	<i>Carcharhinus sorrah</i>	✓			Moderate ²	197	1.5
	Tiger shark	<i>Galeocerdo cuvier</i>	✓			Moderate ²	1,103	8.3
	Blue shark	<i>Prionace glauca</i>	✓			High ¹	830	6.3
Lamnidae	Longfin mako shark	<i>Isurus oxyrinchus</i>	✓	EN	II	High ¹	180	1.4
	Shortfin mako shark	<i>Isurus paucus</i>		EN	II	High ¹	12	0.1
Mobulidae	Shortfin devil ray	<i>Mobula kuhlii</i>			II	Moderate ²	1	<0.1
	Giant devil ray	<i>Mobula mobular</i>		EN	II	High ¹	21	0.2
	Chilean devil ray	<i>Mobula tarapacana</i>		EN	II	High ¹	13	0.1
	Bentfin devil ray	<i>Mobula thurstoni</i>		EN	II	Moderate ²	26	0.2
Dasyatidae	Jenkins whiplay	<i>Pateobatis jenkinsii</i>	✓			None	127	1.0
Rhinidae	Bowmouth guitarfish	<i>Rhina ancylostoma</i>		CR	II	High ¹	16	0.1
	Bottlenose wedgefish	<i>Rhynchobatus australiae</i>	✓	CR	II	High ¹	280	2.1
Sphyrnidae	Scalloped hammerhead shark	<i>Sphyrna lewini</i>	✓	CR	II	High ¹	947	7.2
	Greater hammerhead shark	<i>Sphyrna mokarran</i>		CR	II	High ¹	15	0.1
Stegostomatidae	Zebra shark	<i>Stegostoma tigrinum</i>		EN		None	17	0.1
Other	[31 non-priority species]						784	5.9

*Key: 1 = For sharks, high-grade/first choice fins in international markets; for mobulids, large high-value gill plates. 2 = For sharks, second-grade fins in international markets; for mobulids, smaller lower-value gill plates. Grading as per FAO classifications (Vannuccini, 1999).

7.2.4 The hedonic pricing method

Method background and justification

I adopted HPA to first test the hypothesis that taxa are implicit in catch auction prices, and then obtain marginal implicit prices and welfare estimates for conservation-priority species. The principle of HPA is that the utility provided by heterogeneous goods is based upon the utility yielded by their various attributes (Lancaster, 1966; Rosen, 1974; Taylor, 2003). In general, a class of heterogeneous goods (X) can be broken down into several valued attributes ($X_1, X_2, \dots, X_n$). A combination of these attributes, and the external factors that affect the goods, determines the price (Equation 7.1).

Equation 7.1.
$$\text{Price} = f(X_1, X_2, \dots, X_n)$$

Where X_i ($i = 1$ to n) is the amount of any one of the valued attributes describing a composite good.

HPA can be used to empirically estimate the implicit price of each attribute, by regressing market prices against a vector of explanatory variables representing the bundle of attributes. The regression analysis derives a hedonic price function, which indicates how much the price of the composite good will change as the quantity of each attribute (e.g., X_1) changes, holding all other attributes (X_{-1}) constant (Appendix 4, S7.2). The implicit price function for a given attribute (e.g., X_1) is the partial derivative of the hedonic price function with respect to X_1 ($\delta P / \delta X_1$). In a regression analysis, this is given by the linear model co-efficient for X_1 . The value of the co-efficient represents the buyer's marginal willingness-to-pay to acquire an additional unit of attribute X_1 , all other things equal, which is equivalent to the marginal price of the attribute (Day, 2001; Rosen, 1974; Taylor, 2003) (Appendix 4, S7.2).

In this study, total catches per trip represent the heterogeneous composite goods for which there is a total auction price. Prices are based upon the utility yielded from the marketable attributes derived from the catches, including fins, meat, skin and liver oil. These attributes will vary depending on species composition. Using HPA, I regressed total auction price per trip (in US\$, using a conversion rate of 1 IDR = 0.00007 US\$) against various iterations of catch composition and control (i.e., non-taxonomic) attributes. This enabled me to test whether taxa are implicit in auction prices, understand the influence of catch composition on auction price, and use model co-coefficients to estimate implicit marginal prices for different taxa in the fishery.

The linear models take the following form, where I focus on estimating β , a vector of coefficients indicating how catch composition is associated with catch prices (from β I can estimate the marginal implicit price of taxa included in the model):

Equation 7.2.
$$P_{i,t} = f[\alpha + (\beta \times \text{catch}) + (\gamma \times \text{other})]$$

Where:

P is the price of total catch per trip *i* sold at auction at time *t*.

catch is a vector of variables describing the taxonomic composition (attributes) of the total catch per trip.

Other is a vector of non-taxonomic explanatory control variables, such as year and total catch volume.

Model estimation and selection

I used a linear functional form, in which regression coefficients represent implicit marginal prices (Benoit, 2011; Melichar et al., 2009; Taylor, 2003), and tested various iterations of the catch composition vector. I investigated biologically and economically meaningful catch composition attributes, based on published literature. These attributes are inter-related, since fins (sharks) and gill plates (mobulid rays) constitute 25-35% of the total value of an individual (despite making up less than 5% of the total weight) (Clarke et al., 2006; Wu, 2016); and the economic value of fins and gill plates are determined by size, yield of fin needles, colour and texture, which in turn vary by taxa (Hau et al., 2016, 2018; Wu, 2016). I used catch per vessel per trip as the unit of analysis, since price variation occurs at the vessel/trip level (Appendix 4, S7.3). I compared model fits using Adjusted R-squared and Δ AIC (Akaike Information Criterion).

I first explored the relative benefit of expressing species composition in terms of numbers of individuals (*n*) or weight (kg). The preliminary results indicated that catch expressed in numbers is better at explaining variation in price than catch in weight (Appendix 4, S7.3). This makes market sense since fins and gills are the highest value derivative products (Clarke et al., 2006; Wu, 2016). Using numbers of individuals also reduced potential measurement bias. I then explored the relative benefit of increasing taxonomic specificity for explaining variation in price, by first using clade (i.e., shark and ray) as explanatory variables, then breaking this down into families, then species. The preliminary results indicated that more taxonomic specificity in the explanatory variables improved the explanatory power of the model (Appendix 4, S7.3). For the final the regression analyses I grouped some species into biologically and economically meaningful groups (Table 7.2). For example, hammerhead sharks and thresher sharks were grouped by family, since there are no major species-specific distinctions within these groups in international markets. However, silky sharks, dusky sharks and tiger sharks could not be grouped with all carcharhinids, since there is enough fin variation to warrant different prices for different species (Vannuccini, 1999; Wu, 2016).

I included non-taxonomic control variables: total catch (in terms of numbers of total weight in kg (TW) and number of individuals (TC)), to control for total catch volume; a categorical time trend (TEMP) to control for temporal trends such as seasonal price trends and general increases in price within the broader

economy; and vessel ID (V-ID) as a random effect, to account for variations in relationships between vessels and buyers which might affect prices, as well as exogenous vessel-related factors such as vessel size, engine size, crew number and gear type (Table 7.2). For final model selection I included all taxonomic variables and tested the impact of removal of non-taxonomic variables and inclusion of TC vs. TW on R-squared and Δ AIC (Appendix 4, S7.3).

I tested linear model assumptions (including linearity, normality of residuals, homogenous variance and influence/sensitivity to data) and robustness using: 1) visual inference based on diagnostic plots; and 2) several statistical tests. These tests included Granger causality and Durban-Wu-Hausman to test for endogeneity (i.e., confirm that catch causes price, and is not simultaneously determined with price), Breusch-Pagan Lagrange multiplier (LM) to test for panel effects (auto-correlation/homogenous variance across entities), and Hausman to test for random versus fixed effects for vessel ID (Colonescu, 2016; Torres-Reyna, 2010) (Appendix 4, S7.4).

Table 7.2 Explanatory variables (attributes) included in the hedonic pricing regression analysis

Var type	Code	Description	Data type
Taxa	ALO	Number of thresher sharks (family Alopidae, 2 species)	Numeric
	ALS	Number of silvertip sharks (<i>Carcharhinus albimarginatus</i>)	Numeric
	FAL	Number of silky sharks (<i>Carcharhinus falciformis</i>)	Numeric
	DUS	Number of dusky sharks (<i>Carcharhinus obscurus</i>)	Numeric
	TIG	Number of tiger sharks (<i>Galeocerdo cuvier</i>)	Numeric
	LAM	Number of mako sharks (family Lamnidae, 2 species)	Numeric
	MOB	Number of devil rays (family Mobulidae, 4 species)	Numeric
	DHJ	Number of Jenkins whipray (<i>Pateobatis jenkinsii</i>)	Numeric
	RCS	Number of bottlenose wedgefish (<i>Rhynchobatus australiae</i>)	Numeric
	CAR	Number of ‘other’* carcharhinids (family Carcharhinidae, 6 species)	Numeric
	SPH	Number of hammerhead sharks (family Sphyrnidae, 2 species)	Numeric
Non-taxa	TW/TC	Total catch expressed as weight in kg (TW) / number of individuals (TC)	Numeric
	TEMP	Categorical time trend, by season (4 seasons: East (June—September), Transition II (October—November), West (December—March), Transition I (April—May)) by year (2 years: 2017 and 2018); 8 levels in total	Categorical
	V-ID	Unique ID for each vessel, included as random effect	Categorical

*Not threatened, CITES-listed or of species-specific commercial importance

Prior to analysis, all data rows were checked for errors, with abnormal data points clarified or removed. I removed trips which made negative profits and/or caught fewer than 10 sharks in total, as these can be considered ‘failed trips’ (based on informal discussions with fishers, indicating that they aim to capture at least 20-30 sharks before returning to port, to at least cover operating costs). The resulting dataset contained 581 market transactions, of which 43 were removed due to missing data, resulting in 538 observations used in final model estimation. Taxonomic groups with less than 60 catch records (<0.5% of total catch) were not included as independent model variables, to ensure sufficient statistical power.

All data manipulation, statistical analyses and graphical outputs were prepared in R Studio (RStudio Team, 2020), using packages ‘skimr’, ‘dplyr’, ‘plyr’ and ‘readr’ for data preparation; lme4 for mixed-effects

modelling; 'ggplot2' for plots; and packages 'AER', 'MuMIn', 'lmerTest' and 'performance' for testing linear model assumptions.

7.2.5 Estimation of economic opportunity costs

Based on the results of the HPA I use welfare measurement methods (Day, 2001; Taylor, 2003) to estimate the economic opportunity costs of various plausible policy scenarios to reduce fishing of conservation-priority species. For this part of the analysis I focused only on conservation-priority species for which statistically significant co-efficient estimates were obtained in the HPA.

Cost per average trip

I use price (p) multiplied by quantity (q) to estimate economic welfare changes associated with changes in the supply of conservation-priority taxa per average trip. The model coefficients from the HPA are used as a shadow price (i.e., an estimated price for something that is not normally priced or sold in the market) per unit (p), which is valid for localised, marginal changes (Taylor, 2003). The mean catch per trip for each taxon (across all recorded trips) is used to estimate the change in quantity supplied per trip (q). I use average catch per vessel per trip, so that the changes in supply remain small (i.e., they approximate marginal changes). These are estimates of average economic opportunity costs, based on several assumptions (Appendix 4, S7.5), following methods previously applied to labour markets to estimate the value of statistical life (Taylor, 2003).

Total change in economic welfare

For non-marginal changes in economic welfare (ΔW), such as estimating the economic cost of large changes in quantities of taxa supplied across the entire Tanjung Luar fishery, the total change in welfare (across all vessels per year) is forecast using the hedonic price function (equation 7.3).

Equation 7.3.
$$\Delta W = \sum_{i=1}^N P_i(X_{1i}^1, X_{2i}^0, \dots, X_{ni}^0) - P_i(X_{1i}^0, X_{2i}^0, \dots, X_{ni}^0)$$

Where:

ΔW is the total change in welfare, or in this case net opportunity cost, summed across all (N) vessels for a given year

$P_i(X_{1i}, X_{2i}, \dots, X_{ni})$ is the ex-ante hedonic price function, predicted across the attributes of the i^{th} fishing trip (per vessel per time category) in the initial (i.e., current) state and the new state.

I applied this equation using the estimated hedonic price function, first to predict the hedonic price and total value of average catches per vessel per time category in the current state (i.e., the baseline/business as usual (BAU) scenario), and then predict the hedonic price and total value of average catches per vessel per time category under three new hypothetical states based on plausible policy scenarios to reduce

fishing of three conservation-priority species (bottlenose wedgefish, dusky sharks and silky sharks). I summed and compared the annual value of the BAU and new hypothetical states, based on 2018 predictions only, to estimate a total annual value and ΔW associated with each scenario. This method assumes no transaction costs, and estimates an upper bound of the welfare change associated with the attribute change (Taylor, 2003). This does not consider potential adaptation or substitution by fishers, however, it does consider concurrent changes in total weight associated with changes in quantities of taxa.

7.3 Results

7.3.1 Model outputs

Including taxonomic composition variables improves the explanatory power of price models, in comparison to modelling price against total catch alone (Table 7.3; Appendix 4, S7.3). The modelling process also indicated that total weight and year were important as control (i.e., non-taxonomic) variables, with weight as a better variable to control for total catch volume than number of individuals (Appendix 4, S7.3).

The lowest AIC/highest R-squared model was a linear mixed-effects model including all taxa, plus total weight, a time trend and vessel-ID as a random effect (Table 7.3; Appendix 4, S7.3). Significant marginal prices were estimated for silvertip sharks, silky sharks, dusky sharks, bottlenose wedgefish and other carcharhinids (Table 7.3). The model was a relatively good fit to the data (adjusted R-squared = 0.65), and met model assumptions as per diagnostic plots and statistical tests (Appendix 4, S7.4). The standard errors reflect what would be expected, given the proportion of species in the catch and sample size (Table 7.3; Appendix 4, S7.3) i.e., the confidence intervals are tightest for the species groups with the largest catches. However, unlike the other most frequently caught species, the co-efficient for hammerhead sharks was insignificant.

Table 7.3 Best-fit hedonic model

Variables		Co-efficient	Standard Error	Significance
Taxon-related	Thresher sharks (ALO)	34.31	22.17	
	Silvertip sharks (ALS)	89.44	32.67	**
	Silky sharks (FAL)	55.06	4.75	***
	Dusky sharks (DUS)	157.45	25.38	***
	Tiger sharks (TIG)	54.04	11.74	***
	Mako sharks (LAM)	58.42	40.57	
	Devil rays (MOB)	-4.21	80.11	
	Jenkins whipsnappers (DHJ)	-40.78	41.82	
	Bottlenose wedgefish (RCS)	143.99	33.73	***
	Other carcharhinids (CAR)	-42.78	10.89	***
Non-taxonomic (control)	Hammerheads (SPH)	13.71	12.40	
	Total weight (TW)	0.74	0.10	***
	Temporal (TEMP)	✓		
	- 2017/Transition I	-21.49	145.41	
	- 2017/Transition II	134.84	169.21	
	- 2017/West	-4.36	147.69	
	- 2018/East	329.41	127.11	**

- 2018/Transition I	-224.11	143.21	
- 2018/Transition II	935.60	140.39	***
- 2018/West	89.83	145.92	
Vessel ID (V-ID) (as random effect)	✓		
Adjusted R-squared	0.652		
AIC	8274.1		

Significance codes: *** <0.001, ** <0.01, * <0.05. Standard error at 95% confidence intervals. Model checked for linear model assumptions and robustness (Appendix 4, S7.4).

7.3.2 Price functions per taxa

Conditional plots can be derived for each variable in the model. Each plot represents the hedonic price function for a given taxa, which demonstrate the relationship between total catch price (US\$) and quantity of taxon supplied, other things equal. The gradient of these plots (i.e., the linear model coefficients) can be interpreted as the implicit marginal prices for each taxonomic group included in the model i.e., the marginal increase in total auction price (δP) (in US\$) associated with adding one additional individual of that taxon (δQ_i) (Appendix 4, S7.6). Based on this I can estimate the marginal economic value per individual shark per taxon per trip. The models show that wedgefish, dusky sharks, silvertip sharks and silky sharks are amongst the highest value species in the fishery (Table 7.4, Figure 7.2). For example, the marginal price of a Critically Endangered wedgefish in the Tanjung Luar shark fishery can be estimated at US\$110 – 177, relative to the average shark ($p < 0.001$). Similarly, the marginal price of an endangered dusky shark is estimated at US\$132 – 182 ($p < 0.001$). On the other hand, other carcharhinids (which includes *Carcharhinus plumbeus*, *C. sorrah*, *C. limbatus*, *C. amblyrhynchos* and *Prionace glauca*) have negative marginal values relative to the average shark ($p < 0.001$) (Table 7.4, Figure 7.2).

Table 7.4 Interpretation of hedonic models in marginal price terms. Marginal price shows the additional value of a catch with the addition of one individual of the given taxonomic group. High and low estimates are based on standard errors at 95% confidence intervals.

Taxa	Priority criteria		Marginal price (US\$)			
	Threat status	CITES	Estimate	Sig	High	Low
Bottlenose wedgefish	CR	II	144	***	177	110
Dusky shark	EN		157	***	182	132
Silvertip shark	VU		89	**	122	56
Silky shark	VU	II	55	***	59	50
Mobula rays	EN		-4		75	-84
Thresher sharks	EN	II	34		56	12
Hammerhead sharks	CR		14		26	1
Tiger shark	NT	II	54	***	65	42
Mako sharks	EN		58		99	17
Jenkins whiplay	VU		-41		1	-82
Other Carcharhinids	NT/VU		-43	***	-31	-53

Significance codes: *** <0.001, ** <0.01, * <0.05.

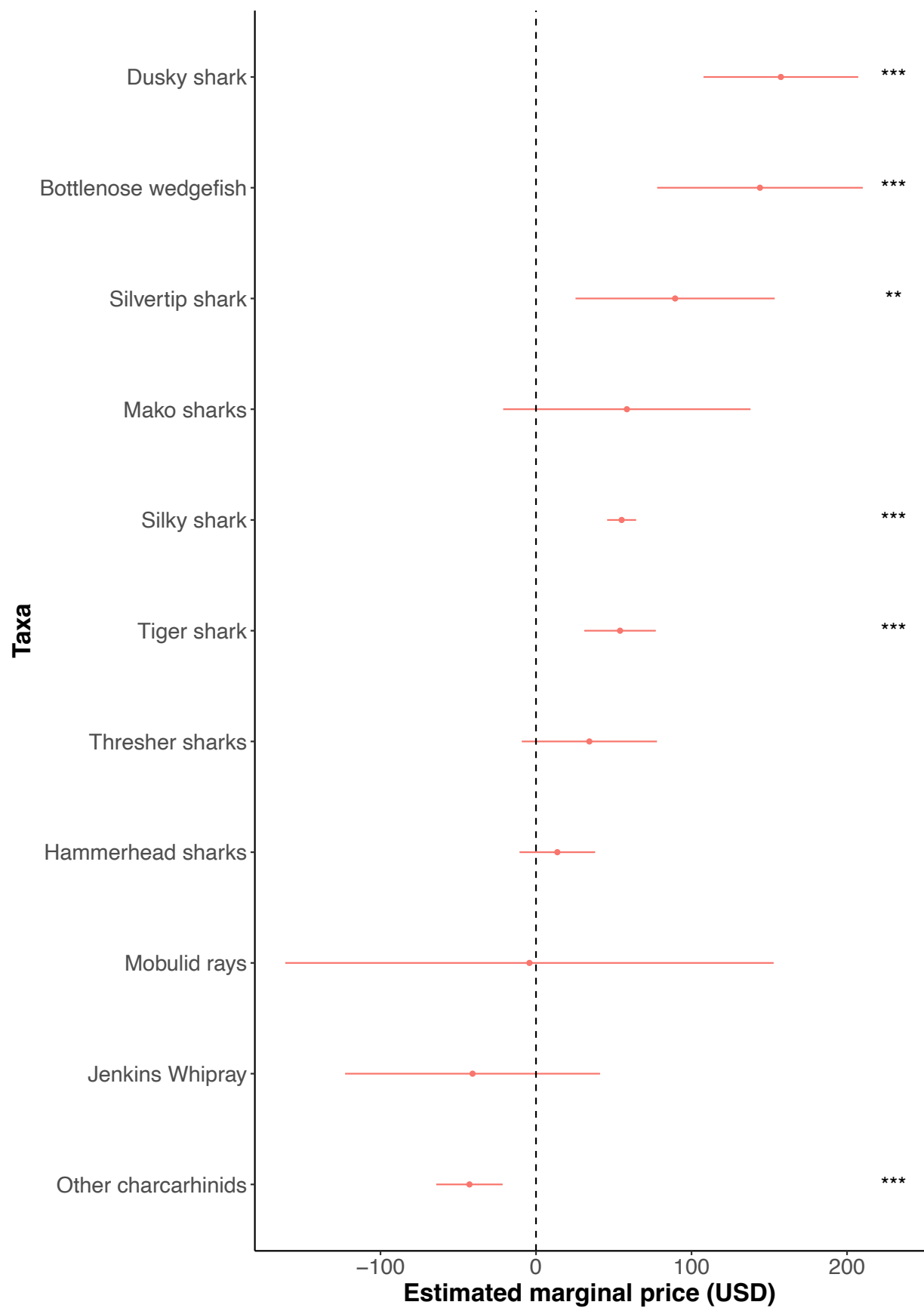


Figure 7.2 Plots of taxa coefficients from best-fit hedonic models, where coefficients represent marginal implicit prices per taxa (i.e., US\$ per individual shark)

7.3.3 Estimating the economic opportunity costs of shark conservation

These price estimates allow for exploration of the economic welfare costs (and potential benefits) of plausible management scenarios (Table 7.5, Figure 7.3).

Cost per trip

On average, 2.3 (S.D. 1.6) bottlenose wedgefish were landed per trip in Tanjung Luar during the survey period. This species is Critically Endangered and CITES-listed, therefore a plausible policy scenario could be a total catch ban. Based on a shadow price of US\$144 per individual, the expected economic welfare loss of a catch ban for bottlenose wedgefish would be US\$325 per trip (13% of average revenue per trip), all other things equal (Table 7.5). By volume, silky sharks make up the majority of Tanjung Luar's shark catch with 16.7 (S.D. 13.4) individuals landed per trip. Management measures such as catch quotas are needed to implement CITES for this species. At an implicit marginal price of US\$55, a quota requiring a 33% reduction in catches could reduce revenue by US\$304 or 12% per trip (Table 7.5), all other things equal.

Table 7.5 Estimates of the economic costs of shark management scenarios for priority species. Estimated marginal prices are derived from significant model coefficients and standard error at 95% confidence intervals (Figure 7.3), estimated economic cost per trip is calculated based on marginal revenue (i.e., q^*p).

Taxa	Priority criteria		Hypothetical policy scenario	Mean catch per trip	Estimated marginal price (US\$)			Estimated economic opportunity cost per trip of policy scenario (US\$)		
	IUCN	CITES			High	Estimate	Low	High	Estimate	Low
Silky shark	VU	II	Reduce catch by 33%	16.7 (S.D. 13.4)	60	55	50	330	304 (12%)	277
Dusky shark	EN		Catch ban	2.5 (S.D. 2.1)	183	157	132	451	388 (16%)	326
Bottlenose wedgefish	CR	II	Catch ban	2.3 (S.D. 1.6)	178	144	110	401	325 (13%)	249

Total change in economic welfare

Ex-ante predictions using the hedonic price function estimate that a 33% reduction in silky shark catches could lead to a US\$59,798 economic opportunity cost per year across the fishery against a 2018 baseline of US\$770 959 (Figure 7.3). This represents a 7.8% loss. Similarly, a dusky shark ban could cost US\$47,886 (6.2%) while a wedgefish ban could cost US\$27,730 (3.4%) across the fishery (Figure 7.3). Combining all these policies would create a combined opportunity cost of US\$135,414; a 17.6% loss to the total value of the fishery against the 2018 baseline.

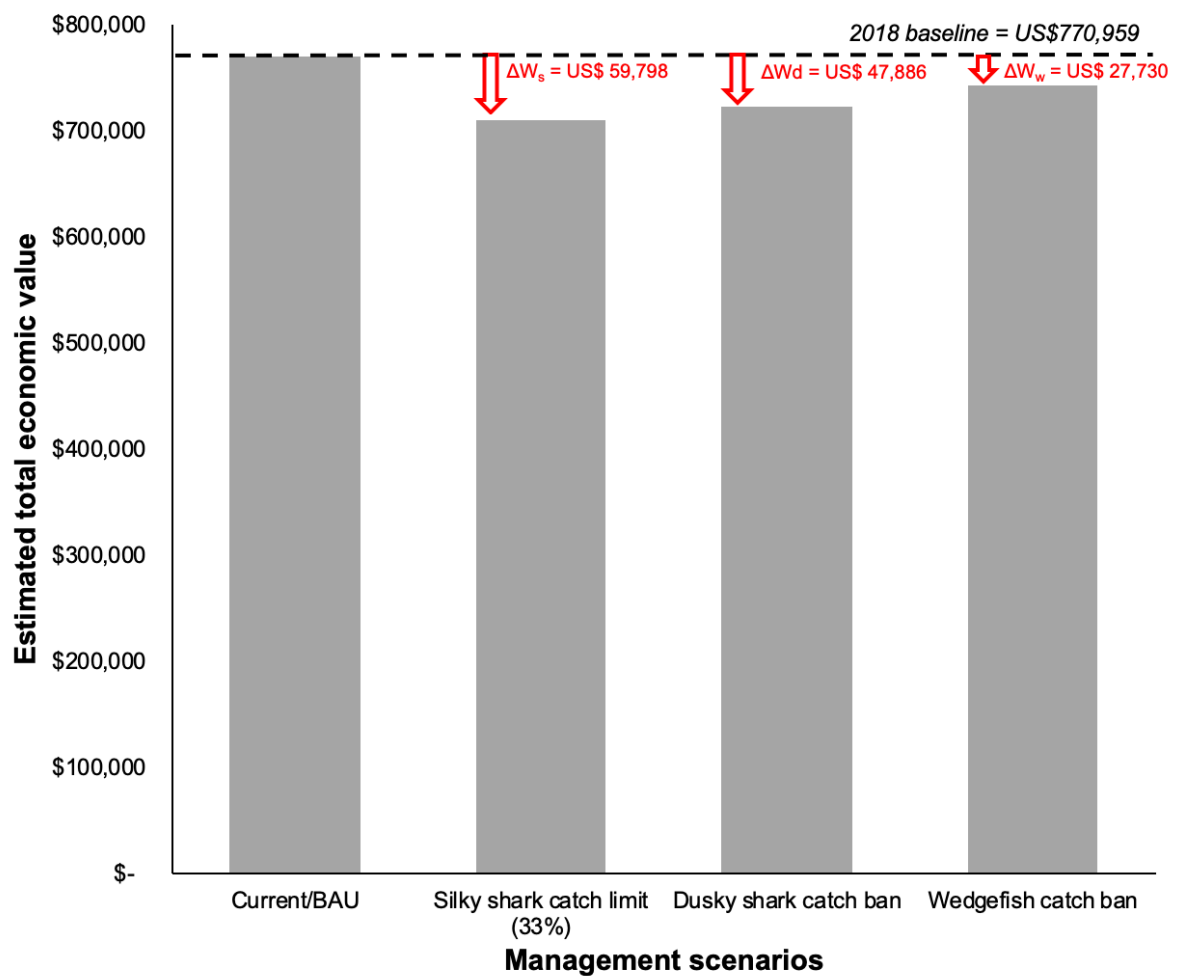


Figure 7.3 Estimated annual opportunity costs of policy scenarios across the entire fishery

7.3.4 Non-taxonomic explanatory variables

As well as the taxa-specific explanatory variables, total weight, temporal trend and vessel-ID were important variables. The total weight co-efficient (0.74) suggests that the marginal price of an extra kilogram of catch is US\$0.74. The temporal variable indicates that auction price is typically lower in Transition I and West seasons, and higher in Transition II and East Season in 2018 ($p < 0.001$), with East season in 2017 as the reference year.

7.4 Discussion

I used HPA to estimate implicit market prices and welfare measures for threatened and CITES-listed sharks in a small-scale targeted shark fishery in Indonesia. The results have important implications for understanding the economic opportunity costs of shark conservation to small-scale fishers, and the need to consider these costs to communities when designing management interventions.

7.4.1 Model interpretation and validation in the context of international shark trade

The price estimates from the hedonic models directionally reflect what would be expected given the state of international shark and ray markets at the time of data collection, which supports the external validity of the method and results. For example, fins from bottlenose wedgefish and dusky sharks are two of the most highly-valued in international markets (Hau et al., 2018; Wu, 2016). Dusky shark fins belong to a specific category (Hai Hu Fin (海虎翅)) and can fetch US\$240 – 430 in China and Hong Kong (Clarke et al., 2006; Wu, 2016). Similarly, fins from shark-like batoids (order Rhinopristiformes) are categorised as Qun chi (群翅 / 裙翅), and recognised as ‘king of shark fins’ due to their high quality and fin-needle yields (Hau et al., 2018; Jabado, 2019). Processed Qun chi are sold for US\$560/kg on average, and can fetch over US\$1,600/kg (Hau et al., 2018). Silky shark ‘Gou’ (i.e., caudal) fins are also moderately valuable, sold for US\$26 – 591 in China and Hong Kong (Wu, 2016), and the most frequently traded fins, based on Hong Kong retail market surveys (Fields et al., 2018).

It is perhaps surprising that mobulid rays were one of the less valuable taxa. However, there is a large standard error around the estimate, which makes it statistically indistinguishable from zero. This is likely due to the small number of market transactions, yielding limited explanatory power; and the need to combine four different species, leading to aggregation bias. For example, larger mobulid gills typically fetch higher values (Hau et al., 2016), such that larger devil ray species (e.g., *Mobula mobular*, which can reach up to 5.2m disc width) are likely to be more valuable than smaller species (e.g., *Mobula thurstoni*, which typically only reach 1.8m disc width). In international markets, devil ray gills sell for US\$73 – 350/kg, depending on the size and quality, though average at US\$87/kg (Wu, 2016). Wu (2016) also noted that some market retailers consider mobulid ray gills a ‘low-price’ tonic ingredient. The listing of all mobulids on CITES Appendix II, and the full protection of manta rays in Indonesia in 2014, may have also impacted market prices. A species-specific assessment may reveal more accurate and diverging prices for different species, should sufficient data be available in the future.

Mako sharks, hammerhead sharks and thresher sharks make up a relatively small proportion of the international fin trade (Fields et al., 2018). While previous studies have claimed that hammerhead and mako sharks are classified amongst the top grade fins (Vannuccini, 1999), more recent studies suggest these species aren’t specifically recognised or valued amongst retailers in Hong Kong. For example, Wu (2016) found one hammerhead fin in Hong Kong valued at US\$134/kg, while no specific categorisation was noted for mako or thresher sharks. The fins from these species are also, on average, smaller than those from dusky sharks, silky sharks and wedgefish (Clarke et al., 2006) and therefore represent lower fin needle yields. As with mobulids, CITES-listings of hammerhead sharks and thresher sharks in 2016 may have impacted market values, though further investigation into broader market dynamics would be required to confirm this.

The other carcharhinids had negative marginal prices, while the marginal price of the Jenkins whiptail was not statistically different from zero. However, individuals from these taxa do not have no or negative absolute market value. Rather, these prices represent marginal values holding all other variables equal. Jenkins whiptail have little-to-no commercial value (other than for local meat trade) and smaller carcharhinids fetch lower values in the fin trade. If these species are captured, they create an opportunity cost by taking up hold space and therefore reducing the number of other more valuable species that could be caught and sold. However, in the absence of other catches, they are still worth transporting to market for sale. I also note that informal discussions with shark fishers suggests there is an informational asymmetry between fishers and traders, in which the fishers are not necessarily aware of the relative market values of the different taxa, which may also be obscured by the ‘bundling’ of the catch. As such, fishers may be unlikely to discard these catches for high-grading purposes, since they do not know which taxa are the most valuable.

The inclusion of total weight in the model also indicates that total catch volume is important as well as catch composition. A higher volume lowers unit transaction costs through economies of scale, thereby increasing profits. However, the marginal value per kilogram is low (US\$0.74), which reflects the low value attributed to non-fin commodities that are volume-dependent, such as meat. This also means that species which are larger on average, such as tiger sharks, dusky sharks and hammerhead sharks, will add additional marginal value to the auction price by virtue of their extra weight. For example, a 100kg tiger shark will add an extra US\$74 on top of its marginal taxa-specific value.

The coefficients for the temporal variables may be explained by seasonal fluctuations in supply and demand, leading to price signals. Adverse weather during the West monsoon and Transition I seasons (December to May) may impact catch supply and therefore auction prices (I. Yulianto et al., 2018). Transition II season (October – November) in 2018 appears to be associated with particularly high prices ($p < 0.001$), which may be indicative of a change in supply-demand dynamics over time. Observations of mean total catch per trip over the time periods suggests a small but significant general decline in total catch over time. This reduced supply may be driving a long-term increase in auction price, particularly since the Tanjung Luar shark fishery is the only targeted shark fishery in West Nusa Tenggara province, and is therefore not closely integrated by price or commodity flows with other shark markets. This also fits with the results of the Granger causality test, which implied that catch Granger-causes (i.e., predictably forecasts) price (Appendix 4, S7.4).

The inclusion of Vessel ID in the linear model may be an indicator of debt and power relationships between boat owners and buyers (Lestari et al., 2017), and differences in skipper skill, vessel characteristics and handling for quality. That this effect is random suggests vessel-specific characteristics are relatively widely felt and diffuse across the fishery.

Finally, it is worth noting that while the model explains a considerable amount of the variation in price (adjusted R-squared = 0.65) some variation remains un-explained. This may be due to variables that were not collected or included in the model, such as within-taxon variation in sizes and variation in catch quality.

7.4.2 The costs and benefits of managing shark fisheries

I provided a first estimate of the potential economic opportunity costs of shark conservation in a small-scale fishery, and the results have implications for displacement effects, market distortions and the interplay between conservation and social justice.

In terms of displacement, assuming perfect species substitutability, such that it is possible for fishers to switch between conservation-priority and non-priority species, fishers might need to catch 110-150% more tiger sharks or silvertip sharks per trip (which are both classified as Vulnerable according to the IUCN Red list; current mean value per trip = US\$217 and US\$293, respectively) to make up for economic losses from a 33% reduction in silky shark catches. Similarly, average economic losses of US\$325 per trip from a bottlenose wedgetail ban could cause even greater displacement effects on other taxa. This raises concerns regarding unintended consequences of wildlife policies, which can induce shifts towards other damaging activities, or on to other vulnerable species and ecosystems, with potentially perverse outcomes (Booth, Mardhiah, et al., 2020; Booth, Clark, et al., 2021; Suuronen et al., 2010). As such, any restrictions on shark fishing should come with clear adaptation and transition plans that can move fishers towards sustainable practices.

In the absence of displacement or adaptation, fisher households may lose significant portions of their income. To put the results in to context, the estimated opportunity costs per trip are higher than the average monthly income of shark fishers (crew) in Tanjung Luar (US\$233) and 2-4 times the minimum monthly wage for manual workers in West Nusa Tenggara Province (US\$133 per month) (Lestari et al., 2017; WageIndicator, 2020). This raises ethical concerns regarding who should bear the costs of shark conservation, as well as compliance management issues, whereby unacceptable social costs or large financial incentives for non-compliance may render policies unfavourable and unimplementable (Keane et al., 2008; Margavio & Forsyth, 1996; R. G. Smith & Anderson, 2004). These estimates provide a first indication of some of the socio-economic trade-offs that might result from management measures for threatened and CITES-listed sharks, and can inform least-cost policy formulation, through understanding how much conservation can be achieved per unit cost (Booth, Squires, & Milner-Gulland, 2019a; Squires & Garcia, 2018).

On the other hand, these values also provide an economic argument for sustainably managing shark populations. Given declines in shark populations worldwide (MacNeil et al., 2020; Pacoureau et al., 2021), it is unlikely that many of these populations can sustain current catch levels in the long-term, particularly in the absence of careful fisheries management. As such, while reduced catches may represent a short-term opportunity cost of foregone catches while stocks re-build, they could provide long-term financial benefits through a continued well-managed fishery for faster-growing species (Simpfendorfer & Dulvy, 2017). This may be feasible for blue and silky sharks, for example, which are relatively faster-growing and more abundant, and for which some examples of sustainable fisheries exist (Bonfil, 2009; LIPI, 2018; Simpfendorfer & Dulvy, 2017). However, this may be more challenging for Critically Endangered species such as wedgefish, which require more stringent catch and trade limits due to their extremely high extinction risk (Kyne et al., 2020).

Despite these research findings, it may be impractical to reduce catches of conservation-priority species in the absence of an entire fishery closure or large reduced catches of associated species (e.g., due to issues with selectivity, post-release mortality, interactions with other fisheries and economic viability (Smart et al., 2020)). The temptation for high-grading and discarding might be high, particularly if there is no feasible operational adaptation to avoid catching restricted species. Similarly, reductions in supply may lead to increases in market prices, particularly for species with high prices and inelastic demand (Courchamp et al., 2006). This would further distort incentives and may drive illegal fishing. Overall, further exploration of supply-demand dynamics and substitutability is required to attain more accurate estimates of opportunity cost, and likely responses of price to market interventions.

7.4.3 Future directions

The results indicate that endangered and CITES-listed species continue to be economically important in small-scale fisheries in Indonesia. This underlines the importance of providing resources for CITES implementation, since meaningful domestic measures must be implemented in major shark fishing nations for CITES to drive conservation impact. Yet many of the world's largest shark fishing nations are also highly-dependent on marine resources, necessitating a nuanced and socio-economic approach (Booth, Squires, & Milner-Gulland, 2019; Golden et al., 2016; Selig et al., 2018). A better understanding of domestic demand (e.g., for shark meat) and local-level drivers of shark fishing (e.g., for profit vs. subsistence) is needed to inform domestic measures and fisheries management. In parallel, demand-side countries can also play a role in driving changes through the supply chain – e.g., through monitoring and enforcing quota and permit systems, or creating incentives for sustainability (X. Zhou et al., 2021) – to further reduce trade-driven overexploitation of sharks. An improved understanding of the entire shark value chain, including domestic and international price leaders, and integration by price and commodity flows with other markets, will be important for identifying future market-based leverage points.

These findings also contribute to broader debates regarding reconciling biodiversity conservation with social justice and human rights (Newing & Perram, 2019; Shoreman-Ouimet & Kopnina, 2015), and in particular the need to ensure the costs of conservation are equitably distributed, and that conservation interventions ‘do no harm’ (Balmford & Whitten, 2003; N. J. Bennett et al., 2019; Giron-Nava et al., 2021; V. F. Griffiths et al., 2019). In small-scale fisheries, one option for simultaneously delivering conservation and social welfare outcomes could be through compensation or payment for ecosystem service schemes, which incentivise fishers to reduce capture of the most threatened species while maintaining material well-being (Bladon et al., 2016; Booth, Arlidge, et al., 2021; Wosnick et al., 2020). Given the high value of the shark and ray tourism industry in Indonesia (Mustika et al., 2020; O’Malley et al., 2013), it could also be possible to gather funding through tourism taxes or donations, and channel this into conservation, including funding protected areas in critical shark habitat, or fisher compensation for economic losses incurred from not catching sharks (Sykes et al., 2018; Vianna et al., 2018). Similarly, bycatch levies could be introduced for commercial fisheries, with the funds invested in critical habitat conservation and assisting small-scale fishers to adapt (Booth, Arlidge, et al., 2021; Gjertsen et al., 2014; Pakiding et al., 2020).

Overall, I reiterate the importance of understanding markets, and considering opportunity costs and equity when designing interventions to reduce overexploitation of traded species. To do so, I encourage other conservation scientists to integrate methods from economics into conservation decision-making, to better understand both supply- and demand-side drivers of overexploitation, explore the economic and social welfare implications of conservation interventions, and mitigate unintended consequences. This is particularly important for reducing shark fishing mortality in small-scale fisheries, which are influenced by both local-level needs and macro-economic market forces (Booth, Squires, & Milner-Gulland, 2019; Collins et al., 2020; MacKeracher et al., 2020). Rapid transformations of global markets and local fisheries are needed to save species on the brink of extinction, such as sawfish, wedgefish and hammerhead sharks. Innovative socio-economic interventions and creative institutional arrangements will be required to support these transformations, to effectively change fishing, trading and consumer behaviour, and achieve positive outcomes for sharks and people.

8. Designing locally-appropriate conservation incentives for small-scale fishers



Photo: a fisher proudly holds up his wedgefish catch for the camera in Lhok Rigaih.

“...it is time for conservationists... to ensure that conservation actions are not only effective, but also... morally responsible. We need ... to engage in honest discussion about genuine conflicts of interest where these exist and work towards negotiated settlements, with full respect for rights as the bottom line”

- Newing and Perram.

8.1 Introduction

Ocean ecosystems are threatened by overexploitation; with large, long-lived marine animals (‘marine megafauna’) – such as turtles, sharks and cetaceans – classified as some of the world’s most endangered taxa (Dulvy et al., 2021; IUCN, 2021). These taxa comprise ancient, diverse and charismatic species, which play critical roles in generating marine ecosystem services and well-being (Pimiento et al., 2020; Stein et al., 2018). As such, loss of these species not only threatens biodiversity itself, but also the ability of the ocean to sustain life on earth. International policy frameworks, such as the Convention on Biological Diversity (CBD) and the Sustainable Development Goals (SDGs), outline ambitious goals for conserving biodiversity and ecosystems, such as “*living in harmony with nature*” by 2050 (CBD, 2020) and “*conserve and sustainably use the oceans... for sustainable development*” (SDGs). However, achieving these goals requires transformative change to address the anthropogenic drivers of biodiversity loss (Díaz et al., 2019).

Almost all threats to nature derive from human actions (Balmford et al., 2021). Marine megafauna are primarily threatened by overfishing – captured throughout the world’s oceans, in fisheries of all types (Lewison et al., 2004). With inherently slow life-history traits, this fishing pressure exceeds the rate at which most marine megafauna populations can replenish. For example, global abundance of oceanic sharks has declined by over 70% in the past 50 years, attributed to an 18-fold increase in relative fishing pressure (Pacoureau et al., 2021). Technologies and practices that reduce fisheries impacts on marine megafauna are well documented (e.g., BMIS, 2021), but less is known about how to encourage their adoption. Doing so is challenging, because it requires changing human behaviour amidst trade-offs between biodiversity conservation and the important socio-economic roles of fisheries (Booth, Arlidge, et al., 2021; Campbell & Cornwell, 2008). This is particularly problematic in small-scale mixed-species fisheries, where almost all catches have economic or subsistence value. Such fisheries are ubiquitous throughout coastal waters, especially in biodiversity-rich low-latitude developing nations, which are often highly dependent on marine resources (Golden et al., 2016; Selig et al., 2018). For example, Indonesia is a global hotspot of marine species diversity and endemism, and also the world’s second largest fishing nation, with >99% of vessels classified as small-scale (FAO, 2018; Selig et al., 2014). In these contexts, efforts to reduce exploitation of marine megafauna may leave coastal communities facing an inequitable burden of the costs of conservation, with negative impacts on the well-being of some of the world’s most vulnerable people (Booth, Squires, et al., 2021; Jaiteh, Loneragan, et al., 2017; Stevenson et al., 2013). This represents a cross-disciplinary challenge for meeting international policy goals under the CBD and SDGs.

Incentive-based mechanisms, such as performance-based payments for ecosystem services (PES) in small-scale fisheries (SSFs) could alleviate threats to marine biodiversity, whilst ensuring coastal communities are no worse off (Bladon et al., 2016). PES can be efficient and effective and, when well designed, can benefit biodiversity and alleviate poverty in terrestrial ecosystems (Ferraro & Simorangkir,

2020; Ma et al., 2017). However, PES remains under-explored and under-utilised in the marine realm, with a lack of empirical data on whether PES would be accepted by coastal communities; if they can deliver marine conservation and well-being outcomes; and how they should be designed as part of an overall policy mix.

At the systemic level, gaining a better understanding of the cost-effectiveness of market-based mechanisms for marine conservation is increasingly important under growing adoption of net-outcome goals for biodiversity (including under the CBD post-2020 framework) (CBD, 2020a; Maron et al., 2021). For example, just as investors and consumers are increasingly demanding ‘carbon neutral’ and ‘deforestation-free’ terrestrial supply chains, so they could demand ‘biodiversity-neutral’ fish supply chains and coastal development projects (Booth, Arlidge, et al., 2021). This will require that entities that damage marine biodiversity offset their impacts through other positive conservation outcomes (Bull et al., 2020). Therefore, a supply of investment-ready marine conservation projects will be needed – which can demonstrate measurable, additional marine conservation outcomes for a given cost – to achieve net outcomes in the marine realm under the CBD’s post-2020 strategy (CBD, 2020a; Jacob et al., 2020). These projects could be based around performance-based payments to promote pro-conservation behaviour.

I used methods from predictive conservation and behavioural sciences (Travers et al., 2021; Travers, Selinske, et al., 2019) – adopting a novel combination of scenario interviews and contingent valuation (CV) – to understand how incentive-based mechanisms might influence fisher behaviour, and resulting conservation and well-being outcomes, in SSFs. I implemented the research in Indonesia: a global priority country for reconciling trade-offs between marine biodiversity and fisheries (Selig et al., 2014, 2018); and focused on two Critically Endangered taxa: hammerhead sharks (*Sphyrna* spp.) and wedgefish (*Rhynchobatus* spp.). In doing so, I aimed to answer the following questions:

1. Can performance-based positive incentives affect fishers’ behaviour relating to capture and retention of Critically Endangered taxa, and associated biodiversity and well-being outcomes? If so how and why?
2. How does the impact of positive incentives compare (and possibly interact) with other instruments, including negative incentives (i.e., fines) and non-monetary social rewards?
3. What magnitude of incentives are needed to change fisher behaviour to halt mortality of Critically Endangered taxa? What can be inferred about the value of these taxa to fishers, and how does this vary across taxa and contexts?
4. Which types of interventions and instrument mixes might be most cost-effective for reducing mortality of Critically Endangered marine species, whilst maintaining or improving the well-being of coastal communities?

This addresses several policy-relevant areas of conservation literature: shifting from research on threats towards research on solutions (D. R. Williams et al., 2020); responding to calls for wider use of behavioural sciences in conservation (Balmford et al., 2021); building a greater understanding of values of marine species from the perspective of resource users (Lew, 2015); providing grounded and contextually rich data for use in decision-making (Christie et al., 2020; Wyborn & Evans, 2021); and offering a novel and replicable combination of methods for doing so. The results can be used within the study sites, and the methods generalised to other similar SSFs throughout the world – to inform local conservation interventions, and global financing mechanisms that could deliver a sustainable and equitable ocean economy.

8.2 Methods

I adopted a mixed methods approach, combining scenario interviews (Cinner et al., 2009; Travers, Selinske, et al., 2019) with CV (Carson & Hanemann, 2005). This provided quantitative estimates of the cost-effectiveness of incentive-based interventions, and qualitative details on fishers' attitudes, preferences and motivations.

8.2.1 Study sites and taxa

I focused on two contrasting case study SSFs in Indonesia: Tanjung Luar in East Lombok, East Nusa Tenggara Province and Lhok Rigaih in Aceh Jaya, Aceh Province (Figure 8.1). The two sites represent contrasting case types, where Lhok Rigaih is a coastal gillnet fishery taking elasmobranchs as marketable bycatch (Simeon et al., 2020), and Tanjung Luar is a semi-commercial pelagic longline fishery taking elasmobranchs as target catch (WCS-IP, 2019; I. Yulianto et al., 2018). I focused on hammerhead sharks (*Sphyrna* spp.) and wedgefish (*Rhynchobatus* spp.), since both are Critically Endangered and CITES-listed, yet commonly caught throughout tropical SSFs, and represent contrasting case types in terms of biological traits, catchability, survivability and use values (Hau et al., 2018; Kyne et al., 2020; Rigby, Dulvy, et al., 2019; Wu, 2016).

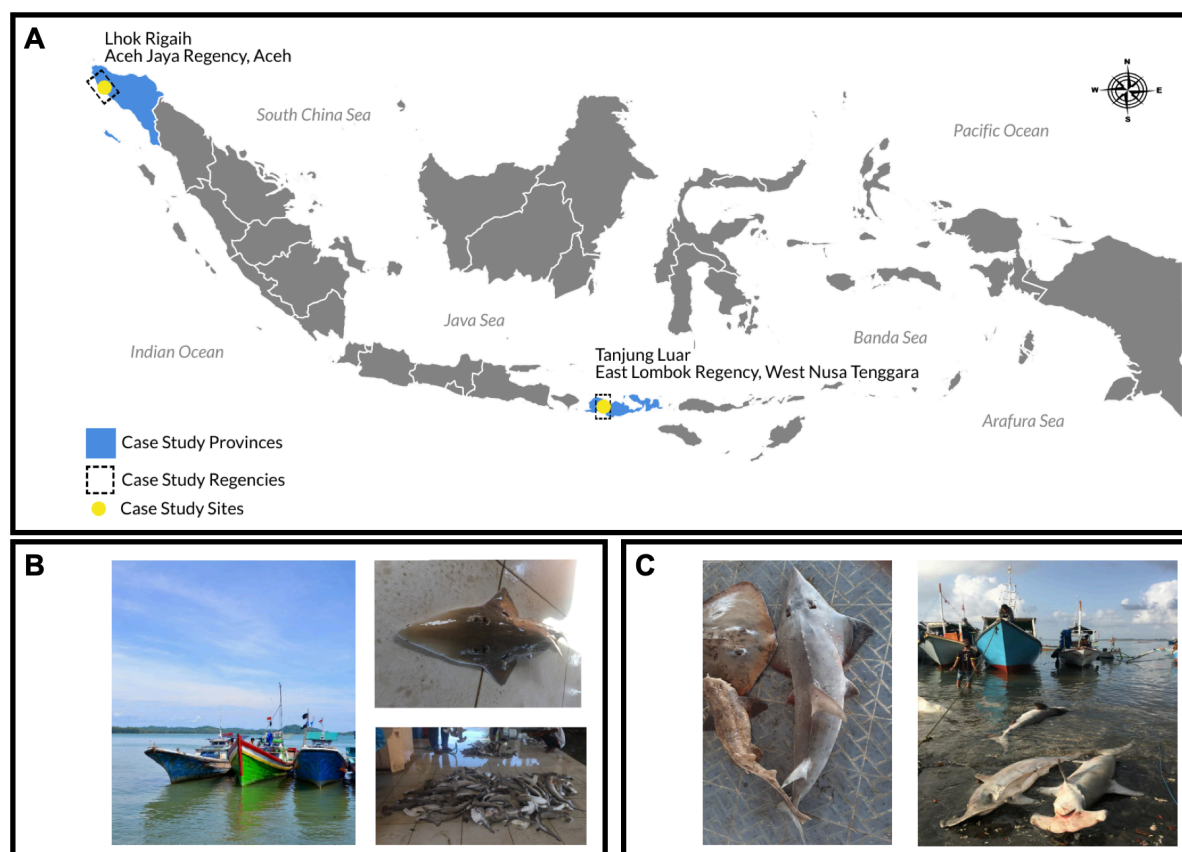


Figure 8.1 Study sites. A) Map of Indonesia, with location of study sites, B) Example study vessels (left) and landings (right) from Lhok Rigaih, C) Example study vessel (right, see larger blue vessel) and landings from Tanjung Luar

8.2.2 Study design

Scenario Interviews

Scenario interviews are commonly used in behavioural sciences and predictive conservation, and involve constructing a set of plausible futures and asking people how they would behave (Cinner et al., 2011; Travers et al., 2016; Travers, Selinske, et al., 2019). I followed a similar design to Travers et al. (2016, 2019), enabling comparison of the relative effects of different hypothetical interventions and institutional arrangements on conservation-relevant behaviour, by considering multiple scenarios within the same study.

Economic theory and empirical studies of fisher behaviour indicate that conditional monetary rewards and exogenously-imposed rules and sanctions can influence marine resource extraction (Arias et al., 2015; Booth, Mardhiah, et al., 2020; Gutiérrez et al., 2011; Wosnick et al., 2020). However, theories of collective action show that trust, norms and institutional arrangements can also shape individual behaviour in ways that differ from those predicted by rational self-interest (Ostrom, 1990, 2000). Therefore, rules and incentives can crowd-out or crowd-in pro-social behaviour, depending on their perceived legitimacy, with complex interactions between intrinsic and extrinsic incentives (Cinner et al., 2021; Gneezy et al., 2011; Grillos et al., 2019; Oyanedel, Gelcich, & Milner-Gulland, 2020).

Following this, I used scenario interviews to examine the effect of three hypothetical interventions (Table 8.1) on fisher behaviour – specifically hammerhead and wedgefish landings – relative to business as usual (BAU). The scenarios were designed following a scoping phase, to understand the socio-ecological contexts of the study sites and ensure plausibility. They included: 1) direct, positive, performance-based monetary incentives (i.e., PES), 2) direct negative monetary incentives, implemented via a regulation and associated sanction (i.e., a fine), 3) a site-specific intervention, based on an understanding of fishers' interests and priorities developed during the scoping phase: fishers in Tanjung Luar were offered an indirect monetary reward via a lottery for children's school fees; fishers in Lhok Rigaih were offered non-monetary social rewards in via community recognition (Table 8.1; Appendix 5, S8.1).

Table 8.1 Scenarios explored in the scenario interviews. A fine and PES scenario was explored in both sites, as well as a site-specific intervention in each site, designed based on a qualitative understanding of fishers' interests and motivations as per findings

Scenario	Site	
	Tanjung Luar (N = 120)	Lhok Rigaih (N = 22)
Business as Usual (BAU)	Everything continues the same as it is now, with no changes in markets or regulations.	
Positive incentive intervention (PES)	An agreement is established to provide direct performance-based compensatory payments for any vessels returning from a fishing trip with zero hammerhead sharks/wedgefish. Someone will be monitoring the port every day, and fishers will be required to provide additional proof of compliance, such as on-board video monitoring during their trip.	
Negative incentive intervention (Fine)	A law is established to fully protect hammerhead sharks/wedgefish, and any vessel found landing a hammerhead sharks/wedgefish would be fined. Someone will be monitoring the port every day, and the fine will be taken from the total trip profit.	
Site-specific intervention	Lottery: An agreement is established for an education fees lottery scheme, so that members of all vessels returning from a fishing trip with zero hammerhead sharks/wedgefish enter a monthly lottery in which the prize is school fees payments for 1 year for a school-aged child in their family. Someone will be monitoring the port every day, and fishers will be required to provide additional proof of compliance such as on-board video monitoring during their trip.	Guardians: A 'shark guardians' group is established in which participating fishers voluntarily agree to land zero hammerhead sharks/wedgefish. In return they receive specialised training and equipment, public social recognition in local newsletters and media channels, and invitations to monthly events with other shark guardians and local leaders.

Each interview included a pre-survey to collect socio-demographic information and ensure familiarity with the study taxa (Appendix 5, S8.1). All respondents were then given separate scenario interviews for each taxon, with the order randomised amongst respondents to control for question-ordering biases. For each taxon, respondents were first presented with a BAU scenario, and then the three experimental scenarios (Table 8.1), the orders of which were also randomised. I focused on understanding 1) the influence of the scenarios on landings of the taxon in question, and 2) the magnitude of the positive/negative incentives needed to induce changes in landings of that taxon (i.e., the Willingness-to-

Accept (WTA) a payment and the Willingness-to-pay (WTP) a fine). I then explored how and why fishers would change (or not change) their behaviour through semi-structured and open-ended follow-up questions. I focused in particular on perceived fairness and impacts on well-being (Appendix 5, S8.1), as both are determinants of the social legitimacy of rules, people's willingness to comply, and the socio-economic impacts of conservation (Arias et al., 2015; Keane et al., 2008; Oyanedel, Gelcich, & Milner-Gulland, 2020). Follow-up questions also allowed triangulation of answers to check for consistency.

Contingent valuation

The scenarios involving direct monetary incentives (i.e., PES and fine) were combined with a CV question. CV is a well-established stated preference method of economic valuation, which can be used to determine individuals' preferences for hypothetical policies (Carson & Hanemann, 2005), and estimating the value of public environmental goods, including endangered marine species (Lew, 2015; Vianna et al., 2018).

I used the payment card method, in which respondents selected their minimum WTA/WTP to reduce their landings to zero from a range of bid values (Appendix 5, S8.1). I designed the CV questions to reduce common biases, including: 1) pre-surveying and piloting bid ladders, to optimize efficiency and accuracy of responses while minimising cognitive burden; 2) including an adapted version of a cheap talk script, to reduce hypothetical bias/increase perceived consequentiality; 3) asking follow-up questions on zero responses, to separate true zeros from protest zeros; 4) randomising the order in which the bid values were presented (i.e., low-to-high vs high-to-low) amongst participants to control for anchoring bias (Carson & Hanemann, 2005).

8.2.3 Data collection

Data were collected during site visits from December 2019 to July 2021. Interviews were conducted primarily in Bahasa Indonesia, with local languages sometimes used for clarification purposes. Interviews were conducted with 142 fishers, including 120 from Tanjung Luar and 22 from Lhok Rigaih, representing roughly 90% of the shark-relevant fisher population in both locations.

8.2.4 Analysis

Descriptive statistics

I used descriptive statistics to summarise the structured, quantitative data. Responses and attitudes to hypothetical interventions were compared against BAU and each other, following Travers et al. (2016, 2019); and I derived mean and median WTA and WTP per site and taxon, and conducted Welch Two Sample t-tests to investigate differences in means, with Rstudio (Base-R and Ggplot2) (RStudio Team, 2020; Wickham, 2016). I also compared average stated preferences with other independent market values

to cross-validate results. I used thematic analysis (coding and grouping) for analysing unstructured qualitative data (Braun & Clarke, 2006).

Modelling

I constructed a model to analyse the effects of the different scenarios and taxa – alongside other socio-economic, demographic and study design control variables (Table 8.2) – on stated willingness to adopt pro-conservation behaviour (i.e., reduce landings of the study taxa). The allowed for validation, by testing whether the response variables correlated as expected with externally-valid constructs; and exploration, to understand which other variables might influence fisher behaviour.

I used a mixed-effects logistic regression with a binary response variable, indicating if respondents said they would (1) or would not (0) reduce landings. I used intervention and taxon as predictor variables, and explored the influence of all other meaningful control variables (Table 8.2) on how well the model fit the data, using backwards selection to find the optimal model with the lowest Akaike Information Criterion (AIC) value (Appendix 5, S8.1). To allow for meaningful comparison I excluded the site-specific scenarios, resulting in 567 observations from 189 interviewees. I used ‘interviewee’ as a random effect to account for pseudo-replication introduced through survey participants providing responses to multiple scenarios, and tested models for each scenario and taxon separately to check for consistency. Modelling was conducted in Rstudio using the `glmer` function in the `lme4` package (Bates et al., 2015). For validation purposes I also constructed linear models for explaining WTP and WTA, given willingness to change behaviour (Appendix 5, S8.1).

The PES and fine scenarios were expected have a positive association with willingness to adopt pro-conservation behaviour relative to BAU. Fishers’ catches of the study taxa and incomes were expected to correlate negatively with willingness to change and WTA, as proxies for perceived or actual use values, and therefore opportunity costs of reducing landings. Fishers in Tanjung Luar were expected to have higher WTA/WTP relative to Lhok Rigaih, based on differences in the fishery types and market access. All other variables were included as control or exploratory variables (Table 8.2).

Table 8.2 Explanatory variables included in a mixed-effects logistic regression with a binary response variable, indicating if respondents said they would (1) or would not (0) reduce landings.

Variable	Description	Expected results, based on theory and previous empirical studies	Included in the optimal model?
Scenario	Categorical variable representing the hypothetical interventions explored in the scenarios. Three levels: BAU,	(+) PES and fine scenarios expected to have higher likelihood of behaviour change relative to BAU, according to basic economic theory and since fishers are motivated by exogenous incentives (Booth, Mardhiah, et al., 2020; Gutiérrez et al., 2011; Wosnick et al., 2020).	Yes

	fine, PES, with BAU as the reference level.	WTA PES expected to be higher than WTP fine, due to loss aversion (Cinner, 2018; Kahneman & Tversky, 1979).	
Taxa	Categorical variable representing the taxa explored in the scenarios. Two levels: hammerhead or wedgefish, with hammerhead as the reference level.	(?) Wedgefish are more valuable per individual (Booth, Squires, et al., 2021; Hau et al., 2018), however hammerheads are caught in higher total volumes in both sites (Simeon et al., 2020; I. Yulianto et al., 2018).	Yes
Site	Categorical variable representing the sites in the study. Two levels: Tanjung Luar and Lhok Rigaih, with Lhok Rigaih as the reference level. Captures operational and socio-economic variation at the fishery-level.	(+) Tanjung Luar expected to have higher WTA relative to Lhok Rigaih, since shark fishing is more targeted, commercialised and of higher value than in Lhok Rigaih (Booth, Squires, et al., 2021; I. Yulianto et al., 2018).	No
Last catch	Continuous variable representing stated last catch of taxa in question.	(-) Last catch expected to correlate negatively with likelihood of behaviour change and positively with WTA, as a proxy for fishers' use values for hammerheads and wedgefish, and thus perceived opportunity costs of reducing catches to zero (Carson et al., 2001; Liebe et al., 2011).	Yes
Income	Continuous variable representing stated monthly household income.	(-) Income expected to correlate negatively with likelihood of behaviour change and positively with WTA. According to basic economic theory and empirical studies of WTP for environmental goods, income typically positively correlates with WTP (Carson et al., 2001; Liebe et al., 2011). However, use of the good is also important, and in this instance, income is indicative of direct use value of catches, and therefore a proxy for opportunity costs of reducing landings.	No
Age	Continuous variable representing fisher age in years, also a proxy for experience (co-varies with fisher experience).	(?) Included as a demographic control variable. No strong pre-existing theory or hypothesis. May serve as a proxy for generational cultural differences and potential openness to innovation.	Yes
Fisher experience	Continuous variable representing fisher experience in years (co-varies with age).	(?) Included as a demographic control variable. No strong pre-existing theory or hypothesis. May serve as a proxy for fisher knowledge/skill, and openness to innovation.	No
Education	Binary variable representing education level in terms of whether or not the fisher completed high-school.	(?) Included as a demographic control variable. No strong pre-existing theory or hypothesis. May serve as a proxy for openness to innovation.	No
Influence score	Numerical dummy variable representing fishers' perceived influence over fishing decisions. Three levels: 1 = no influence, 2 = some influence, 3 = ultimate influence, with no influence as reference level (co-varies with vessel position).	(?) Included as a control variable for vessel-level decision-making dynamics. No strong pre-existing theory or hypothesis. May serve as a proxy for trust in other people's cooperation under theories of public goods and collective action.	No

Vessel position	Categorical variable representing fishers' position in their vessel. Two levels: captain and crew, with captain as reference level. (co-varies with influence score)	(?) Included as a control variable for vessel-level decision-making dynamics. No strong pre-existing theory or hypothesis. May influence perceived or actual use values/opportunity costs, since vessel captains typically receive higher incomes/greater shares of trip profits.	No
Order	Numerical dummy variable representing the order in which the scenarios were presented.	(?) Included to control for study design effects. Would expect a significant co-efficient if there was a significant question-ordering effect.	No
Interviewee	Unique identifier for each interviewee	Included as random effect to control for pseudo-replication caused by multiple treatments for each interviewee.	Yes

Key: BAU = business as usual; PES = payment for ecosystem service, WTA = willingness to accept, WTP = willingness to pay, + = expected positive association, - = expected negative association, ? = unclear direction of association.

Valuation

I used average WTAs and WTPs, and existing data on average catches and trip numbers, to estimate a) the economic value of wedgefish and hammerheads according to fishers; and b) the potential annual costs of PES interventions in the study sites. For Tanjung Luar, these estimates were based on an average of ~300 fishing trips per year, which land 1.7 hammerhead sharks per trip; and ~140 trip per year which land 0.97 wedgefish per trip (WCS-IP, 2019). For Lhok Rigaih, there are ~1,560 shark-relevant trips per year, which land an average of 11.5 hammerheads and 1.4 wedgefish per trip (Simeon et al., 2020).

8.3 Results

8.3.1 The estimated effectiveness of incentives, and their magnitude

Summary statistics

The majority of Tanjung Luar fishers stated that catches of hammerheads and wedgefish would increase in a BAU scenario (86% of respondents and 92% of respondents, respectively; Figure 8.2A), because they would like to increase their fishing effort and income. In Lhok Rigaih, most fishers stated that catches of hammerheads and wedgefish would remain stable under BAU, though some reported they would increase fishing effort for more income (27% and 23%, respectively; Figure 8.2A).

For the hypothetical interventions, the PES scheme had the largest influence on stated willingness to reduce landings (Figure 8.2A). In Tanjung Luar, 98% and 92% of respondents reported that they would reduce landings of hammerheads and wedgefish to zero (respectively), with median CV bids of IDR 5 million (US\$ 357) and IDR 8 million (US\$ 571) per trip, respectively (Table 8.3, Figure 8.3A). In Lhok Rigaih, 100% of respondents reported that they would reduce landings for both taxa. Median CV bids were IDR 300,000 (US\$ 21) and IDR 150,000 (US\$ 10) per trip for hammerheads and wedgefish, respectively (Table 8.3, Figure 8.3A), though four of 22 fishers stated the payment for wedgefish should

vary depending on the size of the individual. In both sites and for both taxa the PES intervention was widely perceived as fair, and to have positive or neutral impacts on material well-being (Figure 8.2B, C).

In contrast, 54% and 64% of fishers in Tanjung Luar stated that they would reduce their landings of hammerheads and wedgefish (respectively) in response to a fine (Figure 8.2A). Willingness was lower in Lhok Rigaih at 18% and 27%, respectively (Figure 8.2A). Follow-up questions indicated that most fishers rejected this policy: it was perceived as unfair and an infringement on their rights, with negative impacts on well-being (Figure 8.2B, C). Many fishers stated they would not comply, by hiding their catch or landing it elsewhere. Correspondingly, many respondents gave protest zeros in the CV question (Figure 8.3B). In Tanjung Luar, of those that did state a WTP, the average value was lower than in the WTA scenario: the median bids were IDR 2 million (US\$ 143) per trip for hammerheads, and IDR 1.5 million for wedgefish (US\$ 107). In Lhok Rigaih, the median bids were IDR 45,000 (US\$ 3) per trip for hammerheads and IDR 80,000 (US\$ 6) for wedgefish (Figure 8.3B).

The site-specific interventions gave mixed results. For Tanjung Luar, the school fees lottery scenario was the least effective of the three hypothetical interventions: 50% of fishers stated that they would reduce landings, and many felt the scheme unfair (Figure 8.3A). For Lhok Rigaih, the guardians scenario was more effective than the fine but less effective than PES. Most fishers (55%) would voluntarily decrease wedgefish landings, while only 27% agreed for hammerheads (Figure 8.3A).

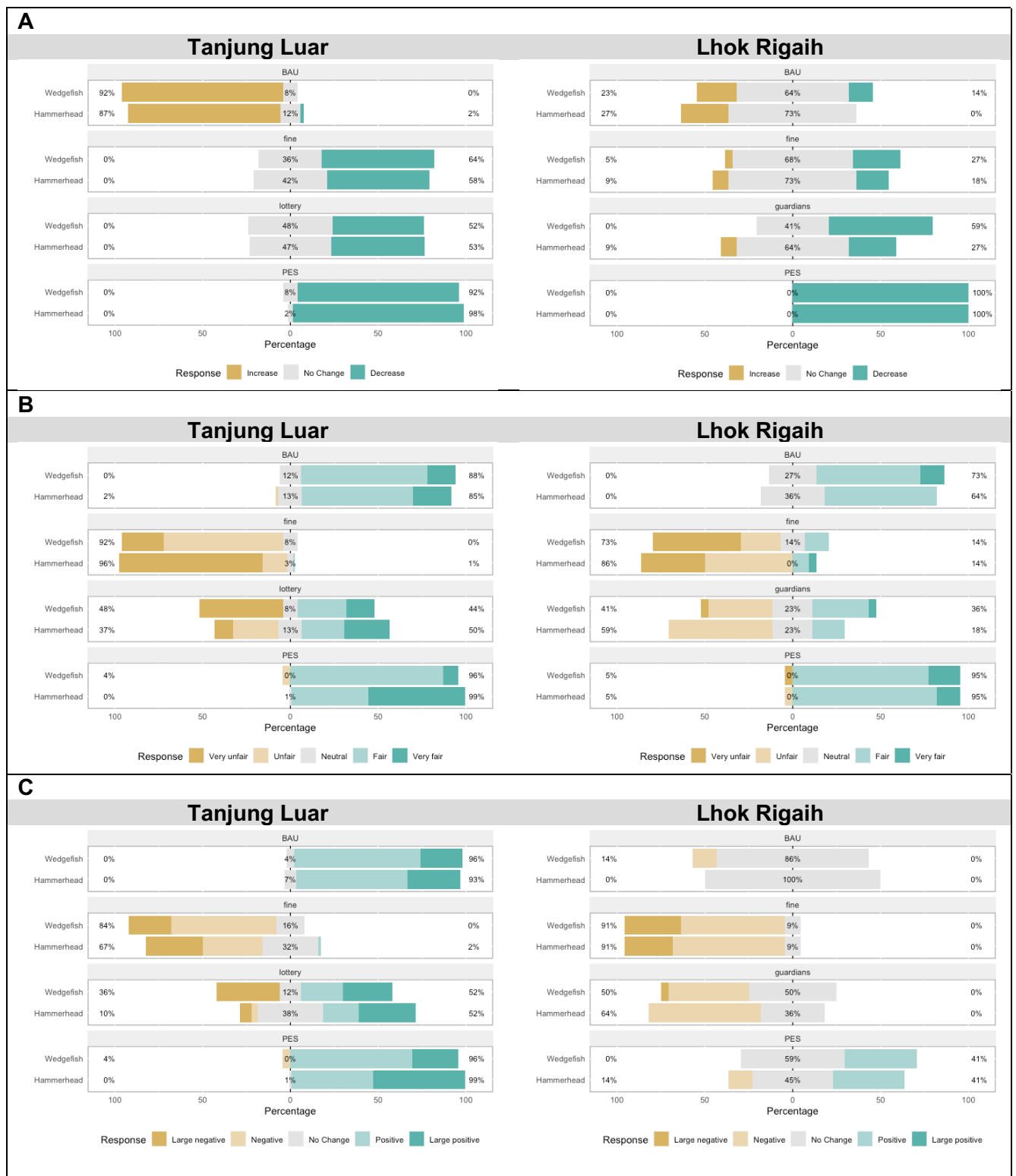


Figure 8.2 Fisher perceptions of different hypothetical interventions for managing capture of hammerhead sharks and wedgefish. Tanjung Luar (left) and Lhok Rigaih (right). A) reported changes in landings under different scenarios, B) perceived fairness of different scenarios and C) perceived impact on household income under different scenarios. Key: BAU = business as usual; PES = payment for ecosystem service.

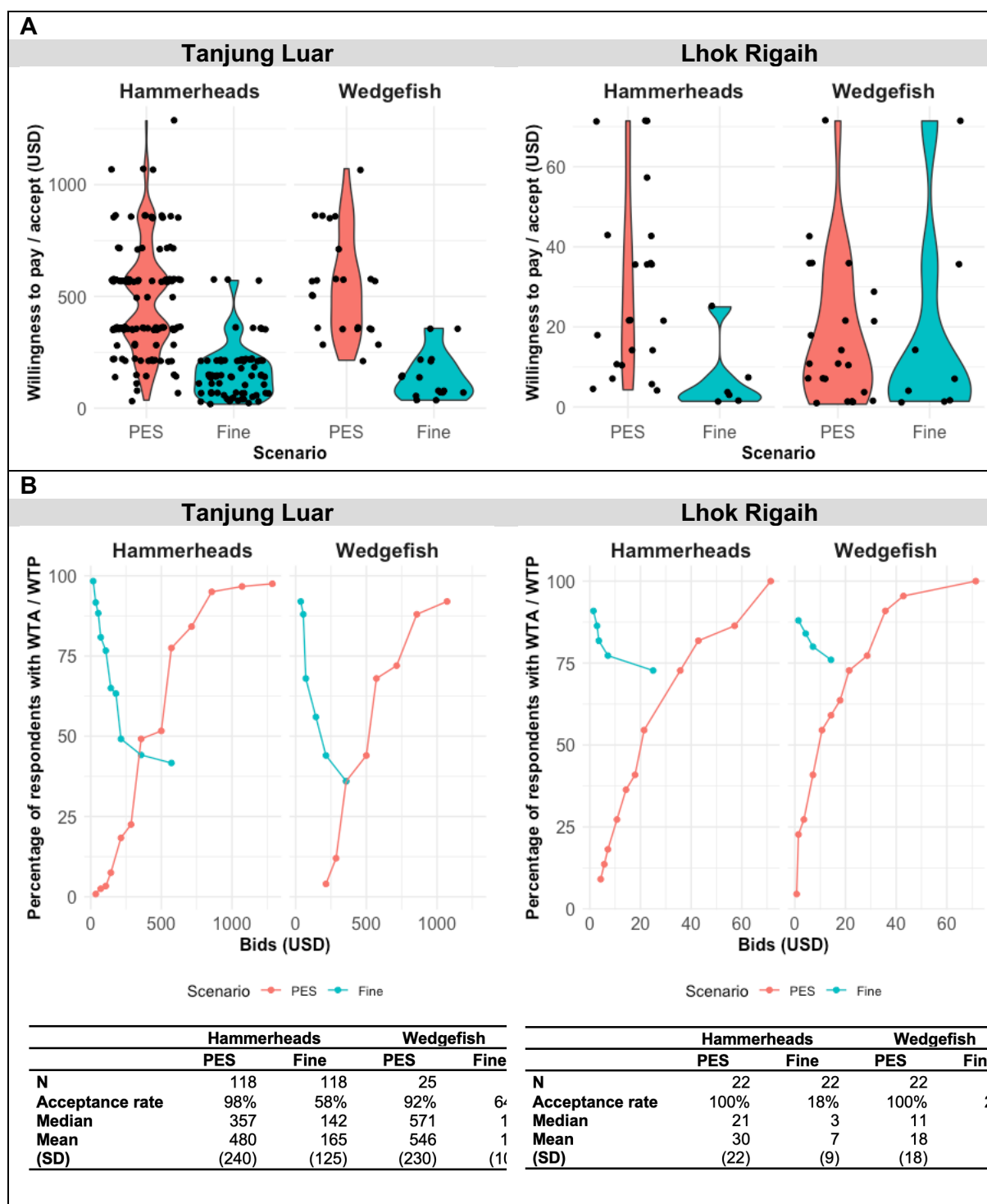


Figure 8.3 Summary of fishers' willingness to pay (WTP) fines / accept (WTA) incentives for reducing catches of Critically Endangered taxa per trip. Tanjung Luar (left) and Lhok Rigaih (right). A) mean and spread of minimum WTA/maximum WTP; B) demand curve. All values converted in to US\$ at approximate exchange rate of IDR 14,000 = USD 1.00. Medians presented in summary statistics are based on lower bounds of median bid range, mean is based on average of lower bounds of selected bid. Protest zeros are the inverse of the acceptance rate. PES = payment for ecosystem.

Predictors of willingness to change behaviour

The optimal model for willingness to reduce landings included scenario, taxa, age and last catch as fixed-effects, and interviewee as a random effect (Table 8.3; Appendix 5, S8.2).

Model outputs indicated that PES and fine interventions were associated with a statistically significant increase in the likelihood of reducing landings ($p < 0.001$) compared with BAU, though the effect size of PES was greater (Table 8.3), which matched expectations. Taxon was not a significant predictor of willingness to reduce landings in principle, though willingness was lower for wedgefish than hammerheads (Table 8.3). Last catch was significantly negatively correlated with likelihood of reducing landings ($p < 0.01$) (Table 8.3). This result is as expected, with catches representing perceived opportunity costs of reducing landings. Fisher age also had a significant negative relationship with likelihood of reducing landings ($p < 0.01$) (Table 8.3) and may serve as a proxy for generational differences in openness to innovation. These effects remained consistent when models were constructed with sub-sets of the data (Appendix 5, S8.2).

Table 8.3 Model outputs for influence of scenarios, taxa and socio-demographic control variables on stated willingness to change behaviour (i.e., likelihood of reducing landings of wedgefish and hammerheads to zero). Logit model coefficients converted from log odds to probabilities with 95% confidence intervals using as per probability = $\exp(\text{coeff}) / 1 + \exp(\text{coeff})$.

Predictor	Model coefficient	Probability estimate	95% Confidence interval	p-value	Sig
(Intercept)	-2.450	0.079	0.013-0.354	0.009	**
Scenario – PES (vs BAU)	9.245	0.999	0.999-1.000	2.45e ⁻¹⁶	***
Scenario – Fine (vs BAU)	4.563	0.989	0.965-0.997	7.59e ⁻¹³	***
Taxa – Wedgefish (vs hammerhead)	-0.107	0.473	0.287-0.667	0.795	
Last catch	-0.092	0.477	0.460-0.495	0.010	*
Age (in years)	-0.046	0.489	0.477-0.500	0.046	*

Sig. Codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 0.5

There were significant differences between Tanjung Luar and Lhok Rigaih for both mean WTA payment and mean WTP fine (Welch Two Sample t-tests, $p < 0.001$ for both WTA and WTP). Both were significantly higher in Tanjung Luar, which aligns with expectations given the differences in market access and use values between the two sites. Differences between each taxon were only significant for WTP in Lhok Rigaih, where hammerhead sharks had a higher absolute value than wedgefish ($p < 0.01$), likely reflecting their higher and more frequent relative catch rates. Validation models of WTA and WTP also generally aligned with expectations (Appendix 5, S8.3).

8.3.2 Mechanisms for behaviour change: why and how

Reasons to change

In Lhok Rigaih 95% of respondents mentioned economic reasons when explaining why they would change their fishing behaviour under the PES scenarios (Table 8.4). However, this was often only part of the answer, other reasons included: intrinsic motivations to protect the ocean or marine animals; inherent desire to comply with rules; and social- or community-related motivations (Table 8.4). Those who would change their behaviour in response to the fine also reported economic, intrinsic and social reasons (e.g., *“I am scared of being fined”*, *“If others agree, I will obey”*, *“We should protect the ocean”*). In the ‘guardians’ scenario, respondents who would decrease their landings stated intrinsic desires to protect the ocean, and/or social motivations to follow their peers (e.g., *“I care about protecting the sea”*, *“there will be more wedgefish”*, *“togetherness to protect the sea”*).

In Tanjung Luar, economic/material reasons were the main stated reasons for changing behaviour (Table 8.4). In contrast to Lhok Rigaih, no fishers mentioned intrinsic or social motivations, though many were motivated by providing resources for their immediate family. Several fishers also stated that PES would provide more stable or secure income in comparison to uncertain fishing outcomes (Table 8.4). In the lottery scenario, respondents who reported they would decrease their landings stated the importance of education for their family (e.g., *“[my children] should not follow in the footsteps of their parents to be a fisherman, they have to go to school”*).

Reasons not to change

Those who reported that they would not decrease their landings in the fine scenario gave economic, practical and fairness-related reasons, due to the perceived economic burden on their household and community (e.g., *“it will impact the livelihoods, and be a burden on the community”*, *“there will be a loss, the fortune will be reduced”*); or the impracticality of avoiding wedgefish and hammerheads, making it unfair if they are punished (e.g., *“I cannot avoid it... it’s not the target but it will reduce fishers’ incomes”*).

In the guardians scenario in Lhok Rigaih, some fishers reported that it would be wasteful and disrespectful to God to release catches, especially if they are already dead (e.g., *“avoiding it is difficult, releasing it is throwing away a gift from God”*, *“if it’s dead I will bring it home, because it wastes fortune in vain”*). The lottery scenario in Tanjung Luar was particularly unpopular: it was perceived as unfair because only a few people would benefit, and it would not bring a large enough financial reward to compensate for income lost.

8.3.3 How to change (or not)

Of those who confirmed they would reduce landings, most reported post-capture release as the method (85% for hammerheads, 96% for wedgefish in Tanjung Luar; and 100% of respondents for both taxa in

Lhok Rigaih; Appendix 5, S8.4). However, 50% of fishers in Lhok Rigaih stated they would only release sharks that are alive. Similarly, fishers in Tanjung Luar stated that they would use dead sharks as bait. Post-capture release could be effective for wedgefish, with 91% of all respondents (across both sites) reporting that they are at least sometimes alive and 72% reporting that they are usually alive when brought on to the boat. However, this may be less effective for hammerheads, with 96% of respondents from Tanjung Luar reporting that they are sometimes alive and sometimes dead; and 86% of respondents from Lhok Rigaih reporting that they are usually or always dead (Appendix 5, S8.4).

Other strategies to reduce catches were also mentioned, including changing fishing grounds to avoid hammerheads in Tanjung Luar (10% of fishers) (Appendix 5, S8.4). Similarly, during semi-structured discussions, Lhok Rigaih fishers indicated there is a spatio-temporal element to hammerhead catches, with a higher catch per unit effort during June to September, and in sandy/muddy areas near the shore. However, avoidance or minimisation strategies were not mentioned as a response to the scenarios – only post-capture release (Appendix 5, S8.4).

Those who stated they would not reduce landings in the fine scenario gave a range of different strategies, including pay the fine and then fish more to make up for the loss; find ways to cheat, such as hiding the shark or landing it elsewhere to sell; or cut it up and use it as bait.

Table 8.4 The five main types of reasons fishers gave for reducing their landings to zero under the PES scenario. Based on post-hoc coding and grouping, with the percentage of respondents that mentioned them for each site and taxon (HH = hammerheads, WF = wedgefish), and selected illustrative quotes. The reasons are not mutually exclusive, with many fishers giving more than one reason.

Reason category	Site					
	Tanjung Luar			Lhok Rigaih		
	HH	WF	Illustrative quotes	HH	WF	Illustrative quotes
Economic	94%	92%	<i>“The loss of money is covered”</i> <i>“That money can cover the operating costs”</i> <i>“the price of sharks has been replaced, so it is enough to meet the daily needs”</i>	95%	95%	<i>“There is compensation so no disadvantage”</i> <i>“There is no loss if there is a reward”, “the compensation can cover the operating costs”</i>
Intrinsic	0%	0%	-	14%	32%	<i>“I can protect the sea”</i> <i>“Sustainability for future generations”</i> <i>“The price is cheap, their breeding is slow, [we] can’t take them every day”</i>
Social	0%	0%	-	9%	9%	<i>“If it has been agreed to be protected by the community, I will follow my friends and the agreement”</i> <i>“Togetherness to protect the sea”</i>

Security	11%	8%	<i>“It’s not certain if I will get hammerheads... but even if the catch is less I will definitely get the money”</i>	0%	0%	-
Obedience	1%	0%	<i>“Because there is a commitment to the agreement”</i>	9%	14%	<i>“Rules are there to be obeyed” “I will obey the applicable rules”</i>

8.3.4 Estimated values

Based on average WTA and WTP estimates per trip (Figure 3, Table 8.5), and available data on catches per trip, it is estimated that fishers in Tanjung Luar attribute economic values of \$83 - \$282 per individual to hammerheads and \$110 - 588 per individual to wedgefish. For fishers in Lhok Rigaih, the estimated values are lower: \$0.3 - 2.60 per hammerhead and \$5.70 - 12.90 per wedgefish (Table 8.5).

Based on available data on trips per year, it would cost \$42,600 - 144,000 to implement a PES scheme to incentivise fishers to stop landing hammerheads in Tanjung Luar, and \$4,680 - 46,800 in Lhok Rigaih (Table 8.5). This could save an estimated 500 adult or sub-adult sharks per year in Tanjung Luar and up to 18,000 juveniles per year in Lhok Rigaih. To implement the same for wedgefish would cost \$14,980 - 79,940, in Tanjung Luar (saving around 140 adult individuals) and \$9,360 - 28,080 in Lhok Rigaih (saving over 2,000 sub-adult individuals) (Table 8.5).

Table 8.5 Value estimates per individual shark (USD), based on fishers’ stated willingness to accept (WTA) and willingness to pay (WTP). Value per individual is calculated based on average catches of 1.7 hammerheads and 0.97 wedgefish per trip in Tanjung Luar, and 11.5 hammerheads and 1.4 wedgefish per trip in Lhok Rigaih. Value per year based on average annual shark-relevant trips of 300 for hammerheads and 140 for wedgefish in Tanjung Luar, and 1,560 for both taxa in Lhok Rigaih

Site	Taxon	Scenario	Est. value per trip		Est. value per year		Est. value per individual shark	
			Median	Mean	Median	Mean	Median	Mean
Tanjung Luar	Hammerhead	PES	357	480	107,100	144,000	210	282
		Fine	142	165	42,600	49,500	83	97
	Wedgefish	PES	571	546	79,940	76,440	588	562
		Fine	107	141	14,980	19,740	110	145
Lhok Rigaih	Hammerhead	PES	21	30	32,760	46,800	1.80	2.60
		Fine	3	7	4,680	10,920	0.30	0.60
	Wedgefish	PES	11	18	17,160	28,080	7.64	12.90
		Fine	6	7	9,360	10,920	4.01	12.20

8.4 Discussion

There is growing discourse regarding promoting equity and social justice in and through marine conservation (N. J. Bennett et al., 2019, 2021), and calls for applying PES in SSFs (Bladon et al., 2016).

However, there remain few studies that practically and empirically demonstrate how this can be achieved. This study – which polls the opinions of people who will be most affected by marine conservation – provides grounded empirical evidence that incentive-based mechanisms could deliver biodiversity and well-being outcomes in SSFs. Performance-based incentives are especially appealing when compared with BAU and more traditional instruments such as direct regulation, and provide a potential socially-progressive mechanism for redistributing the global benefits of marine conservation to those who would suffer the costs.

Moreover, fisher's local knowledge on mitigation strategies generally aligned with other independent catch and survivability studies (Gallagher et al., 2014; Simeon et al., 2020; Wosnick et al., 2020; I. Yulianto et al., 2018); the model results generally aligned with theory and expectations; and the estimated values from the CV are directionally consistent with other independent studies (Booth, Squires, et al., 2021; Hau et al., 2018; Wu, 2016). This helps to validate the methods, which could be applied to pro-actively design and cost other incentive-based conservation interventions based on well-grounded context-specific data, and provide the basis for bycatch-neutral seafood supply chains under net outcome policies (Booth, Arlidge, et al., 2021; CBD, 2020a; Jacob et al., 2020).

8.4.1 Implications and opportunities for designing conservation incentives

The results – particularly the differences between Lhok Rigaih and Tanjung Luar – also provide insights into incentive design, which build on lessons from terrestrial PES schemes and broader literature on the use of extrinsic incentives to promote pro-social behaviour. To be effective, monetary incentives must be based on an understanding of local context, markets and the socio-psychological drivers of the behaviour of interest, and implemented as part of a policy mix (Börner et al., 2017; Engel et al., 2008; Gneezy et al., 2011). For example, results from Lhok Rigaih indicate that intrinsic and social motivations are a more promising entry point for conservation in this site than externally-imposed rules and sanctions. Combining social, cultural and norms-based approaches with PES could be a locally-appropriate and cost-effective intervention mix in Lhok Rigaih, with economic motivations crowding-in intrinsic motivations for collaborative conservation (Grillos et al., 2019). In contrast, the relative ineffectiveness of the lottery scheme, significantly higher WTAs, and no stated intrinsic reasons for pro-conservation behaviour in Tanjung Luar suggests strong economic motivations. In such cases, where economic motivations for exploitation are particularly strong, incentives will need to be adjusted to prevailing market conditions.

The large proportion of negative responses to the fine also highlights the limitations of direct regulation and enforcement. In this case, extrinsic negative incentives may crowd-out pro-conservation norms, while positive incentives may crowd them in (Cinner et al., 2021; Gneezy et al., 2011; Grillos et al., 2019). This result also corroborates previous studies on non-compliance with fishing regulations that are perceived as

unfair and illegitimate (Oyanedel, Gelcich, & Milner-Gulland, 2020). In some cases, civil resistance has led to regulations being rolled back (e.g., Semedi & Schneider, 2021), or committed enforcement efforts are required (e.g., Chapter 4; Booth et al., 2020), which may be more costly to implement than a payment scheme.

This study also highlights the value of using behavioural sciences and predictive methods for pro-actively exploring intervention options before implementation (Balmford et al., 2021; Travers, Selinske, et al., 2019). Scenarios were a useful method for encouraging fishers to think creatively – even a hypothetical incentive revealed heterogeneity in fisher knowledge and performance, which was not otherwise mentioned in direct questioning. Wider application of predictive methods could support robust project design in the future (Travers, Selinske, et al., 2019), and these methods could easily be adapted and scaled across other SSF to fill gaps in well-grounded and contextually-rich data for use in decision-making (Christie et al., 2020). This may be particularly important as growing adoption of net-outcome approaches to marine biodiversity (e.g., under the CBD post-2020 framework and seafood sustainability strategies) lead to the development of new markets and demand for measurable, additional marine conservation outcomes (Chapter 10; Booth, Arlidge, et al., 2021; CBD, 2020a; Jacob et al., 2020). Studies such as this could providing the basis for locally-appropriate investment ready schemes for bycatch-neutral seafood supply chains. However, while the approach leverages the contextual experience of fishers, I acknowledge that responses may not have accounted for complex and unexpected feedbacks that might occur in each intervention. Exploring the extent to which residents' expectations align with real-world outcomes would help evaluate the predictive potential of scenario-based methods such as ours.

8.4.2 Implementation needs

Despite the potential opportunities, challenges remain for implementation of marine PES. Some of these are common to all PES schemes, such as additionality and leakage (Engel et al., 2008; Lim et al., 2017); whereas others are particularly unique or challenging in the marine realm, such as monitoring and compliance management, and mismatches between behaviour and conservation outcomes (i.e., reducing landings does not always translate into mortality reduction, e.g., due to post-release mortality). Perverse incentives also represent a challenge, e.g., where fishers may be incentivised to increase catches or maintain unsustainable practices because they are compensated.

Moving forwards, monitoring and compliance can be supported through improvements in technology (e.g., on-vessel video monitoring, machine learning, and forensic hold monitoring using eDNA (Bartholomew et al., 2018; Mangi et al., 2015)), and establishing institutions which promote peer monitoring or place the burden of proof on fishers (e.g., payments conditional on video footage of live release; (Kotchen & Segerson, 2019, 2020; Muradian, 2013)). To prevent perverse incentives, payments should not exceed market values (Walker & Townsend, 2008), and could be implemented as transitional

mechanisms while fishers adopt more sustainable practices in the long-term (e.g., adopting new gears with lower bycatch ratios). Strong institutions will also be required to ensure agreements are fair, and developed with free, prior and informed consent to avoid abuses of power imbalances between ES providers and financiers, and elite capture at the community level (Pascual et al., 2014).

Sources of sustainable financing will also be crucial for scaling and permanence. Traditional marine conservation funding sources, such as aid funding and philanthropy, could be channelled directly into PES schemes, and may create more cost-effective conservation outcomes than funding indirect activities via NGOs. Novel funding sources – such as marine biodiversity offsets, tourism levies and crowdfunding – could also generate billions of dollars to finance PES (Chapter 9; Chapter 10; Booth, Arlidge, et al., 2021; Gallo-Cajiao et al., 2018; Jacob et al., 2020; Sykes et al., 2018). PES schemes could be particularly attractive for these emerging sources, which often require measurable conservation outcomes for a unit cost. As such markets emerge, the relative risk of different PES investments also warrants consideration. For example, our results indicate that many more individual sharks could be saved per dollar in Lhok Rigaih than in Tanjung Luar, which could make this a more attractive PES project. However, most sharks caught in Lhok Rigaih are juveniles, which represents a more biologically risky conservation investment. One option could be to factor in the likely contribution of different life history stages to population growth, and pro-rata values accordingly.

8.4.3 Concluding remarks

There are growing calls for transformative change towards a sustainable and equitable ocean economy (N. J. Bennett et al., 2019; Díaz et al., 2019), yet few robust examples of what this means in practice.

Performance-based incentives remain a promising yet under-tested conservation intervention in the marine realm, which is a missed opportunity for delivering cross-disciplinary policy goals under the CBD post-2020 framework, the SDGs and the Blue Economy (Bladon et al., 2016; Booth, Arlidge, et al., 2021). This study suggests that, if well designed, PES in SSFs could be cost-effective, and perceived as fair and socially-just from the perspective of target fishers. Based on these findings, I urge NGOs, funders, researchers, policymakers, and the private sector to forge partnerships with coastal communities, and create an enabling environment for exploring, trialling and experimentally testing marine PES projects together with small-scale fishers.

9. Tourism levies: operationalising the beneficiary-pays principle for just and equitable marine conservation

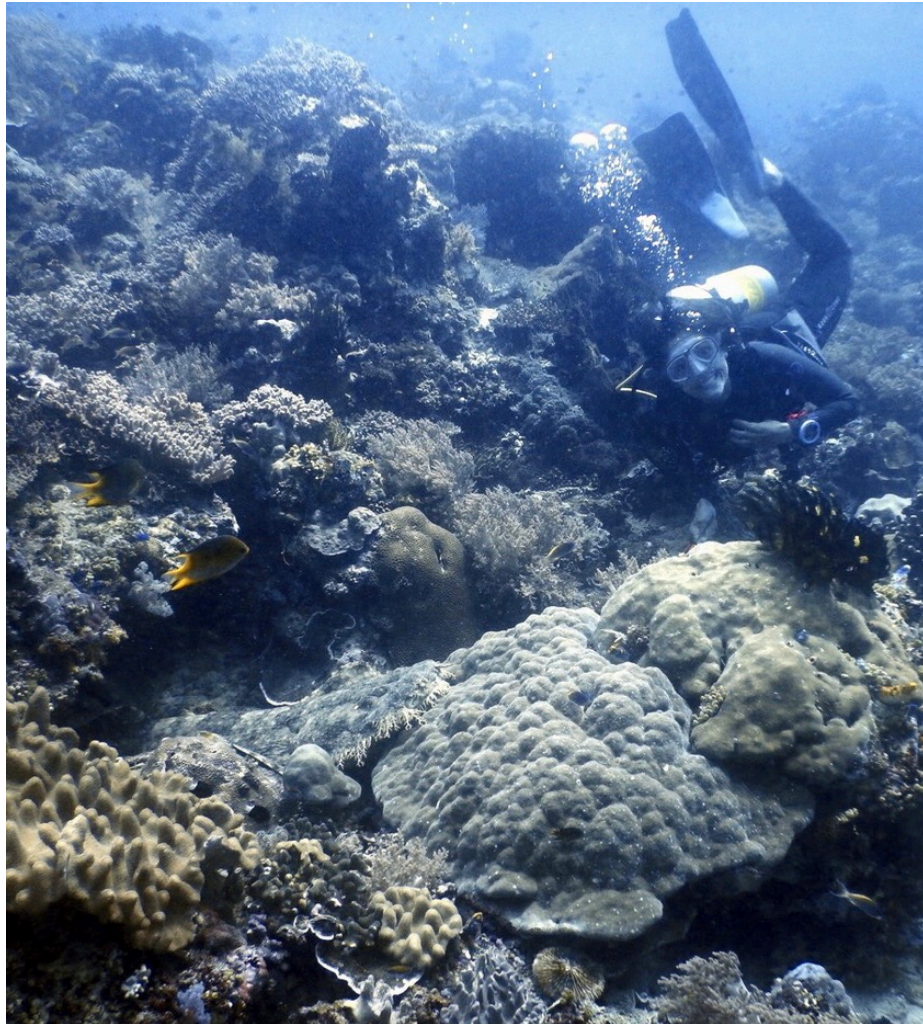


Photo: A tourist (me) with a wobbegong shark (the best shark) in Raja Ampat, Indonesia.

“Worth more alive”...but to whom?

9.1 Introduction

Human actions are driving large-scale degradation of biodiversity and ecosystems (Díaz et al., 2019). Transformative change is needed to ‘bend the curve’ on biodiversity loss within the next decade (Díaz et al., 2019; Mace et al., 2018). Importantly, this change must address the root socio-economic drivers of biodiversity loss, including issues of inequality and social injustice which can exacerbate and be exacerbated by environmental degradation (Mikkelsen et al., 2007; Mirza et al., 2020).

Of the species and ecosystems that are threatened by human actions, large, long-lived marine species (‘marine megafauna’) – such as sharks, rays, turtles, and cetaceans – constitute some of the world’s most threatened species groups (Dulvy et al., 2021; McClenachan et al., 2012). For example, it is estimated that at least 1 in 3 shark species (Class Chondrichthyes) is threatened with extinction, with population declines driven by overfishing, and exacerbated by poor governance and market forces (Dulvy et al., 2021; MacNeil et al., 2020; Pacoureau et al., 2021). In general, overexploitation is the single biggest threat to marine megafauna (Lewison et al., 2004; McClenachan et al., 2012), with people obtaining value from these taxa through consumptive use for economic and subsistence purposes.

Somewhat paradoxically, marine megafauna also have widespread public appeal, with significant socio-economic value attributed to non-consumptive uses, such as nature-based tourism (Gallagher & Hammerschlag, 2011; Troëng & Drews, 2004). On this basis it is often argued that marine megafauna is ‘worth more alive than dead’, due to the high economic value of marine tourism (e.g. Heinrichs et al., 2011). However, such arguments often fail to consider distributional issues, in terms of where and for whom these benefits accrue. In practice, there is often a mismatch: those who benefit from non-consumptive use of marine megafauna (and therefore benefit from conservation) rarely bear the costs of conservation, and vice versa; those who benefit from consumptive use of marine megafauna (and therefore may be negatively impacted by conservation, in terms of restricted access to natural resources) rarely receive the benefits of conservation (Chapter 7; Balmford & Whitten, 2003; Booth, Squires, et al., 2021; Mustika et al., 2020). This is particularly challenging in small-scale fisheries (SSFs) in biodiversity-rich ocean-dependent nations, where vulnerable coastal communities can face significant opportunity costs as a result of marine conservation efforts (Chapter 7; Booth, Squires, et al., 2021; Jaiteh et al., 2016; Selig et al., 2018; Stevenson et al., 2013).

To address marine biodiversity loss, and move towards a sustainable and equitable blue economy, there is a need for socio-economic and behavioural approaches to marine conservation, which can: 1) incentivise pro-conservation behaviour in fisheries, while 2) supporting social equity, and ensuring that conservation interventions respect the rights of indigenous people and local communities and improve wellbeing (Balmford et al., 2021; N. J. Bennett et al., 2021; Travers et al., 2021). One potential mechanism for achieving these goals is a beneficiary-pays approach to marine conservation, in which tourists or tour

operators provide performance-based payments for conservation outcomes to coastal communities (i.e. payments for ecosystem services, PES) (Balmford & Whitten, 2003; Sommerville et al., 2009). This involves collecting marine tourism levies from tourists or the operators, which are invested into community-based conservation programmes (such as support for habitat protection or bycatch reduction), with conditionality based on either actions performed or outcomes achieved (Engel et al., 2008; Sommerville et al., 2009; Sykes et al., 2018). Despite their theoretical potential, few such mechanisms – which directly link marine tourism payments to measurable community-based conservation actions or outcomes – exist in practice (though there are several examples of marine conservation agreements which, for example, use tourism profits to buy out fishing rights in and around diving locations (Sykes et al., 2018)). One challenge is that the economic values of only a small proportion of endangered marine species have been estimated, and most willingness-to-pay (WTP) studies have been conducted on a site-specific basis, and in the global north (primarily the United States) (Lew, 2015). This hinders policy innovation and payment design, particularly since mechanisms to reduce threats to marine biodiversity and improve environmental justice are most needed in global south countries (Balmford & Whitten, 2003; Selig et al., 2014, 2018). Additionally, most WTP studies do not go on to directly link estimated revenues from tourism levies to the potential for feasible, measurable conservation outcomes from real-world investments in fisheries.

This study aims to fill these gaps, by exploring how a beneficiary-pays approach for redistributing the costs and benefits of marine megafauna conservation could be operationalised. I conducted an online global contingent valuation (CV) survey to estimate international tourists' WTP towards community-based conservation for endangered marine species, with a focus on elasmobranchs (i.e., sharks and rays, hereafter 'sharks'), and used regression models to validate the findings. This fills a policy-relevant research gap, by providing a global, generic (i.e., non-site-specific) estimate of international tourists' WTP towards community-based marine conservation for endangered elasmobranchs caught in small-scale fisheries (SSFs). I then used these results to assess the feasibility of implementing local-level beneficiary-pays financing mechanisms for marine conservation in two case study sites in Indonesia. In these sites, real marine tourism markets are in close proximity to SSFs in which Critically Endangered elasmobranchs are regularly captured, and where performance-based payments could support biodiversity and wellbeing outcomes (Chapter 8). As such, this study demonstrates a novel application of economic valuation to inform conservation practice – in terms of designing feasible, sustainable and equitable marine conservation financing; and offering a scalable open-access dataset, method and instrument (available via the Harvard Dataverse (Booth, 2021a)), which can be applied to other locations where mismatches between the costs and benefits of marine conservation need to be reconciled. These economic values can also be used to inform estimates of the values of marine biodiversity to society, and thus inform wider policy-relevant analyses on trade-offs between multiple uses of marine resources (Lew, 2015; Sanchirico et al., 2013); and estimating compensation for damage to marine biodiversity caused during commercial

activities (e.g. bycatch levies or marine offsets (Booth, Arlidge, et al., 2021; Jacob et al., 2020), which is increasingly important in the context of growing adoption of net outcome goals in government and private sector biodiversity commitments (Bull et al., 2020; CBD, 2020b; Jacob et al., 2020).

9.2 Methods

9.2.1 Estimating international marine tourists' willingness-to-pay towards marine conservation

I adopted a stated preference approach – specifically, contingent valuation (CV) – to estimate marine tourists WTP for shark conservation outcomes.

Data collection

I used an online survey to gather data on international tourists' travel behaviours, basic demographic variables, pre-existing pro-environmental behaviours, attitudes towards endangered sharks, and willingness-to-pay (WTP) for conservation of endangered sharks (Appendix 6, S9.1). The target population for the survey was people with an interest in, or prior experience of, marine-focussed international tourism. I designed the survey in Jisc and distributed it to respondents via Prolific, in which I filtered survey distribution to target people with an interest in travel. No other a-priori sampling decisions were applied, due a lack of reliable data on the demographics of international marine tourists (e.g. Kieran, 2019), and since these demographics vary from country to country. I acknowledge that web surveys can introduce coverage and selection biases relative to the general population, however this is acceptable for the purposes of an exploratory study of a sub-group which is not a priori characterised by particular demographic variables (Bethlehem, 2010; Lehdonvirta et al., 2021; Wardropper et al., 2021).

Contingent valuation

I used CV to measure respondents' WTP for community-based shark conservation. CV is a well-established stated preference method to determine individuals' preferences for the provision of non-market environmental goods or services, or for hypothetical public policies (Carson & Hanemann, 2005; Hoyos & Mariel, 2010). It has been widely used for estimating the economic value of biodiversity and ecosystem services, especially in nature-based tourism contexts (Lew, 2015; Majumdar et al., 2011; Vianna et al., 2018).

For the CV question, I presented participants with a detailed hypothetical scenario in which they had taken a beach holiday to a tropical destination and were participating in a marine tourism activity. I explained that a small fishing village existed nearby, where endangered sharks are often caught for food and income, as is the case in many marine tourism destinations in the tropics, particularly in Indonesia (e.g. Glaus et al., 2018; Mustika et al., 2020). The respondents were then asked their WTP a marine

conservation fee (in addition to the price of the tourism activity) to support shark conservation in the local fishing village, by directly and conditionally compensating fishers for reducing their catches of endangered shark species (Appendix 6, S9.1).

I designed the CV question to reduce common biases by: 1) piloting payment card increments, to optimise efficiency and accuracy of responses while minimising cognitive burden and visual complexity; 2) including a cheap talk script and follow-up questions on perceived consequentiality, to reduce hypothetical bias; 3) including follow-up questions on zero responses, to separate true zeros from protest zeros (Carson & Hanemann, 2005). I conducted an initial pilot with 12 postgraduate students, then conducted a further pilot survey via Prolific (N = 25). This ensured payment card bid ranges were sufficiently sensitive to different preferences whilst not being cognitively burdensome and provided feedback on scenario design and survey clarity. The final CV question used a payment card with 18 WTP categories in US dollars, ranging from zero to US\$300. Respondents were asked to select their maximum WTP, allowing measurement of the lower and upper bound of respondent WTP.

An experimental element was also added to the CV question, whereby 50% of participants were exposed to an ‘informational intervention’ treatment before giving their WTP. Treatment participants were presented with some text and a 60-second video on the threats facing sharks and the socio-economic challenges of shark conservation in SSFs (Appendix 6, S9.1). Control participants were given no background information. The aim of this experiment was to test whether a simple informational intervention at the point-of-sale might influence tourists' WTP.

Finally, I measured participants' revealed WTP for shark conservation through a prize draw, in which survey participants had a 1 in 100 chance of winning US\$20 for participating in the survey, but could choose to donate a proportion of the prize to a real community-based shark conservation project in Indonesia (Appendix 6, S9.1).

Analysis and validation

I derived the median bid interval and a conservative estimate of mean WTP using the lower-bound of each bid interval. To verify internal consistency and construct validity I: 1) explored the relationship between stated WTP and real donations; 2) modelled determinants of stated WTP and real donations – using linear and interval regression models with $\log(\text{WTP})$ as the response variable – to test and verify correlations other widely-accepted demographic, attitudinal and behavioural constructs. I expected positive coefficients for indicators of wealth (such as income and holiday budget), education level and existing pro-environmental attitudes/behaviours (Liebe et al., 2011). The informational intervention (treatment Y/N) was also included as an explanatory variable, to test for significance and effect size. Explanatory variables were tested for co-variance, and models were fitted using all meaningful

combinations variables, via `lm` and `survreg` functions in RStudio (RStudio Team, 2020). I used backwards selection and ΔAIC to identify an optimal, statistically-significant model.

9.2.2 Assessing the feasibility of a ‘real world’ beneficiary-pays financing mechanism: two case studies

I applied the results of the WTP survey to explore the real-world application of tourism levies in two case study marine tourism sites in Indonesia: Lombok (West Nusa Tenggara Province) and Pulau Weh (Aceh Province), which - as described in the CV scenario - are in proximity to two SSFs where endangered sharks are regularly caught: Tanjung Luar (West Nusa Tenggara) and Lhok Rigaih (Aceh) (Figure 9.1)

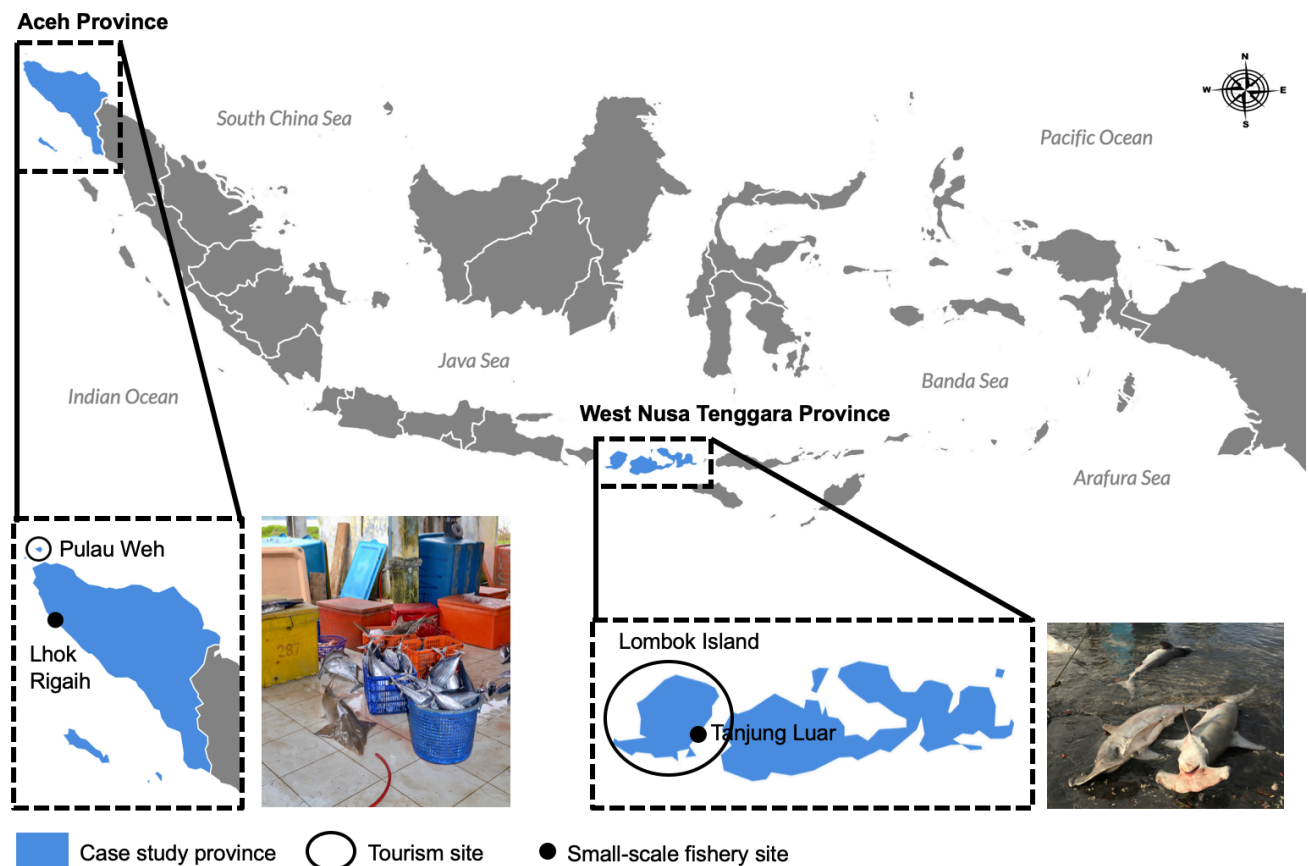


Figure 9.1 A map of Indonesia, showing the case study provinces, tourism sites and small-scale fishery sites, with photographs showing examples of a wedgefish landing in Lhok Rigaih and a hammerhead shark landing in Tanjung Luar

Lombok Island in West Nusa Tenggara is a popular marine tourism destination that attracts scuba-divers, snorkelers, surfers and beachgoers. Over 1 million international tourists visited Lombok in 2015, with the majority being European and Australian, and 22% of visits were motivated by water sports (Horwath HTL, 2017). Tanjung Luar is a small-scale semi-commercial targeted shark fishery on the east coast of Lombok. Vessels typically spend 10-20 days at sea, travelling to specific offshore fishing grounds, where they frequently capture large, mature Critically Endangered hammerhead sharks (*Sphyrna* spp.) and wedgefish (*Rhynchobatus* spp.), as well as other charismatic species such as mobula rays (*Mobulid* spp.) (I.

Yulianto et al., 2018). As the only targeted shark fishery on Lombok Island, and in relative proximity to marine tourism hotspots, it is well-known and somewhat controversial. The fishery is completely legal, and challenging to manage due to fishers' high economic dependence on shark catches, and the limited availability of sustainable and equally-profitable alternatives (Chapter 7; Booth, Squires, et al., 2021; Milner-Gulland, Ibbett, et al., 2020).

Pulau Weh is a smaller and less well-known tourism destination in Aceh Province, which primarily attracts scuba-divers and beachgoers. An estimated 29,827 international tourists visited Pulau Weh in 2018 (Sabang Culture and Tourism Agency, 2019). Lhok Rigaih is a small-scale gill net fishery, located south-east of Pulau Weh, in which juvenile hammerhead sharks and wedgefish are frequently caught as incidental catch (Chapter 5; Simeon et al., 2020). While catches of these taxa are secondary, they hold important economic and subsistence value, such that there is a lack of incentives for fishers to avoid capture and promote release (Chapter 5). Lhok Rigaih is representative of other similar SSFs along the South-West coast of Aceh, though thought to be the largest SSF landing site in the area.

For these case studies I estimated the economic feasibility of adopting tourism levies in Lombok and Pulau Weh, as a beneficiary-pays financing mechanism for marine conservation outcomes in Tanjung Luar and Lhok Rigaih. To do so, I combined available statistics on annual international marine tourists in Lombok and Pulau Weh (Q) (Horwath HTL, 2017; Sabang Culture and Tourism Agency, 2019) with the median WTP per person-day calculated from our CV survey (P) to estimate a plausible range of potential total annual revenues (R) for marine conservation from tourism levies in both Lombok and Pulau Weh (where $R = Q \times P$). Median WTP was used, rather than prediction based on the regression results, due to limited data on the demographic characteristics of tourists visiting Lombok and Pulau Weh. I then compared the estimated annual revenues for each tourism site with available data on the estimated annual costs of compensatory performance-based payment schemes to halt catches of hammerheads and wedgefish in Tanjung Luar and Lhok Rigaih (Chapter 8).

I also combined these quantitative estimates with qualitative data from the marine tourism sector in Lombok and Pulau Weh, to ground truth the data with 'real-world' priorities and industry insights on implementation and feasibility. These data were collected using semi-structured interviews, focus group discussions (FGDs) and structured surveys with marine tourism companies and marine tourists who operate in/have visited Lombok and Pulau Weh (Appendix 6, S9.2; S9.3). Questions focused on perceptions of the state of and threats to the marine environment, potential solutions, the role of tourists and the tourism sector in delivering these solutions; and preferences for different causes and institutional arrangements. FGDs included interactive listing and ranking, problem tree analysis, and conceptual and timeline mapping (CMP, 2020; Newing et al., 2010) (Appendix 6, S9.2; S9.3). In total I gathered insights

from 10 marine tourism operators and surveyed 197 tourists, and conducted thematic and descriptive analysis of the data using coding and grouping (Appendix 6, S9.3).

9.3 Results

9.3.1 Tourist Willingness-To-Pay

1,033 respondents completed the online marine tourism survey. 74 nationalities were represented, with the majority from Europe and Northern America (80%). The mean age was 30 (SD = 11), and the median income bracket was \$35,000 to \$49,999. Females were slightly over-represented at 58% of the total sample (Appendix 6, S9.4).

Most respondents found the scenario realistic and credible (95%) and likely to happen to them in real life (88%) (Appendix 6, S9.4). The 12% of respondents who reported otherwise were removed from the final sample for the WTP calculation. Of this dataset (N = 884), the median WTP fell within the US\$10 – US\$14.99 per person per day bid interval, while the mean lower-bound of WTP was \$16.56 (SD 26.5) (Table 9.1). The cumulative distribution of bid responses dropped off relatively quickly (Figure 9.2), though some respondents were willing to pay \$300, which was the highest bid offered on the payment card. Thirty-nine people (3.8% of all respondents) reported zero WTP, with most (N = 17) stating they could not afford it (“*I travel very economically*”, “*I am not a rich person*”), and others disagreeing with some aspect of the scenario (e.g., “*I think there would be a lot of corruption*”, “*There are far more important problems in the world*”).

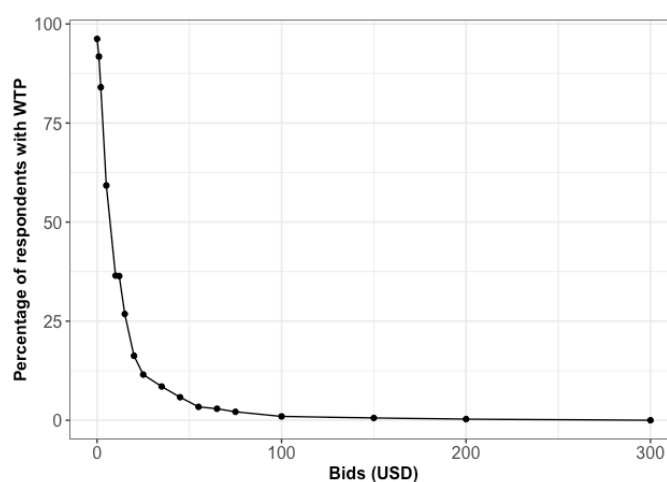


Figure 9.2 Cumulative distribution of WTP responses showing the percentage of respondents who were willing to pay the amount specified by each bid.

Table 9.1 Contingent valuation summary statistics

	Stated WTP	Donation
Mean	16.56 (SD 26.5)	7.05 (SD 6.4)
Median interval	10 – 14.99	6.00
% of zeros	3.8% (N = 17)	21.8% (N = 225)

The validation model of determinants of WTP (Table 9.2) showed that holiday budget and pre-existing charitable behaviour were significantly correlated with WTP ($p < 0.001$ and $p < 0.01$, respectively), as expected. Income also significantly correlated with WTP ($p < 0.001$), but was not included in the final model due to co-variance with holiday budget (Appendix 6, S9.4). People with positive attitudes towards shark species protection and people who had seen sharks in the wild also had a significantly higher WTP ($p < 0.01$ and $p < 0.05$, respectively) (Table 9.2). Positive but non-significant coefficients were associated with age and university-level education, while negative but non-significant coefficients were associated with male participants (Table 9.2). Some differences in WTP were also found between different nationalities on average (Appendix 6, S9.4), however these variables dropped out of the full model, as they were not significant when controlling for other variables. The informational intervention was not a significant determinant of WTP, and was not included in the final model (Appendix 6, S9.5).

Stated WTP also correlated strongly with, and was a significant predictor ($p < 0.001$) of, real donations (Appendix 6, S9.5). Linear regression models of determinants of donation amount also yielded similar predictors, with pre-existing charitable behaviour and positive attitudes towards shark conservation both significant predictors of donation amount ($p < 0.001$ and $p < 0.01$, respectively) (Appendix 6, S9.5).

Table 9.2 Outputs of final interval model of determinants of $\log(WTP)$.

Explanatory variable	Description	Value	Std Error	p-value	Sig.
(Intercept)		2.060	0.137	<2.00E-16	***
Last holiday budget (US\$ 1,000)	Continuous	0.023	0.005	6.30E-06	***
Age (years)	Continuous	-0.001	0.003	0.688	
Gender – male (vs. female)	Categorical	0.005	0.070	0.937	
University education - yes	Binary (Y/N)	0.030	0.072	0.681	
Pro-environmental behaviour - yes	Binary (Y/N)	0.174	0.069	0.011	*
Seen sharks - yes	Binary (Y/N)	0.227	0.081	0.005	**
Attitude to shark protection	Five-point scale (-2 to +2)	0.176	0.053	0.001	**

Sig. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
Gaussian distribution
Loglik(model)= -1568.4 Loglik(intercept only)= -1596
Chisq= 55.23 on 7 degrees of freedom, $p = 1.3e-09$
Number of Newton-Raphson Iterations: 3
N=670 (89 observations deleted due to missingness)

9.3.2 Assessing the feasibility of beneficiary-pays financing mechanisms in two sites

Marine tourism operators confirmed that marine megafauna (notably: sharks, rays, turtles and marine mammals) were important for healthy marine ecosystems and thriving marine tourism businesses, along with pelagic and reef fish and coral reefs (Appendix 6, S9.3). Tourist perceptions aligned with this, with reef sharks being the most frequently reported marine animal that tourists were “most excited to see”

overall (Figure 9.3). However, marine resources were also recognised as degraded and threatened, particularly in Lombok. For example, just 35% of tourists who had visited Lombok felt the marine environment was in a positive condition (Figure 9.3). Of the environmental issues tourists noticed, damaged reefs were the most common (65%) as well as too much pollution (55%) and too few fish/marine animals (43%). The operators also ranked overfishing as the biggest threat to sharks (Appendix 6, S9.3). These threats were felt to derive from local communities (e.g., those fishing for food) as well as tourists and inexperienced tourism operators (e.g., tourists treading on corals, or poor operator practices such as dropping anchor on coral) (Appendix 6, S9.3).

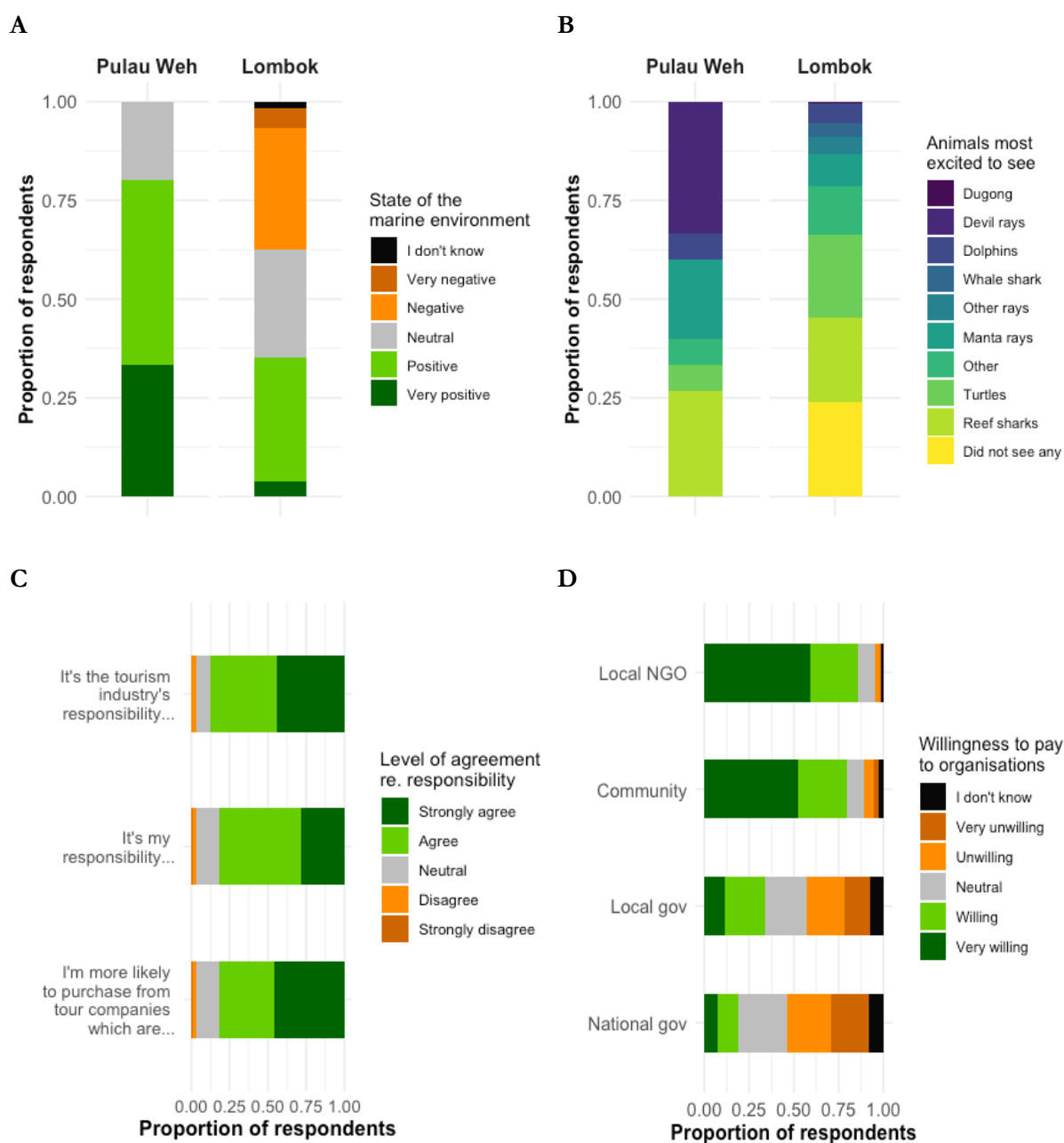


Figure 9.3 Summary of key tourist perceptions, including a) perceived state of the marine environment in Pulau Weh and Lombok, b) animals which tourists were most excited to see c) attitudes regarding responsibilities and preferences for tackling environmental issues, d) attitudes regarding which institutions should be funded.

Solutions suggested by tour operators included outreach and engagement, fishing restrictions (including marine protected areas (MPAs)), and livelihood-based interventions for communities; training and codes of conduct for tour operators; and habitat restoration efforts, such as coral and mangrove planting. All marine tour operators exhibited a general willingness to contribute towards marine conservation, including an interest in establishing tourism levies, provided schemes were transparent and money was spent locally. Similarly, most tourists agreed that tourism companies and tourists themselves had a responsibility to tackle environment issues (88% and 82%, respectively), and 82% agreed or strongly agreed that they would be more likely to purchase goods and services from tourism companies which are helping to tackle environmental issues (Figure 9.3). Further, 92% of surveyed tourists stated they would be willing to pay towards marine conservation projects in those areas, though they also exhibited preferences for which institutions should be funded: there was strong support for funding environmental NGOs or direct payments to local communities, but comparatively little support for funding national or local governments, primarily due to a lack of trust that the funding would be re-invested locally (e.g. *“Worry that funding to national govt could be diluted or shifted elsewhere”, “Everything should be managed on the local level”, “I’m happy to support local initiatives that support the local community, over larger companies or government”, “I believe that local NGOs... tend to know where the problems lie and typically have close ties already with the local community”*) (Figure 9.3).

Estimated revenues from tourism levies

In 2015 Lombok hosted 1.03 million international visitors, for an average of 2.3 nights each, 22% of whom were primarily motivated by water sports, especially diving (Horwath HTL, 2017). This equates to a conservative estimate of 226,600 marine tourists per year, which assumes no growth in tourism markets since 2015. Assuming each of these tourists were willing to pay the median conservation levy of US\$ 10-14.99 per person per day, and participated in just one day of marine tourism activities per trip, this could generate US\$ 2.2 – 3.7 million in conservation finance for Critically Endangered sharks in Lombok, or US\$ 4.5 -7.5 million if each visitor participates in 2 days of marine tourism activities (Table 9.3). It is also estimated that by 2026 there could be up to 1.8 million foreign visitors to Lombok (Horwath HTL, 2017), which would almost double these estimated revenues.

In 2018, Pulau Weh hosted 29,827 international visitors, who were primarily motivated by snorkelling and scuba diving (Sabang Culture and Tourism Agency, 2019). If each participated in just one day of marine tourism activities per trip, this could generate approximately US\$ 300,000 – 500,000 in marine conservation finance per year, or up to US\$ 1 million per year if each visitor participates in 2 days of marine tourism activities (Table 9.3).

Table 9.3 Estimated annual revenue (\$US) from marine tourism conservation levies in Lombok and Pulau Weh. Estimated international marine tourists per annum based on Horwath LTD (2017) for Lombok and Sabang Culture and Tourism Agency (2019) for Pulau Weh

Location	International marine tourists per annum	Days of marine activities per person	Median interval lower-bound (10 per day)	Median interval upper-bound (14.99 per day)	Mean (16.56 per day)
Lombok	226,600	1	2,266,000	3,396,734	3,752,496
		2	4,532,000	6,793,468	7,504,992
Pulau Weh	29,827	1	298,270	447,107	493,935
		2	596,540	894,214	987,870

Estimated costs of community-based conservation, and comparison with revenues

In Chapter 8 I estimated that fishers in Tanjung Luar would be willing to accept performance-based compensatory payments of \$83 - \$282 and \$110 – 588 per fishing trip to halt landings of hammerhead and wedgefish, respectively. Based on average total trips per year by all fishers operating out of the port, and their average catches of hammerheads and wedgefish per trip, it would cost \$42,600 – 144,000 to incentivise fishers to stop landing hammerheads in Tanjung Luar (saving roughly 500 individuals per year), and \$14,980 – 79,940 to stop landing wedgefish (saving roughly 140 individuals per year) (Chapter 8). This suggests that even the highest estimated cost for a PES scheme, which would produce major conservation benefits for Critically Endangered hammerhead sharks and wedgefish in Lombok (i.e., ~\$US 220,440 per year), is an order of magnitude lower than the minimum estimated annual revenue from marine tourism levies on visitors to Lombok (US\$ 2.27 million, Table 9.3).

In Lhok Rigaih, it would cost an estimated \$4,680 – 46,800 per year to incentivise fishers to reduce their catches of juvenile hammerheads, and \$9,360 – 28,080 for wedgefish (Chapter 8). In this case, the minimum estimated annual revenue from marine tourism levies in Pulau Weh (\$US 298,270, Table 9.3) is around four times greater than the highest estimated cost for a PES scheme in Lhok Rigaih (i.e., \$US 74,880 per year), and could save over 18,000 juvenile hammerheads and 2,000 sub-adult wedgefish. As the largest SSF landing site in the area, this scheme would produce meaningful conservation benefits for these taxa in Aceh, and with the potential revenues available, could be replicated in other similar smaller fisheries throughout the region.

9.4 Discussion

9.4.1 Interpretation

This study provides a first attempt to estimate a generic willingness-to-pay for conservation outcomes for endangered sharks amongst international tourists. The regression model and real donations validate the findings, with WTP correlating with income and holiday budget, as well as other indicators of pro-environmental attitudes and behaviour. The lack of significance of the information intervention, combined with the significance of other predictors of pro-environmental attitudes and behaviour,

suggests that provision of information at point-of-sale may do little to increase WTP, and rather that WTP is dependent on pre-existing pro-environmental norms. The preliminary assessment of two potential beneficiary-pays mechanisms in Indonesia indicates that even relatively modest tourism levies could generate large and impactful sources of funding for community-based PES schemes, which, in turn, could have meaningful conservation benefits for Critically Endangered taxa (Chapter 8). Moreover, well-designed, locally-implemented, conservation programmes could have wide support from tour operators and tourists alike, and could also boost tourist satisfaction and marine tourism revenues.

The data (which has been made available in an open access repository (Booth, 2021b)) could be applied to other studies or policy planning processes which require estimates of the travelling general public's economic values for the conservation of endangered shark species; and the methods to link this global estimate to local situations could be applied to other case study locations, conservation issues and types of nature-based tourism, where there is a need to more equitably redistribute the costs and benefits of conservation.

9.4.2 Biases and limitations

The survey sample exhibits both coverage bias and selection bias (Bethlehem, 2010; Lehdonvirta et al., 2021). It is heavily biased towards European and Northern American segments of the international marine tourism market. The results should therefore be used with caution when attempting to extrapolate more broadly. While nationality was not identified as a significant variable in predicting WTP, sample sizes for some nationalities – notably China, Japan and North Korea, which are thought to make up large and growing segments of the international marine tourism industry (particularly scuba-diving, e.g., based on international equipment sales (Kieran, 2019)) – were small (Appendix 6, S9.4). Other studies on environmental attitudes and WTP for environmental outcomes amongst Chinese consumers suggest concern and WTP for conservation are quite low (Fabinyi & Liu, 2014). This means the results may skew high relative to a more internationally balanced sample, and countries or sites which have a particularly large share of visitors who are not European or North American may not be able to apply the data and model reliably.

For the case study sites, applying the median CV from the survey assumes the online panel is representative of the international tourists who will a) visit Lombok and Pulau Weh, and b) participate in a marine tourism activity as described in the survey. Based on available data on tourism demographics, European visitors represented 50% of all foreign arrivals in Lombok in 2015, with 18% from Australia and 10% from Americas (Horwath HTL, 2017). Tour operators also indicated that most of their guests are European, which suggests the study panel is a reasonable fit for the Lombok tourism demographic. Our estimates of 1-2 days of marine tourism activities may be conservative, however, since Horwath HTL (2017) noted that European visitors tended to stay considerably longer in Lombok than the average

visitor, with typical trip lengths of 4-7 nights. I was not able to access data on the nationalities of international visitors to Pulau Weh, however tour operators indicated there is a mixture of Europeans, North Americans, Australians and Malaysians. The number of Malaysians included in the online survey panel was low, creating an unknown bias in the revenue estimate. However, a previous study on divers' WTP daily fees for shark conservation in Malaysia, in which European divers were compared with divers of domestic/Asian origin, did not find region of origin to be a significant predictor of WTP (Vianna et al., 2018). I also did not find nationality to be a significant predictor of WTP in the full CV model (Appendix 6, S9.5).

This sample also does not capture people within very high-income brackets, who may be willing to pay much more towards marine conservation, since WTP is income dependent. This means the results may skew low relative to the international marine tourism market overall, particularly since some marine tourism destinations are very high-end. This is unlikely to be problematic for the Lombok and Pulau Weh extrapolations, since most tourists visiting these areas are typically budget or mid-range, but means that the data cannot be reliably applied to more expensive, luxury destinations, such as Raja Ampat, the Maldives or French Polynesia. The online WTP survey also did not capture Indonesian nationals, which means that WTP and total conservation revenues for domestic tourists to Lombok and Pulau Weh could not be estimated. While WTP per person may be lower for Indonesian nationals, given lower Purchasing Power Parity compared with Europe and North America, the domestic tourism market in Indonesia is significant. For example, domestic visitors to Lombok reached 952,648 in 2015, representing 48% of total visitors (Horwath HTL, 2017). Based on this, even a small domestic tourist fee (e.g., US\$ 1 per person per day) could considerably increase annual revenues from marine tourism conservation levies.

Finally, self-selection bias is a common issue in online surveys, however since the target population was the broad spectrum of people who are interested in travel, this is not necessarily problematic for this study (Bethlehem, 2010; Lehdonvirta et al., 2021). Moreover, I did not market the study as a marine conservation study but simply as a marine tourism study, which should have helped to guard against people with an interest in conservation tending to take the study more than those with no interest in conservation. This is supported by the observed variability in responses to questions about respondents' existing pro-conservation behaviours and attitudes (Appendix 6, S9.4).

9.4.3 Implications and next steps

Coastal communities and small-scale fishers (SSF) often face an inequitable burden of the costs of marine conservation (Chapter 7; Booth, Squires, et al., 2021; Jaiteh et al., 2016; Stevenson et al., 2013). This must be addressed in order to achieve '*a sustainable and equitable blue economy*' and move towards socially-just marine conservation (N. J. Bennett et al., 2019, 2021). However, there is a lack of operational mechanisms for incentivising pro-conservation behaviour and redistributing the costs and benefits of

marine conservation. Nature-based marine tourism is often promoted as a win-win solution to trade-offs between marine conservation and coastal livelihoods, yet is often difficult to implement in practice, since those who benefit from consumptive use of marine resources are not necessarily well-placed to work in and benefit from the tourism sector (Balmford & Whitten, 2003; Mustika et al., 2020).

This study has shown how the beneficiary-pays principle could be applied to the marine tourism industry to generate a feasible, socially just, and scalable ocean financing mechanism, which can support small-scale fishers to reduce their impacts on marine life. In the case study sites, this could operate through performance-based PES schemes to mitigate fishing mortality of specific Critically Endangered species (Chapter 8). If implemented elsewhere, such mechanisms could also fund marine protected areas (MPAs); the provision of bycatch reduction technologies or more sustainable fishing gears for communities who otherwise could not afford it; and/or a range of other activities such as beach cleans and habitat restoration (Bladon et al., 2016; Gjertsen et al., 2014; Pakiding et al., 2020; Sykes et al., 2018; Vianna et al., 2018). Indonesia hosts an estimated 18 million reef-associated tourists per year, with a total of around 70 million reef-associated tourists per year globally (Spalding et al., 2017). Assuming all of these tourists are willing to pay a minimum of US\$ 10 during just one day of marine tourism activities, at least US\$ 180 million could be generated for marine conservation in Indonesia via marine tourism levies, and over US\$ 70 million globally. This could contribute considerably towards financing MPAs and closing marine biodiversity financing gaps (Balmford et al., 2004; Johansen & Vestvik, 2020). Moreover, such investments could not only deliver biodiversity and well-being improvements in SSFs, but would help to maintain valuable natural assets upon which marine tourism companies and the blue economy depend, as well as improve customer satisfaction and tour operator reputation.

Despite this potential, I acknowledge that this study represents a simple preliminary economic feasibility assessment. Putting such mechanisms into practice would require strong institutions and long-term engagement by facilitating organisations, together with monitoring and enforcement, to ensure that funding is appropriately collected, re-distributed and used to create measurable conservation outcomes. However, such operating costs could also potentially be covered by marine tourist levies, given the potential surplus of income relative to the cost of community-based PES schemes. Moving forwards, I urge researchers, NGOs, tourism operators and policy makers to explore the feasibility of applying beneficiary-pays approaches to other locations where mismatches between the costs and benefits of marine conservation need to be reconciled, as a central mechanism for delivering a sustainable and equitable ocean economy.

10. Bycatch levies: operationalising the polluter-pays principle to reconcile trade-offs between blue growth and biodiversity conservation

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Photos: Trawlers and their bycatch. Central Java, Indonesia.

*“You say you want a revolution
Well, you know
We all want to change the world
...
You say you got a real solution
Well, you know
We’d all love to see the plan”*

- The Beatles.

10.1 Introduction

Ocean ecosystems support the well-being of billions of people (Golden et al., 2016; Halpern et al., 2015), and are projected to contribute US\$ 3-trillion to the global economy by 2030 via ‘blue growth’ (OECD, 2016). Blue growth aims to deliver economic growth through exploitation of marine resources, while also preventing their degradation (Boonstra et al., 2018). However, there is a need to reconcile expectations for economic growth with biodiversity conservation (N. J. Bennett et al., 2019; Boonstra et al., 2018), because economic activities in the ocean inherently jeopardise marine ecosystems (Halpern et al., 2015; Maxwell et al., 2013; Nash et al., 2017).

In parallel to the blue growth agenda, the UN Convention on Biological Diversity (CBD) is developing a post-2020 framework to “put biodiversity on a path to recovery” (CBD, 2020b, 2020a). Ambitious targets and an integrated approach are needed to bend the curve on global biodiversity loss, while also delivering improvements to human well-being (Bull et al., 2020; Leclerc et al., 2020; Mace et al., 2018). As such, the CBD post-2020 framework is likely to include net outcome goals, either explicitly or implicitly (Bull et al., 2020; CBD, 2020a; Milner-Gulland, Addison, et al., 2020). Net outcome goals imply that some biodiversity may be lost, provided it is gained elsewhere, resulting in an overall balance (no net loss (NNL)) or increase (net gain) in biodiversity (Chapter 3). Transformative change is needed to integrate net outcome goals into fisheries management objectives, and thus establish profitable yet biodiversity-neutral (i.e., $\geq$ NNL) fisheries within the coming decade.

Fisheries bycatch (defined here as incidental catch that is either “unused or unmanaged” (Davies et al., 2009)) poses a major threat to marine biodiversity (M. A. Hall et al., 2000), and is one of the most intractable challenges for reconciling blue growth with biodiversity conservation. Bycatch is costly to fishers and society, because it threatens biodiversity, ocean ecosystems and the viability of fisheries (Branch et al., 2006; Crowder & Murawski, 1998; Davies et al., 2009; Gilman et al., 2007; Lewison et al., 2004). However, mitigating bycatch is also costly to fishers and ocean managers, as it can be data intensive, require new or cumbersome technologies, and create opportunity and management costs (Campbell & Cornwell, 2008; M. A. Hall, 1996; J. T. Watson et al., 2009). In economic terms, bycatch is a negative externality, comparable to carbon emissions and air pollution (V. L. Smith, 1969), which occurs when an economic transaction by a private economic entity (e.g., a fishing firm) imposes a cost on society, which is unpriced or only partially priced by markets. As a result, the environmental costs of fishing to society exceeds the private cost of fishing to firms, and the market maximises private benefit for the firm, but not total benefit to society (i.e., social welfare, wherein economic welfare is a subset of social welfare, which can be measured in monetary terms) (Figure 10.1).

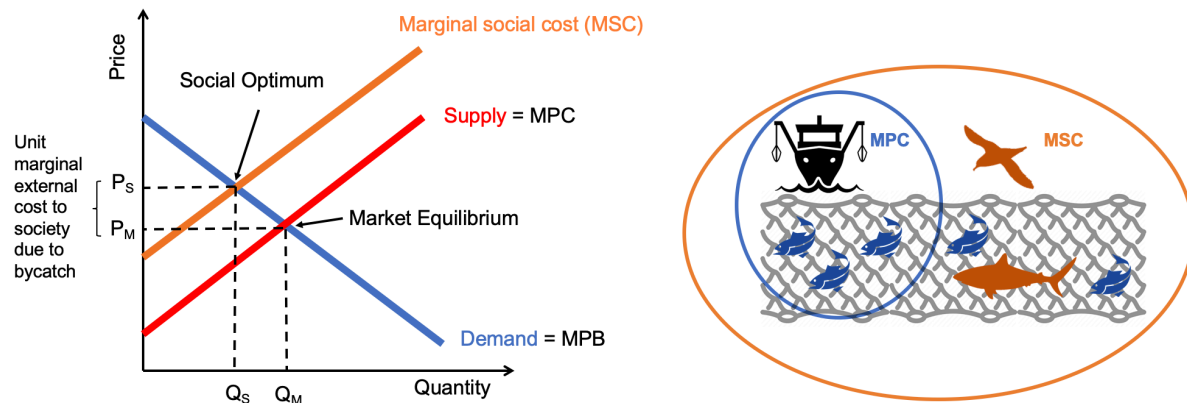


Figure 10.1 Bycatch as an externality. The difference between the market equilibrium and socially-optimal quantity of production when there is a negative externality (MPC = marginal private cost, MPB = marginal private gain, MSC = marginal social cost), depicted through: (a) a supply-demand relationship (where P_M = price at market equilibrium, P_S = price at social optimum, Q_M = quantity demanded at market equilibrium, Q_S = quantity demanded at social optimum) and (b) a figure of fishing operations, where target catch is coloured blue, bycatch is coloured orange

A mix of instruments is needed to mitigate bycatch and restore bycatch-affected populations (Innes et al., 2015). The types of instruments that can be used fall in to two broad categories: direct regulation and market-based approaches (Figure 10.2). Direct regulations are implemented via command-and-control, while market-based approaches can be mandated through public policy (e.g., taxes) or adopted voluntarily (e.g., certification schemes). To date, most bycatch mitigation measures focus on direct regulation, with mandated technical fixes such as gear specifications or catch bans. Such measures can be simple, easy to enforce and have low management costs; yet also inflexible, and can fail to deliver socially-optimal outcomes or incentives for innovation when used in isolation (Figure 10.2) (Campbell & Cornwell, 2008; Fulton et al., 2011). As decision-makers usher in the blue economy, there is growing discussion around market-based fisheries management (e.g., Bladon et al., 2016; Sumaila et al., 2016), and particularly the need to end harmful subsidies, which currently far out-weigh conservation spending (Deutza et al., 2020; Sumaila et al., 2016). However, incentive-based instruments for bycatch management, such as bycatch levies, have received comparatively little attention. While fisheries economists have explored the potential of bycatch levies for decades (Dutton & Squires, 2008, 2011; M. A. Hall, 1996; Innes et al., 2015; Pascoe et al., 2010; Wilcox & Donlan, 2007) there remain few real-world applications (See section 2.5). This represents a missed opportunity for closing the gap on marine conservation financing (Sumaila et al., 2020).

One possible reason for the limited application of bycatch levies is that they have yet to be deemed societally acceptable. Under the 'Polluter Pays Principle' (i.e., the principle that society holds a right to a healthy environment, so those who create environmental costs should bear the costs of managing them) bycatch levies require a willingness-to-accept (WTA) monetary compensation to society for damage to biodiversity (Squires et al., 2018). This entails a utilitarian approach to valuing biodiversity, with

consensus on permissible levels of bycatch and its monetary value, which remains controversial. For example, proposed compensatory offsets for seabird bycatch (Wilcox & Donlan, 2007) were met with value-based concerns regarding costs to biodiversity and limits of permissible risk (Finkelstein et al., 2008; Żydelis et al., 2009). Levies also require a willingness-to-pay (WTP) compensation amongst fishing firms. WTP may be limited in the absence of regulation or clear reputational risks or advantages, since levies are likely to reduce profit margins, particularly if price premiums cannot be secured from consumers (Blomquist et al., 2015; Roheim et al., 2018). Securing WTP will be even more challenging (and arguably unethical) in small-scale fisheries, where profit margins are already low.

However, with growing adoption of net outcome approaches for biodiversity conservation more broadly (Bull & Strange, 2018; Shumway et al., 2018), and the need to close the global biodiversity financing gap (Deutza et al., 2020), it is timely to re-visit the potential of bycatch levies in fisheries. Here I describe how well-managed bycatch levies could help to deliver net positive outcomes for marine biodiversity, alongside ‘blue growth’ (i.e., economic welfare outcomes) for society (Gjertsen et al., 2014) (Figure 10.2), potentially leading to win-wins for blue growth and biodiversity. I explore key considerations for designing bycatch levies based on economic theory and real-world examples, and discuss the next steps for mainstreaming beneficial bycatch levies into the blue economy.

10.2 Bycatch levies for net biodiversity outcomes

Mitigating bycatch requires changing the behaviour of fishing firms (i.e., individual fishers, vessels, fleets and companies) and consumers. This can be achieved either through direct regulation or market-based interventions. Direct regulations typically mandate fixed standards on catch, bycatch, fishing process, or technology, while market-based interventions create incentives for individual firms to alter their behaviour towards a societal optimum, allowing for individual innovation (Figure 10.2). The most basic type of market-based intervention for bycatch is a levy that sets a price on each unit of bycatch. Such levies can be established voluntarily or mandatorily, and provide a ‘double dividend’ through two types of benefits: first, by incentivising bycatch prevention; second, by raising revenues for compensatory conservation, to improve the status of bycatch-affected populations or to finance socially-underprovided research and development for bycatch-minimising technologies (Dutton & Squires, 2011; Segerson, 2011). In principle, these two dividends can together deliver NNL or net gain outcomes for bycatch affected species (Figure 10.2); though there will be inevitable time lags, uncertainties and governance challenges for securing compensatory outcomes in practice (Bull et al., 2013; zu Ermgassen et al., 2019, 2021).

10.3 Designing bycatch levies for net biodiversity outcomes

Bycatch levies are not a silver bullet for all types of market failure that exacerbate bycatch (e.g., weak property rights and asymmetries in power and information). Their scope is limited to addressing bycatch as a negative externality, by putting a marketable price on its external cost; and to situations where at least some of the actors causing bycatch issues are prepared to pay levies, and society is prepared to accept compensatory actions to recover lost biodiversity rather than requiring that bycatch is prevented entirely. Levies can be established through public regulation, such as mandatory bycatch taxes (e.g., New Zealand and Namibia's quota management systems, Section 10.3.6) (Pascoe et al., 2010)); or voluntarily, such as conservation payments as part of corporate social responsibility (Kotchen, 2013; Segerson, 2013) (e.g., the International Seafood Sustainability Foundation (ISSF), Section 10.3.6) (Janisse et al., 2010; Squires & Garcia, 2018). Though mandatory and voluntary adoption of bycatch levies are limited at present, changes in environmental governance, investor expectations and wholesale and retail markets could drive expansions of marine biodiversity offset markets in the coming decade (Jacob et al., 2020), and substantially increase the scope for levies to be implemented.

Moreover, the extent to which bycatch levies deliver each dividend, and therefore their pathway to conservation outcomes, depends on their design. Different design choices – including how to target, price and invest the levies – have different pros and cons. As such, the optimal levy design choices depend on the fishery and socio-economic context in which is it implemented (Table 10.1).

10.3.1 Targeting levies

A key design choice is setting the level of production for targeting the levy (Kotchen, 2013); and relatedly, the outcome on which to base the levy (Kotchen, 2013; Segerson, 2011, 2013). Regarding level of production, a levy could be applied at an industry-wide scale (e.g., applied across all producers of a given species, nationally), a fleet-wide scale (e.g., applied to a particular sub-set of jointly-operating vessels, which fish in a specific seascape or use a certain bycatch-intensive technology) or vessel-specific. Regarding outcomes, the levy could be set according to fishing practices (i.e., process-based) or actual bycatch events (i.e., performance-based)) (Figure 10.3, Table 10.1) (Segerson, 2011). Different design choices will have different pathways to conservation impact, and entail different trade-offs between incentives for innovation and uncertainty/management costs. As such, the optimal design choice for targeting a levy depends on the fishery and management context (Figure 10.3, Table 10.1).

In terms of the first dividend, a bycatch levy can incentivise firms to mitigate bycatch in two ways: 1) altering the bycatch-catch ratio, e.g., through innovation; or 2) altering the level of effort, which lowers both bycatch and catch. These two strategies are not equally feasible across all fisheries (M. A. Hall, 1996). Their feasibility depends on the relationship between target catch production and bycatch, in particular, the homogeneity and frequency of bycatch events (Segerson, 2011). Across different fisheries

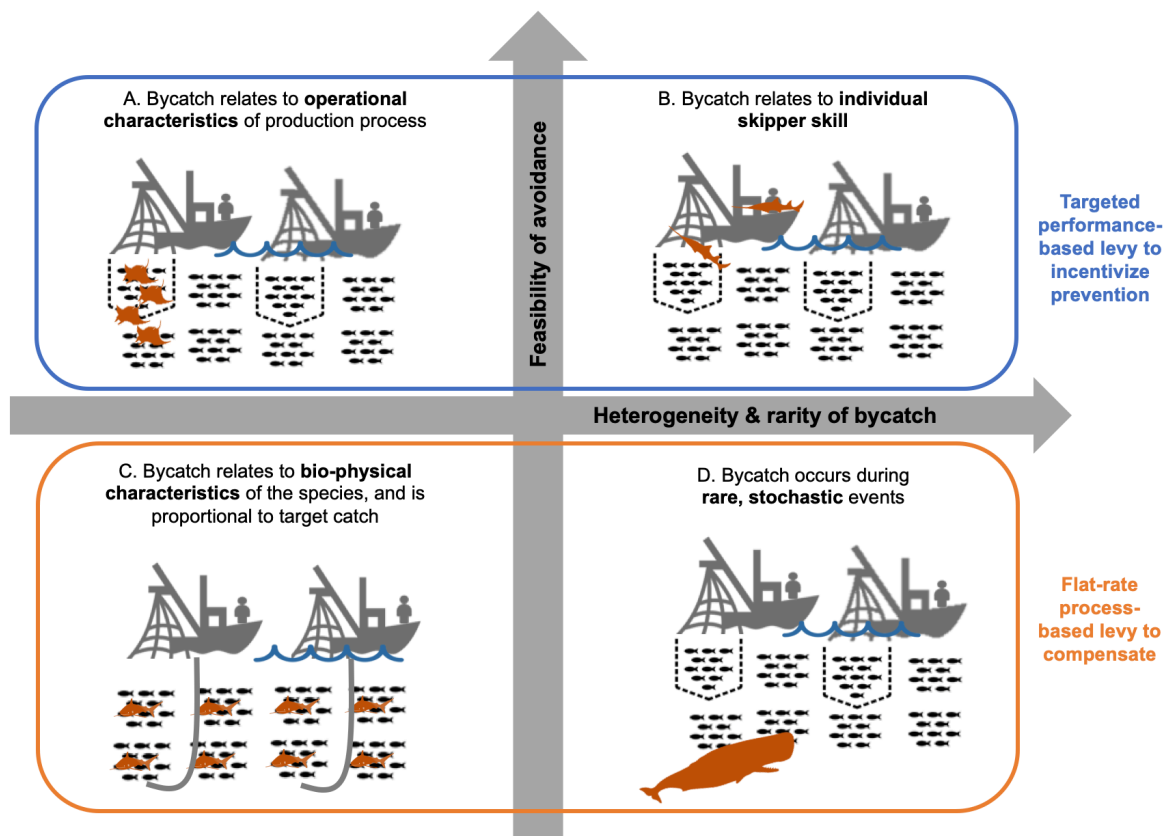


Figure 10.3 A schematic of different possible relationships between target catch production and bycatch, and how different levy designs might lead to conservation impact for different situations. Situation A, where bycatch is homogenous and avoidable, could be exemplified by mobulid ray bycatch in purse-seines during spatio-temporal aggregations; situation B, where bycatch is rare avoidable could be exemplified by sawfish bycatch (and on-board handling/release) in demersal trawls; situation C, where bycatch is homogenous, frequent and difficult to avoid could be exemplified by epipelagic requiem shark bycatch in pelagic tuna longlines; situation D, where bycatch is a rate stochastic event, can be exemplified by sperm whale bycatch in drift nets.

Table 10.1 An overview of different bycatch levy models, their relative pros and cons, and the situations to which they may be most suited. (* short-term = under current levels of technology; technological innovations may alter bycatch-catch ratios in the

Model		Pros and cons				Fishery situations	Example bycatch problem	Pathway(s) to conservation impact
		Potential for incentivising bycatch prevention (in the short-term*)	Potential for conservation investments through tax revenues	Uncertainty and management costs	Potential for collective or peer influences			
Flat-rate process levies	Industry-level flat-rate levy on fishing process, based on target catch production	Low – limited incentives for innovation (though could drive technological innovation in medium- to long- term)	High – continuous predictable revenue	Low – flat rate based on effort or production, which are relatively easy to monitor	Low – limited incentives for cooperation or peer pressure	Bycatch is proportional to target catch (homogenous, predictable, fixed bycatch-catch ratio, highly costly to avoid)	Epipelagic shark bycatch in pelagic tuna longlines	Compensatory through revenues
	Fleet-level flat-rate levy on fishing process, based on fishing operations (with option for performance-based rebates)	Moderate – incentives for fleet-level innovations	Moderate – continuous revenue but will vary over time and by fleet	Moderate – requires some information on fishing operations to determine impact	High – requires fleet cooperation, potential for peer monitoring between fleets	Bycatch varies with broad-scale biophysical or operational characteristics (somewhat homogenous and predictable, some variation in bycatch-catch ratio, can be avoided at moderate cost)	Mobulid ray (<i>Mobulid</i> spp.) bycatch during seasonal aggregations	Compensatory through revenues and preventative through behaviour change
Individually-targeted performance levies	Individual-level targeted levy on bycatch events, based on interactions or mortality	High – incentives for vessel-level innovation	Low – variable and diminishing revenue as vessels adapt	High – based on actual bycatch events, which are relatively costly to monitor	High – competition between vessels creates potential for peer monitoring between vessels	Bycatch varies with skipper skill (heterogenous and predictable, high variation in bycatch-catch ratio, can be avoided through individual behaviour change)	Sawfish (<i>Pristis</i> spp.) bycatch in demersal trawls	Preventative through behaviour change
	Fleet-level targeted levy on bycatch events (with options for risk-pooling through insurance/assurance bonds)	Low – prevention options are limited (though could drive technological innovation in medium- to long- term)	Moderate – compensation occurs but relies on one-off events, revenue is variable	High – based on actual bycatch events, which are rare and costly to monitor	High – potential for cooperative risk-pooling across-fleet and peer pressure	Bycatch is rare and stochastic (heterogenous, unpredictable, high but uncontrollable variation in bycatch-catch ratio, cannot be avoided due to stochasticity)	Sperm whale (<i>Physeter macrocephalus</i>) bycatch in drift nets	Compensatory through revenues

bycatch-catch ratios can range from homogenous and frequent, in which bycatch relates to inherent biophysical characteristics of the target and bycaught species, and is directly proportional to target catch (e.g., mako shark (*Isurus spp.*) bycatch in swordfish fisheries); to heterogenous and rare bycatch, in which bycatch is stochastic and unpredictable (e.g., leatherback sea turtle (*Dermochelys coriacea*) bycatch) (M. A. Hall, 1996; Segerson, 2011). In the middle are situations where bycatch varies with broad-scale characteristics of the fishing process, such as fishing ground, gear and season, or with individual skipper skill (Figure 10.3, Table 10.1). These characteristics influence the possible mechanisms and costs of avoidance, where to target the levy, and therefore how it might lead to conservation impact. For example, if bycatch is homogenous and altering the bycatch-catch ratio is unfeasible in the short-term, a flat-rate industry-wide process levy could lead to some bycatch prevention by incentivising reductions in fishing effort, and could provide stable revenues for compensatory conservation. Levies could also operate at other points in the supply chain, such as via processors, wholesalers and consumers. Conversely, if bycatch is heterogenous and avoidable, with possibilities for lowering the bycatch-catch ratio, an individually-targeted performance levy could incentivise individual vessels to innovate or exit the industry. If bycatch is stochastic, a standard fleet-level compensatory lump-sum may be appropriate, with fisher risk-pooling and insurance to smooth the financial risk and improve fishers' perceived fairness (Lent & Squires, 2017; R. Zhou & Segerson, 2016) (Figure 10.3, Table 10.1).

Bycatch levy design choice also entails trade-offs between incentives for innovation, uncertainty/management costs and stability of compensatory revenues for conservation. The closer the link between the cost of the levy and true bycatch performance, the stronger the incentive to alter the bycatch-catch ratio; however management costs will be higher (Segerson, 2011) and revenues for conservation via the second dividend will vary. For example, in flat-rate process-based levies, such as all vessels paying a standard bycatch charge as part of their fishing permit, there will be stable revenues for conservation and lower monitoring uncertainty, since the cost of the levy is pre-determined and true bycatch performance is not measured. However, incentives for short-term behaviour change to reduce bycatch will be limited, though this levy may drive technological innovations to reduce target catch-bycatch ratios in the long-run. Conversely, individually-targeted performance-based levies, such as a charge for each individual of a threatened species landed by a vessel, create stronger incentives for bycatch prevention. However, revenues for conservation will vary, and the degree of uncertainty faced by fishers and managers will be higher. There will also be information asymmetries and high transaction costs to the management authority, with a need for robust monitoring (such as human or electronic fisheries monitors) to measure bycatch events and ensure compliance (Table 10.1). One option for reducing costs to managers is setting standard rates or advanced payment based on an expected mean value of bycatch (e.g., according to past observer data), with performance-based rebates which place the burden of proof on to the firms themselves (Lent & Squires, 2017). Regardless, least-cost NNL is achievable in each model (Squires & Garcia, 2018), through prevention of bycatch in an individually-

targeted performance-based levy, and compensation for bycatch in a flat-rate process-based levy (Table 10.1). However, it is likely that the second dividend will erode in the long-run, as firms shift to better practices and lower bycatch-catch ratios (where technologically feasible). As a result, the first dividend will likely become the dominant pathway to NNL in the long-run. This requires that any conservation actions which are funded by compensatory offsets are future-proofed, either by ensuring that investments are established as transitional from the outset (e.g., to cover transition to a new technology or practice) or securing new sources of income over time as the second dividend erodes.

10.3.2 Pricing levies

A socially-optimal levy price is one at which the marginal private cost of bycatch prevention is equal to its marginal social benefit (Gjertsen et al., 2014; Squires & Garcia, 2018). However, calculating optimal prices is often unfeasible (Kroege, 2013), particularly given challenges surrounding valuing biodiversity (Ledoux & Turner, 2002). Rather, measuring marginal costs and improving cost-effectiveness may be more realistic, where the costs of different bycatch mitigation approaches are assessed and compared (Booth, Squires, & Milner-Gulland, 2019a; Bull & Milner-Gulland, 2020; Gjertsen et al., 2014; Squires & Garcia, 2018). One option is setting prices based on the average costs of compensatory conservation, such as habitat protection (Gjertsen et al., 2014), invasive species eradication (Holmes et al., 2016; Norton & Warburton, 2015; Wilcox & Donlan, 2007), or funding bycatch technology/performance-based incentives for bycatch abatement in small-scale fisheries (Chapter 8). Alternatively, societal values for bycaught species could be inferred from tourist expenditures or public willingness-to-pay surveys using benefit transfer methods (Gallagher & Hammerschlag, 2011; Mustika et al., 2020; O'Malley et al., 2013; Vianna et al., 2018).

For example, in the Hawaii swordfish longline fishery, it is estimated that 0.01 leatherback turtles are taken as bycatch per thousand hooks (2004-2012) (Swimmer et al., 2017). The annual cost of leatherback turtle nesting habitat conservation is US\$1,558 per adult female leatherback (Gjertsen et al., 2014). As such, a flat-rate fleet-wide levy of US\$15.58 per thousand hooks could be applied, or an individually-targeted production levy of US\$1,558 per bycaught individual. In the Eastern Pacific tuna purse seine fishery, bycatch mortality of mobulid rays is estimated at 0.16 mobulid rays per set (Croll et al., 2016). As a proxy societal value, an individual mobulid ray is worth up to US\$1,620 per individual per year in the Maldives dive industry. A flat-rate fleet-wide tax of US\$259.2 per set could be applied, or, at an estimated 200 sets per trip, a vessel could be required to purchase US\$51,800 in bycatch quota before being allowed to fish, with post-trip rebates for good performance (R. C. Anderson et al., 2011). However, such figures may under- or over-estimate socially-optimal rates, e.g., by failing to account for the opportunity costs of bycatch reduction or for intangible, indirect (e.g., via ecosystem processes), diffuse and shared values of the species concerned to society (Farley, 2012; Kenter et al., 2011). For example, US\$51,800 per trip may be a prohibitively large sum, which no purse seine vessels are willing or able to pay, resulting in fishers

being unable to conduct any fishing trip, and a loss of this fishery value to society. Conversely, the Maldives dive industry figures may underestimate the societal value of mobulids. For example, US\$1,620 is based on an annual tourism value per individual, rather than the value of an animal over its entire lifetime, which may be as high as US\$1-million (O'Malley et al., 2013). Societal values are also likely to shift as species recover or become rarer, or as technology changes. Thus, adaptive management will be needed to update pricing over time.

Levy rates could also account for extinction risk and monitoring uncertainty via risk premiums. For example, if a species' population size is low, the severity of a single bycatch event is high, warranting a risk multiplier. This could allow weighting of levies according to species endangerment, which may be important for less charismatic bycatch-affected species (e.g., wedgefish (*Rhinidae* spp.) and short-nosed sea snakes (*Aipysurus apraefrontalis*)), which require management but may have limited tangible societal value. Possible substitution or displacement effects also warrant consideration to avoid unintended cross-taxa conflicts: levy pricing should be designed holistically, considering all bycatch affected species within a given fishery (Gilman et al., 2019). Rates could also increase with mortality per trip/vessel, as per New Zealand's deemed-value quota management model (Pascoe et al., 2010) (Section 10.3.6). Multipliers could also be applied to roughly account for the uncertainty created by imperfect monitoring and uncertain outcomes of compensatory conservation (Bull et al., 2013; zu Ermgassen et al., 2019). Indeed, precautionary multipliers are a determinant of successful terrestrial NNL implementation (zu Ermgassen et al., 2019). Where uncertainty also increases management costs, these costs could be included as part of the price calculation (though they should not be so high as to incentivise non-compliance) (Pascoe et al., 2010).

Finally, equity and fairness should be considered when setting prices, since market-based interventions can have mixed impacts on social welfare and distribution of costs and benefits (Dissou & Siddiqui, 2014; Engel et al., 2008). Potential issues include distributional impacts on fishers who will be directly affected by the levy, as well as wider societal impacts on downstream users and beneficiaries. For example, bycatch levies may be unfeasible and unethical in small-scale fisheries in the Global South, where there is limited capacity to collect levies, and limited resource and adaptive capacity to pay or change behaviour (Bene, 2006; Golden et al., 2016). Rather, a progressive approach could support distributive justice, with levies paid by larger-scale commercial fisheries or downstream high-end consumers, and invested in bycatch mitigation protocols in SSFs. For example, in the US California drift gillnet fishery, a voluntary turtle bycatch levy funded sea turtle bycatch mitigation in Mexican SSFs (Janisse et al., 2010) (Section 10.3.6). In-kind payments, such as volunteering time towards species restoration, may also be feasible in SSFs (Arlidge et al., 2020), while bank-backed insurance schemes can help to smooth financial risks where individual vessels cannot cover costs of rare, stochastic bycatch events (Lent & Squires, 2017; R. Zhou & Segerson, 2016). Fairness can be improved by adopting open and inclusive processes for designing levies,

which enable process justice. This can ensure rights for all participants, identify distributional concerns, and also help to improve uptake and compliance through building social legitimacy (Levi et al., 2009; Oyanedel, Gelcich, & Milner-Gulland, 2020).

10.3.3 Investing revenues

Potential beneficiaries of levy revenues are those who can safeguard or restore bycatch-affected populations, including fishers and those who have property rights (i.e., legal or de facto control or ownership) over critical habitat or bycatch-affected populations. Revenues could be distributed over multiple scales (Table 10.2). For example, levies could be fiscally-neutral at fishery or national levels, with poorly-performing firms paying penalties, and well-performing firms receiving incentives. This could be suitable when bycatch is heterogenous within or across fleets, by incentivising innovation within fleets, or discouraging the most bycatch-intensive technologies. Alternatively, revenues could be distributed internationally, with wealthy countries investing in habitat conservation, technological change for bycatch reduction, and performance-based catch mitigation elsewhere (Dutton & Squires, 2008; Pakiding et al., 2020) (Table 10.2). This can also contribute to greater equity, with revenues from larger commercial fleets helping to finance improvements in low-income SSFs. Inter-temporal equity could also be accounted for by wealthy nations with historically large fisheries impacts committing to net gains for marine biodiversity under the CBD's post-2020 framework (CBD, 2018), and by considering future generations' preferences for biodiversity (i.e., accounting for inter-generational equity).

Table 10.2 Potential scales of distribution and beneficiaries for investing bycatch levy revenues

Scale of levy distribution	Payer	Potential beneficiaries	Suitable situations	Pathway to conservation impact
Local (e.g., fleet-level)	Individual vessels with high bycatch impact	Individual vessels with low bycatch	Bycatch is heterogenous within fleets, predictable and avoidable (i.e., varies within a fleet based on skipper skill)	Preventative through incentivising behaviour change within a fleet. Could achieve No Net Loss against 2020 baselines.
National or sub-national (e.g., jurisdiction, stock or fisheries management area)	Fleets or companies with high bycatch impact	Fleets or companies operating in the same geography which demonstrate low or reduced bycatch; owners/managers of critical habitat or parts of the bycatch affected stock	Bycatch is homogenous within fleets, but heterogenous across fleets (e.g., predictable and avoidable based on broad-scale operational and biophysical characteristics). Could be progressive by charging commercial high-revenue fleets for bycatch, and investing funds in fleet-wide bycatch mitigation in small-scale fisheries, or in performance-based rewards for improving the status of bycatch affected populations.	Preventative through incentivising behaviour change across fleets (i.e., rewards less damaging fishing practices); Compensatory through paying for habitat or population conservation. Could achieve Net Gain against 2020 baselines.
International (e.g., between countries)	Countries or companies with high	Fleets or companies operating	Bycatch is homogenous within fleets, but unavoidable due to tight coupling with target catch	Compensatory through paying for habitat or population

bycatch impact	elsewhere which demonstrate low or reduced bycatch; owners/managers of critical habitat or populations of bycatch-affected species elsewhere	production; bycatch is stochastic and unavoidable. Could be progressive by charging high-revenue commercial fleets for their bycatch in Global North countries, and investing funds in bycatch mitigation or conservation actions to improve the status of bycatch affected populations in Global South countries.	conservation (Dutton & Squires, 2008), or performance-based catch mitigation elsewhere. Could achieve Net Gain against 2020 baselines.
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10.3.4 Monitoring, compliance and adaptive management

Conditionality and additionality are key features of incentive-based approaches. Conditionality requires that incentives are contingent on verifiable performance, and additionality requires that an action would not have occurred otherwise. Therefore, robust monitoring and compliance management are essential. This is challenging in fisheries, due to high monitoring costs for marine biodiversity and fisheries (Arias et al., 2016; Ban et al., 2011; Berkes et al., 2006). Since there are trade-offs between incentives and management costs (Table 10.1) (Segerson, 2011), monitoring and enforcement capacity should be considered in the design phase. For fisheries with high compliance and observer coverage, it may be possible to link levies to performance, whereas process-based levies which can be determined through third-party monitoring (e.g., GlobalFishingWatch) may be more appropriate under poor observer coverage. Advances in satellite monitoring, electronic on-vessel monitoring and machine learning will reduce costs and uncertainties in bycatch monitoring in the coming decade (Bartholomew et al., 2018; Mangi et al., 2015). Furthermore, advances in DNA metabarcoding can also enable forensic monitoring, e.g., using eDNA to reconstruct catch through hold sampling (Cardenosa et al., 2019; Harper et al., 2019; Russo et al., 2020). In the interim, peer-monitoring and placing the burden of proof on firms could reduce monitoring costs (Kotchen & Segerson, 2020).

Monitoring and enforcement are also important when investing revenues, to prevent leakage (i.e., where securing biodiversity in one location leads to the loss or degradation of biodiversity elsewhere), and ensure additionality, equivalence and permanence of biodiversity gains (Bull et al., 2013; Engel et al., 2008). Projects will require good governance and third-party monitoring and verification to ensure that these conditions are met and positive outcomes for biodiversity are delivered (zu Ermgassen et al., 2021). I found little evidence of impact assessment or adaptive management in real-world applications of bycatch levies (Section 10.3.6). Data on fisher behaviour, bycatch mortality and the status of bycatch-affected populations are needed to inform adaptive management and understand if and how bycatch levies can deliver at least NNL. Socio-economic impacts of bycatch levies on affected people and downstream users also need to be understood, to prevent perverse social outcomes (Dissou & Siddiqui, 2014). This could allow for adaptive management of levy designs, to ensure they are effective, progressive, and deliver at least NNL for people and biodiversity (V. F. Griffiths et al., 2019).

10.3.5 Levies as part of a policy mix

Finally, it will be important to consider bycatch levies as one potential instrument within an overall policy mix for addressing bycatch, which can be used in conjunction with other complementary regulatory and market-based instruments (as opposed to an ‘either-or’ dichotomy) (Engel et al., 2008). For example, governments may wish to mandate the use of certain bycatch-reducing technologies (BRTs), while also using bycatch levies to incentivise additional reductions in mortality, or provide compensation for residual mortality over and above that which can be reduced through BRTs. This may be particularly important for highly-threatened and/or low-fecundity species, where compensatory mitigation strategies alone could be particularly risky (Finkelstein et al., 2008). For such species, bycatch levies could be used as the final step in a precautionary mitigation hierarchy of actions, where avoidance, minimisation and remediation measures (such as time-area closures and input standards) are mandated, and bycatch levies are used to further incentivise their adoption and compensate for residual unavoidable mortality (Chapter 3; Arlidge et al., 2020; Booth, Squires, & Milner-Gulland, 2019a; Milner-Gulland et al., 2018; Squires & Garcia, 2018). Similarly, other market-based mechanisms can be used in conjunction with bycatch levies, such as cap-and-trade or performance-based incentives, where levy revenues are re-invested in subsidising socially-underprovided bycatch-minimising technologies (Segerson, 2011) (Table 10.3).

10.3.6 Real-world examples

There are few real-world examples where bycatch levies have been implemented in fisheries (Table 10.3). In Namibia, a mandatory bycatch fee is applied to prevent fishers from targeting species they do not have quota for, which is set at a rate that is higher than the quota fee. In 2012/3, over NAD 6 million (~US\$400,000) was levied in bycatch fees, representing 4.5% of total government revenues from fisheries that year (MFMR, 2013; Pascoe et al., 2010). In the USA, there are two examples of flat-rate process-based voluntary levies, which have been paid by groups of swordfish and tuna fishers, as compensatory funding for sea turtle conservation in nesting habitats and small-scale fishers (Janisse et al., 2010; Pakiding et al., 2020; Squires et al., 2018). These levies do not provide incentives for innovation or bycatch reduction, though it is estimated that they have supported significantly more turtle conservation per dollar cost than at-sea mitigation measures such as gear changes, effort restrictions, and spatio-temporal closures (Gjertsen et al., 2014; Squires & Garcia, 2018). In New Zealand, a ‘deemed value’ is used to reduce management-induced discards (i.e., discards of over-quota catches). In this system, a mandatory charge is levied against landings for which fishers do not have sufficient quota. The deemed value rates are set to disincentivise discarding at sea, while not incentivising targeting of fish for which the fishers do not have quotas. Fishers are allowed to land and sell over-quota catch, but pay a fee per kilogram, which varies by stock and increases with amount caught in excess of quota (Pascoe et al., 2010; Sanchirico et al., 2006; Walker & Townsend, 2008).

Table 10.3 Examples of real-world applications of bycatch levies

Fishery/entity	Target species	Fishing area	Bycaught species	Management objective	Mechanism	Preventative measures	Compliance management	Amt levied per annum	Scale of tax distribution	Refs
International Seafood Sustainability Foundation (ISSF), US-based tuna processors	Tuna	Worldwide	Sea turtles	Yield revenue to finance conservation, reputational (enhance environmental standing in consumer markets)	Voluntary, industry-led – flat rate tax per ton of longline-caught tuna.	Gear standards/modifications, live release protocols	N/A – flat rate payment for voluntary participating companies	US\$100,000 (2016)	International – finances sea turtle conservation in nesting habitats and small-scale fisheries in Pacific, Indian and Atlantic Oceans.	(ISSF, 2016; Pakiding et al., 2020; Squires & Garcia, 2018)
Namibian large pelagic fishery (longline)	Swordfish, blue & short-fin mako sharks	Pelagic offshore. South Atlantic Ocean (Namibian EEZ and high seas).	Tuna, thresher shark, marlin	To prevent unlicensed catch of species that are not included in quota.	Mandatory bycatch fee levied for non-target species	Unclear	Unclear	US\$400,000 (2012/3)	Unclear, likely fiscally-neutral	(MFMR, 2013; Pascoe et al., 2010)
New Zealand multi-species commercial fisheries	Multi-species	Several quota management areas within EEZ.	93 species included in quota management system	Reduce discards of over-quota Bycatch (i.e., management-induced discards)	Mandatory individual bycatch quotas, with bycatch levy through deemed-value system.	Quota	Dual reporting system for fishers and purchasers to fill forms matching catches to permits. Some observer coverage	Unclear	Unclear, likely fiscally-neutral	(Pascoe et al., 2010; Sanchirico et al., 2006; Walker & Townsend, 2008)
California swordfish (drift gillnet) / the Federation of Independent Seafood Harvesters (FISH)	Swordfish	Pelagic offshore. Pacific Ocean (Along East coast, between state waters and EEZ).	Sea turtles leatherback (<i>Dermochelys coriacea</i>)		Voluntary, industry-led – donations, not linked to level of bycatch	Spatio-temporal closures	N/A voluntary donations	US\$71,000 (2004)	International – Invested in turtle nesting habitat conservation in Baja California, Mexico. Impact unclear.	(Janisse et al., 2010; Squires & Garcia, 2018)

Aside from these real-world applications, the hypothetical benefits of bycatch levies have been explored and quantified for turtles, marine mammals and seabirds in several case studies (Gjertsen et al., 2014; Lent & Squires, 2017; Pascoe et al., 2011; Sanchirico, 2005; Wilcox & Donlan, 2007). Various models predicting the impacts of bycatch levies on fisher behaviour and discards in general have also been developed (Androkovich & Stollery, 1994; Herrera, 2005; Mukherjee, 2016; R. Singh & Weninger, 2009).

To my knowledge, the social and biological impacts of the real-world examples given here have not been assessed. However, early examples from terrestrial offsetting indicate that such policies can incentivise prevention of environmental damage, and deliver positive ecological outcomes, provided there are appropriate risk multipliers when environmental levies are priced, and good governance to ensure compensatory conservation outcomes are delivered (Pascoe et al., 2019; zu Ermgassen et al., 2019, 2021). In the future, it will be important to understand how fisher behaviour, bycatch-affected species and welfare and distribution are impacted by bycatch levies, to learn lessons for future applications, and understand whether and how they deliver positive outcomes for biodiversity and people in practice.

10.4 Discussion

I have examined the potential role of voluntary and mandatory bycatch levies in delivering net outcomes for marine biodiversity. If well-designed, and implemented as part of an overall policy mix, bycatch levies could be a useful instrument for reducing bycatch during at-sea fishing operations and generating sustainable financing for compensatory conservation investments. However, mainstreaming bycatch levies remains dependent on firms' willingness-to-pay (WTP) levies and society's willingness-to-accept (WTA) compensation for damaged to nature.

Since biodiversity offsets are increasingly mandated by governments and investors for terrestrial and coastal developments (Bull & Strange, 2018; Shumway et al., 2018), it is perhaps only a matter of time until compliance markets for bycatch levies and offsets also develop. Governmental implementation of bycatch levies could also be further stimulated via a net outcome approach to the CBD post-2020 framework (CBD, 2020a; Milner-Gulland, Addison, et al., 2020). In the interim, voluntary WTP could be instigated through several avenues. Transparency and advocacy initiatives, to educate consumers and the public about bycatch issues, may increase firms' reputational risks of not addressing bycatch (Ruysschaert & Salles, 2018; Schouten & Glasbergen, 2011), thus creating a 'strategic threat' as an incentive for voluntary participation (Segerson, 2013). For firms which are already under significant pressure to reduce bycatch, and may face high opportunity costs of direct regulation, such as complete fishery closures (e.g., the New England lobster pot fishery, which is under high public scrutiny due to whale entanglements (Greater Atlantic Regional Fisheries Office, 2019)), adopting levies and investing in offsets could improve firms' reputations and allow them to continue operating (Janisse et al., 2010; Squires et al., 2018). In turn, this could increase profits, economic welfare and revenues for compensatory conservation through levies

on the additional economic surplus that is created. Bycatch levies could also create opportunities for improved access to investors, market share and prices (e.g., via eco-labels, exploiting WTP among seafood consumers), and thus positive incentives for voluntary participation. For example, seafood certification and corporate ocean stewardship schemes could introduce NNL or net gain policies as part of their standards; while the financial sector offers leverage points through growing investor expectations regarding sustainability (Jouffray et al., 2019; Österblom et al., 2017). Just as investors and consumers are demanding ‘deforestation-free’ terrestrial supply chains (CDP, 2014; Donofrio et al., 2017), so they could demand ‘bycatch-neutral’ seafood supply chains.

Given that just thirteen transnational companies control 19–40% of the largest and most valuable fish stocks (Österblom et al., 2015), adoption of voluntary bycatch levies by a coalition of ‘keystone actors’ could raise substantial revenue for marine conservation. For instance, the end value alone of the 7 most commercially-important tuna species is over US\$ 42 billion (Galland et al., 2016). If bycatch levies were valued at 12% of this figure (a reasonable amount given estimated global discard rates (Davies et al., 2009)), they could raise sufficient revenue to effectively protect 20–30% of the world’s oceans (Balmford et al., 2004), in turn creating additional long-term benefits for biodiversity, food security and climate

For bycatch levies to deliver net positive outcomes, a supply of investment-ready marine conservation projects is needed, which can deliver measurable and additional catch mitigation or population increases for a given cost (e.g., Chapter 8). This is so that conservation gains for bycatch-affected populations and species can be quantitatively balanced against the losses caused by bycatch, as per biodiversity offsets in terrestrial systems (Bull et al., 2013; Jacob et al., 2020). However, there are few examples of such schemes, and the cost-effectiveness of marine conservation activities, in terms of biodiversity saved per dollar spent, is poorly understood. NGOs and governments could help to supply projects, in partnership with local communities, though they must be able to demonstrate additional conservation impacts and quantify the costs involved (Bladon et al., 2016; Bull et al., 2013; Engel & Palmer, 2008). For marine biodiversity markets to develop at scale, third-party certification may be required, with intermediaries to match revenues with beneficiary projects, and external auditing to ensure funds are appropriately invested (e.g., as per lessons from conservation trust funds and REDD+ (Angelsen, 2009; Bonham et al., 2014; Spergel & Mikitin, 2013)). Further development of low-cost third-party methods for monitoring and verifying bycatch performance in fisheries (e.g., video monitoring, satellite monitoring and eDNA) and outcomes of marine conservation projects will be needed to reduce uncertainties and management costs in the future (Bladon et al., 2016).

If successfully mainstreamed, wide adoption of bycatch levies could move fisheries towards more socially-optimal outcomes for people and nature, by incentivising bycatch prevention, raising substantial revenue for conservation, and enhancing economic welfare. If paired with effective, additional marine

biodiversity offsetting, this could help governments, companies and individuals contribute towards NNL or net gain goals for biodiversity under the CBD's post-2020 framework, and transform current trade-offs between fisheries and conservation into win-wins for blue growth and biodiversity.

11. Discussion



Photo: Me chatting to a shark fisher in Tanjung Luar.

“Conservation is primarily not about biology but about people and the choices they make” – Balmford & Cowling.

“We have far more in common than that which divides us” – Jo Cox.

11.1 Summary of novel contributions

This thesis aimed to apply interdisciplinary methods to contribute to more effective design of shark and ray conservation interventions, which can simultaneously deliver conservation outcomes for sharks and well-being outcomes for people.

To achieve this, I critically assessed current approaches for understanding and managing threats to sharks, and highlighted gaps relating to understanding and addressing socio-economic issues (Objective 1, Chapter 2). Moreover, I showed that filling these gaps is important for both practical and ethical reasons. Practically – considering socio-economic issues can make shark conservation more effective, by addressing the root causes of overfishing and effectively changing behaviour. Ethically – conservationists have a moral duty to consider the impacts of conservation on people, and ensure that we ‘do no harm’ to the most vulnerable (Booth, Squires, & Milner-Gulland, 2019b). Based on this, I proposed that socio-economic feasibility assessments be incorporated into traditional fishery risk assessments for sharks (Chapter 2). I then outlined how the mitigation hierarchy could meet this need, providing a novel decision-making framework for managing trade-offs between shark conservation and fisheries objectives (Objective 2, Chapter 3) (Booth, Squires, & Milner-Gulland, 2019a). I demonstrated how the mitigation hierarchy could help to integrate socio-economic issues into fisheries management planning, support identification of effective and feasible management measures for shark conservation across multiple contexts and management-relevant scales, and allow for differentiated pathways towards overarching outcome goals, such as no net loss or net gain. As a side project to this thesis I also worked with colleagues to apply the mitigation hierarchy to managing shark bycatch in trawlers in India (Gupta et al., 2020).

In Chapters 4 and 5, I focused on investigating socio-economic drivers of shark and ray fishing, and barriers to pro-conservation behaviour (Objective 3), as well as the effectiveness of existing interventions to address these drivers and barriers (Objective 4). For example, in Chapter 4 I used a Theory of Change to outline some of the key socio-economic drivers of manta ray fishing in Lamakera, and conservation strategies to address them. I then assessed the impact of an integrated conservation intervention, which aimed to stop manta hunting through targeting multiple drivers of hunting behaviour using a combination of intrinsic and extrinsic and positive and negative levers. The findings indicated that a multifaceted approach – which included community outreach and livelihood-focused incentives, to reduce barriers and create positive motivations for compliance, and law enforcement, to create perceived net negative incentives for non-compliance – successfully reduced manta ray mortality relative to baselines and counterfactuals. Key lessons for designing future interventions were also outlined, including: a) understand diverse drivers of human behaviour; b) adopt data-driven problem-oriented planning; c) continuously document and share learning; d) establish partnerships with diverse stakeholders to develop resilient institutions for enduring impact (Booth, Mardhiah, et al., 2020). In Chapter 5, I outlined a socio-

psychological approach for understanding and managing the socio-economic drivers of shark bycatch and barriers to bycatch mitigation, and applied it to the Lhok Rigaih gill net fishery. This chapter demonstrated the utility of considering bycatch as a spectrum rather than a clearly delineated component of catch, where the position of a species on this spectrum depends on fishers' beliefs regarding the outcomes of bycatch-relevant behaviour. This spectrum, combined with a socio-psychological approach for codifying fisher beliefs, can help to identify conflicts and synergies between bycatch mitigation and fishers' beliefs, thus informing more effective and socially-just interventions for marine megafauna conservation.

In chapters 6-8 I moved towards using predictive methods, to explore how hypothetical conservation interventions could impact shark conservation and human well-being in small-scale fisheries, and outline design options for cost-effective future interventions (Objective 5). For example, in Chapter 6 I used Boosted Regression Trees to predict the hypothetical cost-effectiveness of different input-oriented fisheries management measures in Tanjung Luar, based on the estimated conservation benefit (as reduced risk of capture of threatened and CITES-listed species) and socio-economic cost (as relative profit forgone). In Chapter 7, I applied hedonic price analysis to estimate the hypothetical economic opportunity costs of several output-oriented fisheries management measures for fishers Tanjung Luar (such as catch limits and retention bans for several threatened and CITES-listed sharks). Both chapters highlighted difficult trade-offs between conservation goals and the economic welfare of small-scale fishers, as well as trade-offs between taxa. The results indicate that in such cases, economic interventions – such as compensation schemes or large-scale fishery buy-outs – may be required to deliver shark conservation goals whilst doing no harm to vulnerable coastal communities. In addition, hard choices and triage may be required, so that slow-growing critically taxa can be prioritised for protection, while faster-growing taxa can continue to provide socio-economic benefits for coastal communities. Building on these findings I then explored the hypothetical effectiveness of incentive-based approaches to shark conservation (Chapter 8). Specifically, I used scenario interviews with contingent valuation to investigate how performance-based compensatory payments could incentivise pro-conservation behaviour for Critically Endangered species, and reconcile trade-offs between conservation and well-being. The results indicate that such payments could be cost-effective and widely accepted by fishers.

Finally, in Chapters 9 and 10, I explored two potential financing mechanisms for shark conservation in small-scale fisheries (Objective 6). In Chapter 9, I focused on operationalising the beneficiary-pays principle, wherein marine tourists who value healthy shark populations pay towards shark conservation. In Chapter 10, I applied the polluter-pays principle, wherein large-scale commercial fisheries and/or seafood companies pay bycatch levies – as a form of double-dividend Pigouvian tax – to incentivise bycatch reduction and compensate for the negative impacts of bycatch in their supply chains. In both cases, these revenues could be used to pay for shark conservation outcomes in small-scale fisheries and

coastal communities, and could generate revenues which are orders of magnitude higher than those required to fund performance-based compensatory payments in my case study sites.

Overall, this thesis has achieved the aim and objectives outlined in Section 1.3. I have provided novel insights on using interdisciplinary approaches for shark conservation, and how they could be incorporated into risk assessments and decision-making frameworks. I have collected empirical data on the socio-economic drivers of shark fishing, and assessed an integrated conservation intervention aimed at addressing them, both of which can be used to inform the design of future conservation intervention. I have used predictive methods to explore how hypothetical regulatory and incentive-based interventions could impact shark conservation and human well-being, and used economic methods to estimate their costs and investigate long-term revenue streams. These findings can be used to inform the design of cost-effective shark conservation interventions – in the study fisheries, as well as other similar SSFs elsewhere – which can simultaneously deliver conservation outcomes for sharks and well-being outcomes for people.

In addition, in delivering these research aims, I have developed and applied several novel methodological approaches. For example: In Chapter 3, I offer the mitigation hierarchy as a novel decision-making framework for shark conservation. In Chapter 4, I provide an example of a robust non-experimental impact assessment, using a theory-based mixed-methods research design. While this is not the first application of theory-based impact assessments in conservation, it represents one of the first examples of using counterfactual thinking to provide evidence for effective approaches to shark conservation in small-scale fisheries, and could be applied to similar interventions where experimental designs are unfeasible. In Chapter 5, I suggest a novel socio-psychological approach for understanding and managing bycatch in SSFs, drawing on the Theory of Planned Behaviour. In Chapter 7, I demonstrate a novel application of hedonic price analysis to wildlife trade. In Chapter 8, I develop and apply a novel combination of scenario interviews with contingent valuation for designing locally-appropriate conservation incentives. All these approaches could be applied to other conservation science problems, including in shark fisheries, and for other taxa and ecosystems.

11.2 Cross-cutting themes

Several common themes have emerged throughout the chapters of this thesis, which have relevance across and beyond the study sites and taxa. In this section I discuss these themes and their implications for conservation science and practice.

11.2.1 Understanding and managing trade-offs

Complex trade-offs and hard choices are ubiquitous in conservation science (Hirsch et al., 2011; McShane et al., 2011), and have been a prominent theme throughout this thesis. In general, they exist between

conservation and human well-being objectives (Hirsch et al., 2011; McShane et al., 2011), as well as within conservation and well-being agendas, which are themselves embodied by a plurality of objectives, values and stakeholders across different contexts and scales (e.g., cross-taxa conflicts, displacement effects, distributive injustice, inter-generational inequality (Daw et al., 2015; Gilman et al., 2019; Law et al., 2018; G. G. Singh et al., 2018; Sumaila, 2004)). Trade-offs also occur between knowing and doing in conservation planning (Knight & Cowling, 2010). This thesis demonstrates examples of all of these types of trade-offs in shark conservation, and offers some approaches for understanding and managing them.

Outcome trade-offs

The clearest direct trade-offs in shark conservation are those between conservation objectives and economic welfare obtained from fisheries. For example, Chapter 7 quantified some of these trade-offs, by estimating the economic opportunity costs of reducing catches of threatened and CITES-listed sharks in Tanjung Luar. Chapter 5 also shows that small-scale fishers believe they would experience well-being trade-offs in reducing bycatch of Critically Endangered sharks, because they provide non-trivial contributions to their food security and income. Other studies – in Indonesia and other Global South nations – have highlighted similar trade-offs, where shark conservation actions impinge on the rights and well-being of impoverished coastal communities (Castellanos-Galindo et al., 2021; Collins et al., 2020; Jaiteh et al., 2016; Jaiteh, Loneragan, et al., 2017). This also contributes to a broader literature on socio-economic trade-offs in marine conservation in general (though most studies have focused in particular on spatial closures/MPAs, as opposed to taxa-specific policies (Gill et al., 2019; J. T. Watson et al., 2009)).

Chapter 6 highlights within-conservation trade-offs, with cross-taxa conflicts in fisheries management options, such that approaches which reduce risk to some threatened species increase risk to others, and vice-versa. Similarly, Chapter 4 highlighted a displacement effect, wherein protection of manta rays has led to increased capture of devil rays as the closest available substitute. These cross-taxa conflicts and unintended displacement effects are also documented for other fisheries management and marine conservation interventions (Agardy et al., 2011; Gilman et al., 2019; Suuronen et al., 2010), and underline the value of applying predictive methods in conservation planning to highlight potential unforeseen consequences ahead of implementation (Travers, Selinske, et al., 2019). In cases where shark fishing has an important welfare function, and entire fishery closures or buy-outs are unfeasible, managing small-scale shark fisheries for multiple outcomes may require conservation triage (Hayward & Castley, 2018) so that slow-growing Critically Endangered taxa can be prioritised for protection while less threatened faster-growing taxa can continue to provide socio-economic benefits for coastal communities.

Trade-offs between different components of well-being, as well as the distribution of costs and benefits across space and time, are also highlighted. For example, direct consumptive use of sharks, as a source of food and income, is currently degrading the indirect and existence values of sharks (e.g., supporting

nutrient cycling, regulating trophic cascades, recreation and tourism (MacNeil et al., 2020; Mustika et al., 2020; Roff et al., 2016)). This also creates trade-offs between short- and long-term benefits (i.e., inter-temporal equity) and between different places and groups of people (i.e., distributional equity). For example, many of the species landed and sold in the study fisheries are being exploited at unsustainable levels, therefore the benefits they provide to ecosystems and people are diminished for other people and future generations. More broadly, while the case study fisheries in this thesis are contributing to overall patterns of decline, the impacts of bycatch in large scale commercial fleets, both now and in the past, are likely to be orders or magnitude larger than the impacts of these small-scale fisheries (Oliver et al., 2015). Similarly, those who value healthy populations of sharks and rays, either for their existence values or because they profit from shark-based tourism, will benefit from conservation measures, whilst fishers will bear the costs (Mustika et al., 2020). This raises an on-going question about who should pay for the cost of conservation (Balmford & Whitten, 2003), with a need to acknowledge and balance inequities to ensure that marine conservation practice is socially just. However, doing so also raises further questions regarding which and whose values should be used when seeking to balance these inequities – i.e., how to reconcile differences between global values for sharks (e.g., global trade, global tourism), industrial bycatch values, and local values expressed by fishers and by tourists; and indeed, how to practically measure and operationalise equity in conservation decision-making and marine policy. With calls for social equity as a core tenet of the blue economy (N. J. Bennett et al., 2019), operationalising equity, and reconciling equity with economic efficiency, represent priorities for future applied research.

Managing trade-offs

All of these trade-offs represent forms of conservation conflicts, i.e., “*situations that occur when two or more parties with strongly held opinions clash over conservation objectives and when one party is perceived to assert its interests at the expense of another*” (Redpath et al., 2013). Addressing them requires an ethical code of conduct for shark conservation, to ensure it is morally responsible and grounded in principles of human rights. “*We need to not only to build on common interests between conservationists and local communities wherever these occur, but also to engage in honest discussion about genuine conflicts of interest where these exist and work towards negotiated settlements, with full respect for rights as the bottom line.*” (Newing & Perram, 2019). At the very least, shark conservation should ‘do no harm’ to the most vulnerable. It also requires embracing approaches for conflict management – e.g., as is increasingly demonstrated in efforts to promote human-wildlife coexistence in terrestrial landscapes – with integration of ecological and social sciences to map relations and conflicts, coupled with stakeholder processes to decide how to manage them (Iwane et al., 2021; König et al., 2020; Pooley et al., 2021; Redpath et al., 2013).

Such approaches are unlikely to entirely resolve and eliminate shark conservation conflicts, particularly in the face of continued shark population declines and growing pressures on marine resources. Indeed, most conservation conflicts are not eliminated, but rather managed such as to minimise their destructive nature

(Redpath et al., 2013). However (to adapt from literature on coexistence with terrestrial predators), it may be possible to achieve a “*sustainable though dynamic state in which humans and [sharks] co-adapt to sharing [seascapes], where human interactions with [sharks] are effectively governed to ensure populations persist in socially legitimate ways that ensure tolerable risk levels*” (N. H. Carter & Linnell, 2016; König et al., 2020). In shark conservation, this will require mechanisms for balancing cost-benefit inequities.

There are several potential approaches for seeking to balance these inequities at different scales. For example, relatively wealthier ocean stakeholders, who value and benefit from sharks via their existence and indirect uses, could contribute towards meeting the costs of shark conservation for small-scale fishers (Chapter 9). Similarly, large and highly profitable commercial seafood companies, which cause significant harm to shark populations due to bycatch, could compensate for this damage via progressive bycatch levies (Chapter 10). Both funding mechanisms could fund conservation actions in coastal communities and small-scale fisheries that otherwise cannot afford it, following a similar ethos to that of REDD+ and Nationally Determined Contributions under the Paris Agreement. Distributional and inter-temporal equity could also be accounted for by wealthy nations with current and historically large fisheries impacts committing to net gains for marine biodiversity under the CBD’s post-2020 framework (Figure 11.1) (CBD, 2020b), and by considering future generations’ preferences for biodiversity. In parallel, nations which are less economically wealthy but still biodiversity rich, with lower historic biodiversity footprints, could adopt managed net loss of certain elements of biodiversity (e.g., elements which are widespread and of lower conservation concern, and provided that the sum of different countries’ contributions combine to deliver >NNL), with net gain of more threatened elements paid for by other countries (Figure 11.1) (Milner-Gulland, Addison, et al., 2020). For example, managed silky shark fisheries could continue in Indonesia, with protection and restoration of wedgefish populations funded by countries with large historic bycatch footprints throughout their range.

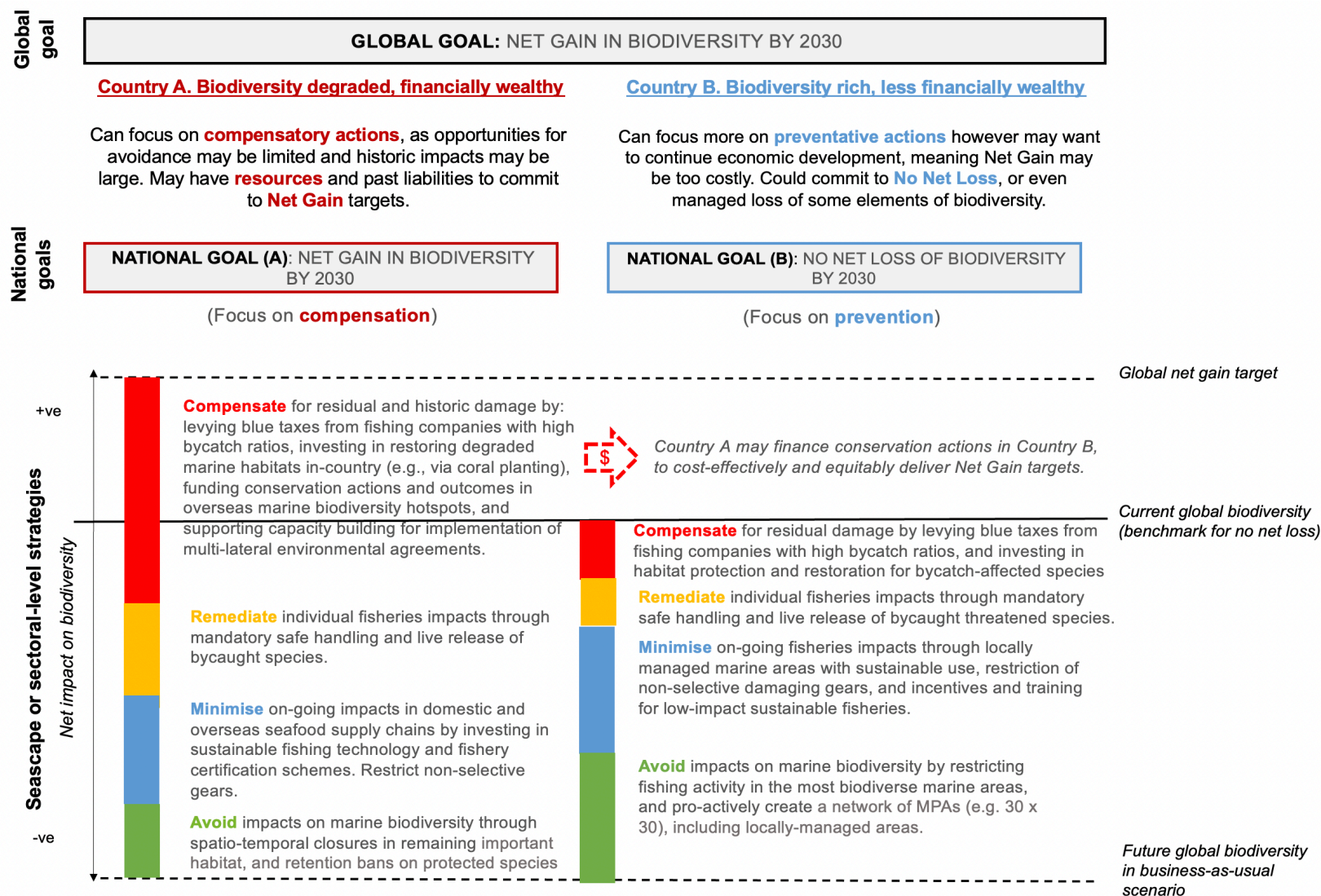


Figure 11.1 A generic example of how different countries can set different national goals for marine biodiversity, which suit their national resource contexts, sum to achieve an overarching global target for net gain in biodiversity, and are economically progressive (modified from the Conservation Hierarchy)

When designing management measures, there are also further trade-offs between knowing and doing (Knight & Cowling, 2010). This creates difficult choices regarding how much information and certainty is enough to act, particularly when (in)action will have real-world implications for wildlife and people, and when the selection and implementation of (in)actions are the product of complex processes that are not entirely science-based (Knight & Cowling, 2007). For example, in the case of bycatch levies, a socially optimal levy price is one at which the marginal private cost of bycatch prevention is equal to its marginal social benefit (Chapter 10). However, calculating optimal prices is often difficult and time consuming. Rather than spending time and effort on ‘knowing’ the optimal price, it may be more practical to implement a scheme using average costs (e.g., of compensatory conservation, or societal WTP), with the aim of improving cost-effectiveness (e.g., Gjertsen et al., 2014), and adopting adaptive management to update prices as new data and experiences are documented. On the other hand, if there is a large downside risk associated with setting the wrong levy price, then investing in additional research could pay-off. Similar trade-offs also occur with respect to goal setting and intervention design. For example, some management goals – such as those which require robust population models (e.g., ‘net gain goals’ as outlined in Chapter 3, Table 3.3); and some management measures – such as those which require detailed and accurate catch data (e.g., individually-targeted bycatch levies, as outlined in Chapter 10, Table 10.1) can only feasibly be adopted in data-rich high-capacity situations. However, goals, management measures and monitoring systems can be adapted to any situation, regardless of data paucity and capacity. For example, even when there is little or no existing data, goals and baselines for implementing a mitigation hierarchy approach could be based on fisher interviews and landings monitoring, with refinement over time as monitoring improves (Chapter 3, Table 3.3). Chapter 4 demonstrated the utility of this approach, where on-going and increasingly sophisticated monitoring, coupled with adaptive management, helped to develop an increasingly robust intervention and patrolling strategy, with increasingly successful outcomes over time.

Given the urgency of the biodiversity crisis, conservation scientists should not let the perfect be the enemy of the good, with further research potentially displacing or postponing action beyond the point at which it is meaningful (Lindenmayer et al., 2013; Milner-Gulland & Shea, 2017). Ultimately, solutions to knowing-doing trade-offs depend on the triviality of the uncertainty (Milner-Gulland & Shea, 2017), and the downside risk of being wrong relative to a counterfactual of inaction or an alternative action. One option is to conduct Value of information (VoI) analyses, which estimate the marginal payoff of additional research – in terms of the likelihood of a change in a decision and the positive impact of that change – if additional information is provided (Canessa et al., 2015). For example, VoI supported research prioritisation and conservation for whooping cranes in North America, by helping to understand which uncertainties most hampered decision-making (Runge et al., 2011). Another option is to set

(precautionary) burdens of proof at the outset of planning processes, and ensure they are sufficiently met by available evidence before risky decisions are made (Booth, Pooley, et al., 2020).

More broadly, creating ‘fail safe’ cultures within conservation organization could also mitigate against detrimental forms of risk aversion, and support learning and adaptive management (Catalano et al., 2019). Indeed, novel and radical approaches will likely be required to deliver the transformative change needed to avert shark population declines. Such approaches are unlikely to emerge in the absence of sufficient psychological safety to take measured risks under uncertainty (Catalano et al., 2018, 2021). Similarly, there is a need to generate safe and productive spaces in which shark researchers and publics can interact, to support co-production of conservation evidence that is useful for decision-makers, and collaborative decision processes (Christie et al., 2020; Toomey et al., 2017).

11.2.2 Embracing social and behavioural sciences, for more effective and equitable shark conservation

Another key message of this thesis is that better integration of human dimensions into shark conservation science will support more effective and equitable conservation actions. This is not a particularly novel observation - there have been calls to integrate human dimensions into conservation biology and wildlife management for more than 20-years, and it is widely acknowledged that conservation is not primarily about biology, but rather about people and their decisions (Balmford & Cowling, 2006; Jacobson & McDuff, 1998). Yet conservationists often continue to pay insufficient attention to the complexities of human behaviour, with an ongoing need (and opportunity) for applying advances in behavioural science to conservation problems (Balmford et al., 2021; Travers et al., 2021; D. R. Williams et al., 2020).

As this thesis highlights, shark conservation is no exception. Firstly, achieving conservation success for sharks means tackling overfishing. This ultimately requires re-shaping the tactical and strategic behaviours of fishers and fishing firms, and the underlying beliefs and motivations that drive those behaviours. For example, fisher decisions influence the proximate technical causes of shark mortality, and these decisions are in turn influenced by a range of distal socio-economic factors (Chapter 3). These distal factors include fishers’ beliefs and motivations regarding catch-relevant behaviour (e.g., Chapters 4 and 5). Secondly, as well as the practical imperative of delivering better conservation outcomes, there is also a moral imperative to consider human dimensions, to guard against inequitable costs and trade-offs (e.g., as per the estimated economic losses to small-scale fishers from shark conservation in Chapter 7). Yet, as outlined in Chapter 2, there remains a ‘socio-economic implementation gap’ (Figure 2.2) in shark conservation, with a great deal of research focused on understanding status and trends of shark populations, and relatively little focused on designing, implementing and testing conservation responses. Moving forwards, embracing behavioural sciences in shark conservation could help to better understand threats, design responses, and enhance the overall effectiveness of present and future interventions

(Balmford et al., 2021; D. R. Williams et al., 2020). Moreover, if conducted via collaborative processes, and within appropriate sites and spatial scales, such research could provide well-grounded and contextually rich evidence, with high relevance to decision-makers (Christie et al., 2020; Wyborn & Evans, 2021).

Based on the principle that conservation science should be impact-focused, Williams et al. (2020) recently proposed a research framework suggesting how conservation research might progress if it is to address real-world problems. It includes diagnosing mechanisms and drivers that cause direct threats to species, and then conducting research on proposing, designing, implementing, and testing responses to them. This framework could be adopted for guiding impact-focused shark conservation science in the future, with methods from social and behavioural sciences being fundamental to understanding drivers and designing responses (Figure 11.2). For example, theories from behavioural sciences can help to diagnose drivers of bycatch-relevant behaviour and barriers to pro-conservation behaviour; and predictive methods can be used to design interventions, and pro-actively explore how responses might alleviate threats, as well as influence other important social and economic goals (Figure 11.2). Integrating these methods into an overall strategy for how shark science informs shark conservation practice could ultimately lead to greater conservation impact.

Chapters 4, 5 and 8 also build on and substantiate existing literature highlighting the need for a cross-disciplinary understanding of unsustainable and illegal natural resource use. When designing behavioural interventions there is a need to consider the motivations, capacities and cognitive biases of individuals; the wider physical, social, economic and institutional context in which they operate; and complex interactions between these factors (e.g., interactions between opportunity, motivation and legitimacy; conflicts and synergies between economic incentives and social norms) (Balmford et al., 2021; Cinner, 2018; Cinner et al., 2021; Gneezy et al., 2011; Grillos et al., 2019; Oyanedel, Gelcich, & Milner-Gulland, 2020; Oyanedel, Gelcich, & Milner-Gulland, 2020). Importantly, different intervention mixes will be required for different places and people. For example, in Tanjung Luar, economic incentives may be the most appropriate and powerful approach for mitigating capture of Critically Endangered species, whereas social, bio-cultural and norms-based approaches could be well-suited to Lhok Rigaih (Chapter 8). This heterogeneity reiterates the value of behavioural sciences in designing interventions that are locally appropriate - addressing context-specific drivers of behaviour and guarding against unintended social and ecological consequences (Travers et al., 2021; Travers, Selinske, et al., 2019). Indeed, rather than the continued search for scalable one-size-fits-all solutions, this thesis points to the value of scaling-up frameworks and methods, which can then be used to understand unsustainable behaviours within their specific contexts, and assess portfolios of management options. This could help to fill gaps in context-specific conservation evidence (Christie et al., 2020, 2021), which inform a patchwork of context-specific responses, with differentiated pathways towards united outcomes.

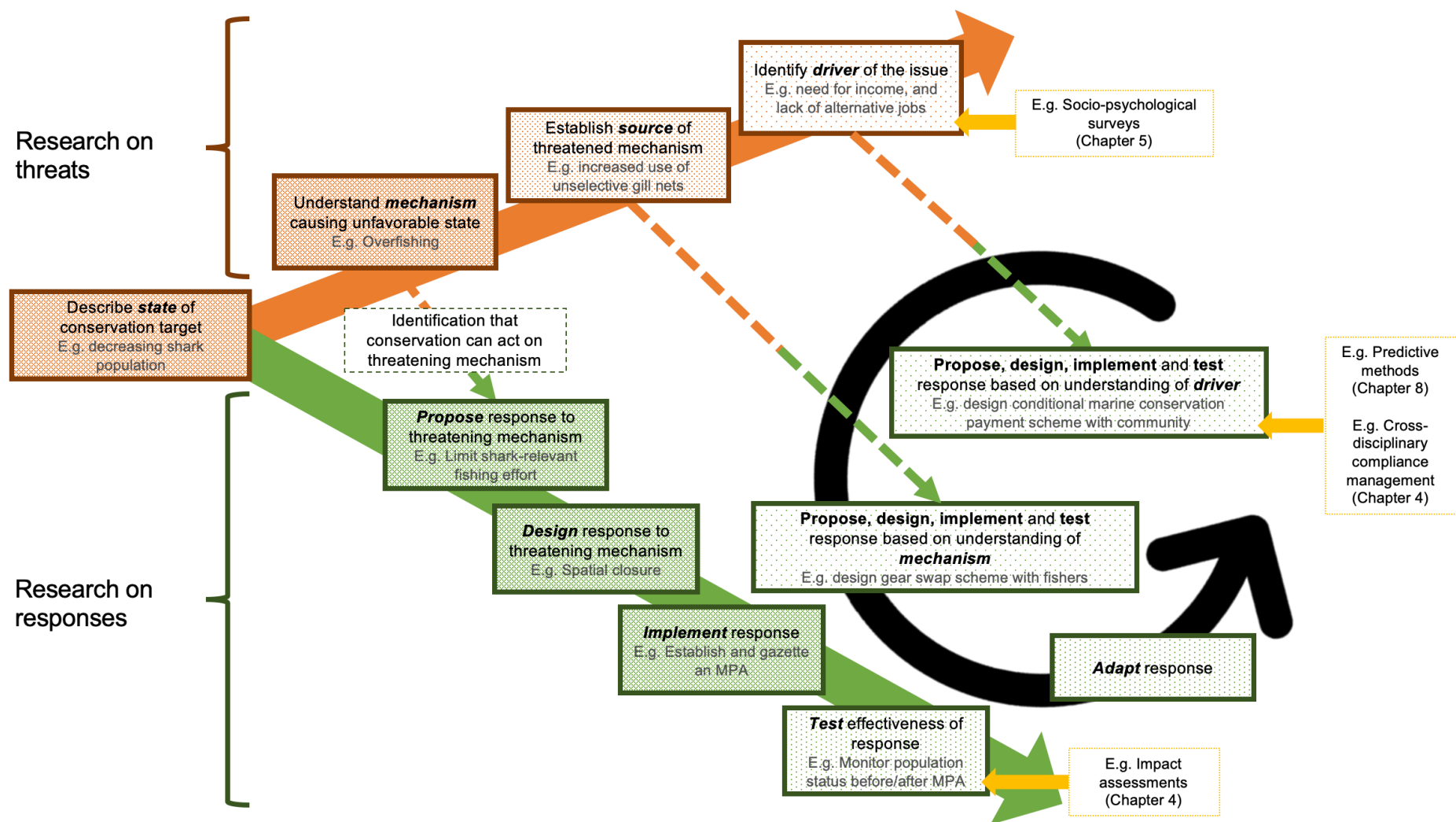


Figure 11.2 A research framework for shark conservation science, adapted from Williams et al. (2020) to include specific examples for sharks, and adding in adaptive management and examples of where social and behavioural research methods are needed in the research process. The shading of the boxes conceptually represents the amount of research already conducted in that domain, with darker shading representing relatively more research effort, and lighter shading representing relatively little research effort.

11.2.3 Transforming the political economy of shark fishing: beyond the fin trade and blanket bans

To achieve transformative change for shark conservation, behavioural change in fisheries also needs to be supported by structural interventions – such as policies, institutions and systems (Balmford et al., 2021; Naito et al., 2021). To date, international trade in shark fins and China's demand for shark fin soup, has monopolised the attention of NGOs, activists and the general public as the greatest systemic threat to sharks (Shiffman et al., 2020; Shiffman & Hammerschlag, 2016a). However, while the high economic value of certain body parts can drive targeted shark fishing (e.g., in Tanjung Luar and Lamakera) and retention of sharks in non-target fisheries (e.g., wedgefish in Lhok Rigaih), an over-focus on the fin trade is arguably based on oversimplified assumptions about the drivers of shark fishing mortality (Chapter 2). This may be to the detriment of more nuanced policy-making and more effective management interventions, with an over-focus on direct regulation and blanket approaches that do not necessarily foster sustainability (Castellanos-Galindo et al., 2021; Shiffman et al., 2020; Shiffman & Hueter, 2017; Tolotti et al., 2015; X. Zhou et al., 2021).

This thesis challenges some common assumptions regarding shark fishing, and demonstrates that it can have numerous causes, which stem from both the micro- and macro-environment. The causes range from accidental bycatch to local socio-cultural drivers, to food security and demand for meat, to high profit from the international fin trade. Moreover, these causes are often interrelated, with integrated markets for different shark-derived products, and complex interactions between local, national and global demand. For instance, targeted manta ray hunting in Lamakera was driven by complex interactions between local traditions and norms, and international demand for mobulid gill plates (Chapter 4). Increasing use of gill nets also further complicates the issue of mobulid catches, with 'accidental bycatch' creating a potential perceived loophole for fishers to continue exploiting mantas. In Lhok Rigaih, capture of wedgefish and hammerheads is primarily due to accidental bycatch in non-selective gill nets, however local demand for shark meat and high prices for wedgefish fins exacerbate the issue, with fishers disincentivised to avoid or minimise their bycatch, and incentivised to retain any individuals they do catch (Chapter 5). Of the three case study sites, the Tanjung Luar fishery may be most strongly linked to/driven by the international fin trade. Even so, my analysis of the relationship between catches and prices in Chapter 7 showed that catch Granger-causes (i.e., predictably forecasts) price, which is indicative of a supply-driven market. There are also barriers and local drivers which create inertia towards continued shark fishing, even in the face of declining fin prices. For example, shark meat is widely consumed in the local community, with any excess traded throughout the rest of West Nusa Tenggara province or exported to other islands. Other commodities - such as skin, cartilage and liver oil – are also used locally and domestically (Prasetyo et al., 2021). Moreover, most fishers also state that they would not leave the fishery, even if their catches declined by half, primarily due to a lack of alternative skills, capital and equipment (Lestari et al., 2017;

Milner-Gulland, Ibbett, et al., 2020). Further, anecdotal observations during the COVID-19 pandemic suggest that even large shifts in international market dynamics did not change local fisher behaviour. For example, some traders anecdotally reported challenges regarding onward sales and exports, due to closures of traditional trading routes, yet shark fishing continued unabated in Tanjung Luar. Similar patterns have also been observed following export bans of certain species. For example, while it has been technically illegal to export hammerhead shark fins from Indonesia since 2014, hammerhead shark landings have continued in Tanjung Luar (WCS-IP, 2019). More broadly, shark fishing in Indonesia also occurs within the context of an ocean-dependent and growing population, and declining fish stocks (Golden et al., 2016; Selig et al., 2018). Therefore, there may be few opportunities for promoting legal, sustainable and equally-profitable marine-based alternatives to current fishing practices, in the absence of major technological innovation or market reform. This points to the need for new institutions, adaptive capacities and resource streams, to better align coastal economies with the health of the ocean.

On the demand side, there is evidence that shark-fin sales and prices have declined in China and Hong Kong during the past decade (Ho Ka Yan & Shea Kwok Ho, 2015; X. Zhou et al., 2021). In parallel, there is growing regulation of international shark trade due to CITES-listings, and enforcement thereof (Booth, Pooley, et al., 2020; Lo, 2021). Governments and the private sector are also removing shark fins from menus, official banquets and cargo shipments (Cripps, 2013; Ho Ka Yan & Shea Kwok Ho, 2015; Watts, 2012; X. Zhou et al., 2021). Despite these patterns, there remains little evidence of associated declines in global shark mortality, or progress towards sustainable fisheries (Davidson et al., 2016). At the same time, the size and value of international shark meat trade, and the role of Western economies in this trade, is growing in prominence (Niedermüller et al., 2021). In parallel, it is increasingly acknowledged that incidental catch in non-target fisheries (whether accidental or as high-value secondary catch) is the greatest source of shark fishing mortality globally (Oliver et al., 2015; Worm et al., 2013). Moreover, just thirteen multi-national corporations control 11-16% of global marine catch, and 19-40% of the largest and most valuable stocks (Österblom et al., 2015). This means there is a need to move beyond piecemeal approaches to shark conservation, which focus on specific issues (e.g., shark fin soup) and approaches (e.g., trade bans), towards a more systems-based approach which situates shark conservation within the broader context of globalisation and telecouplings with global supply chains and food systems (Díaz et al., 2019; Halpern et al., 2019).

Moving forwards, more careful critique of prevailing assumptions about the drivers of shark fishing, and more careful diagnosis of social structural problems, could help to target and design well-informed structural interventions, that also work synergistically with behavioural interventions in fisheries (Hinsley & 't Sas-Rolfes, 2020; Naito et al., 2021). For example, Naito et al. (2021) recently proposed an integrative framework for pro-environmental social change, which includes parallel processes of behavioural and structural interventions. This framework could be applied to structuring a patchwork of interventions for

shark conservation, which operate across micro- and macro-economic scales, thus combining context-specific behavioural approaches at the local level (as discussed in Section 11.2.2) with broader structural change, to create parallel transformations in behaviours and social structures (Figure 11.3).

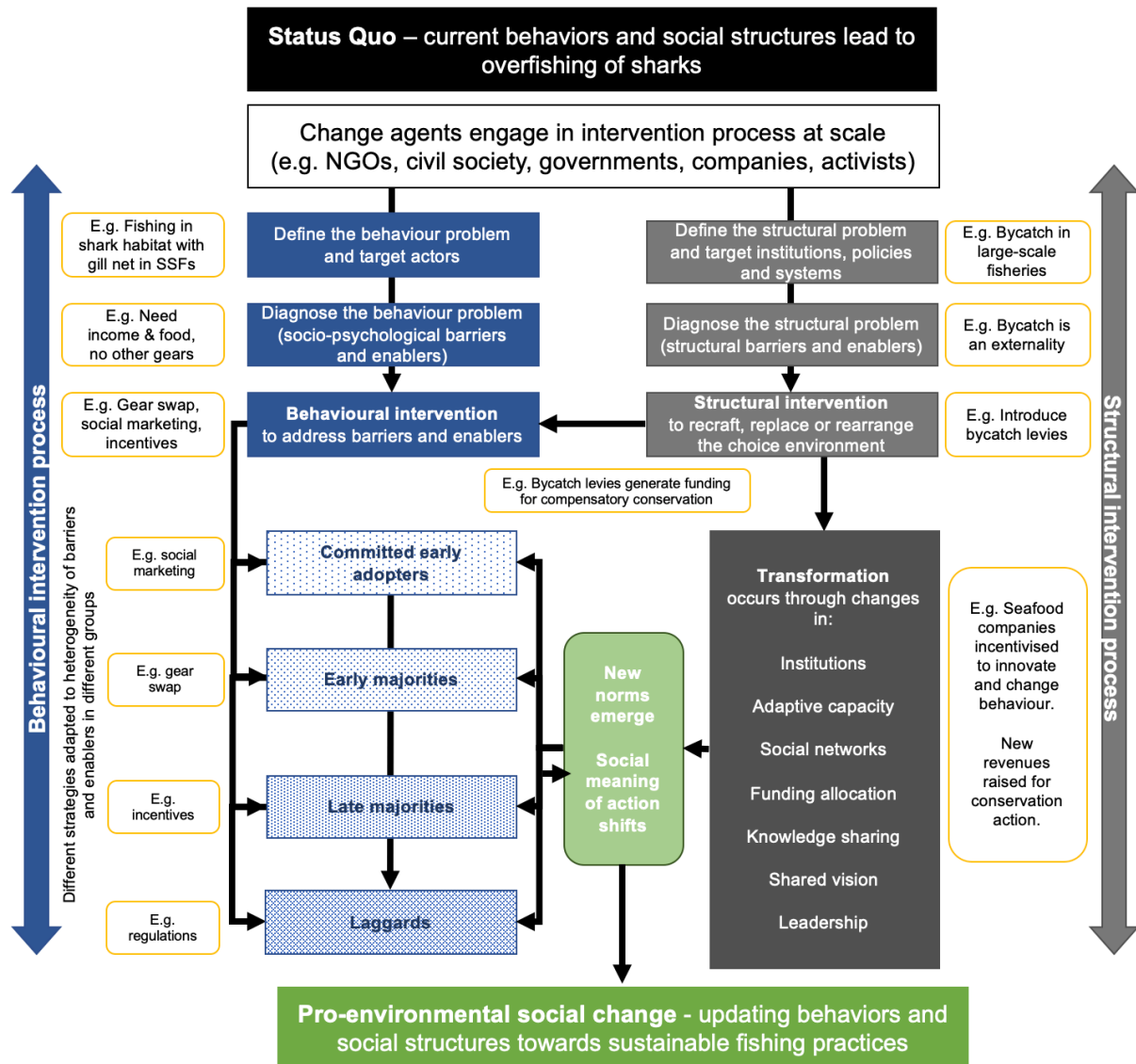


Figure 11.3 An integrative framework for pro-environmental social change from Naito et al. 2021, which includes parallel processes of behavioural (blue) and structural (grey) interventions, with examples for shark conservation taken from this thesis (yellow boxes)

As part of this dual approach, structural interventions for shark conservation could benefit from a more holistic understanding of patterns of shark decline, which: a) acknowledges and addresses the role of the entire global seafood industry – in particular general commercial fisheries expansion and bycatch – in shark population declines (Pacoureau et al., 2021), rather than framing it as a primarily Asia- or trade-driven threat; b) considers the realities of wildlife markets (Challender et al., 2015a, 2015b; Hinsley & 't Sas-Rolfes, 2020), including: relationships between supply and demand; interactions between markets for

different shark-derived commodities; and telecouplings with other markets and supply chains, particularly for food; c) accounts for current and historic disparities in marine ecosystem service distribution and ocean exploitation. For example, Leclere et al. (2020) recently outlined that “*bending the curve of terrestrial biodiversity needs an integrated strategy*”, and this strategy should not only include traditional conservation actions, such as protected areas, but also requires transformations in food systems, such as sustainably increased crop yields and diet shifts (Leclere et al., 2020). A similar strategy is required to bend the curve on marine biodiversity loss, with concomitant changes in sustainable seafood supply and demand, to mitigate the on-going marine biodiversity impacts fisheries; and pro-active conservation actions, to actively restore and recover marine biodiversity.

11.2.4 Implementing robust interventions in the real world

While the research in this thesis points to several opportunities for shark conservation interventions, and in particular the untapped potential of market/incentive-based approaches, success will ultimately depend on how well real-world rules and institutions fit the dynamic socio-ecological context in which they are implemented (Brooks et al., 2012; Muradian et al., 2013; Muradian & Gómez-Baggethun, 2013; Waylen et al., 2010; Wright et al., 2016). By using both impact assessments and predictive methods, I have sought to build a better understanding of the conditions under which interventions could work, and the instrument mixes which could facilitate success in different situations. However, many of the ideas presented remain hypothetical, and need to be tested in practice. While the predictive methods used in this thesis leveraged the contextual experiences of fishers, tourists and managers – which strengthen their internal validity and potential value in decision-making (Addison et al., 2016; Christie et al., 2020) – the hypothetical scenarios were necessarily reductive, and responses may not account for complex and unexpected feedbacks that might occur. Exploring and testing the extent to which these ‘predictions’ of this thesis align with real-world outcomes would help evaluate the predictive potential of scenario-based methods such as those used here.

Moving forwards, effective monitoring and compliance management will be necessary for securing behavioural change and conservation outcomes under any intervention, particularly when fishers may be inclined to cheat (e.g., as highlighted in Chapter 8). Monitoring and compliance management is notoriously difficult and costly in the marine realm (Boonstra & Österblom, 2014; Sumaila et al., 2006), but can be improved through several avenues. Firstly, using data to identify spatio-temporal ‘hotspots’ of non-compliance can enable efficient distribution of patrol effort in space (Arias et al., 2016). Secondly, improvements in technology – such as advances in on-vessel video monitoring, species ID using machine learning, and forensic hold monitoring using eDNA (Bartholomew et al., 2018; Harper et al., 2019; Mangi et al., 2015) – can all help to support low-cost detection of non-compliance. For example, participation in an incentive scheme could be contingent on always using an on-vessel video, and providing post-trip hold samples, with the burden of proof placed on fishers themselves (e.g., payments conditional on submission

of footage/samples). Third, establishing institutions which promote cooperation or competition could facilitate peer monitoring, with peer pressure to comply (Kotchen & Segerson, 2019, 2020; Muradian, 2013). Fourth, any positive incentives or sanctions should be set at a rate which promotes compliance. According to basic economic theory, this requires that the risks of non-compliance are greater than the rewards. For example, Sumaila et al. (2006) estimated that fines for Illegal Unreported and Unregulated fishing would need to increase by 24 times for the expected cost of enforcement to be at least as much as the expected benefits of non-compliance (Sumaila et al., 2006). However, such sanctions may be considered socially unacceptable. An alternative real-world example is New Zealand's deemed-value system, which is an incentive-based rather than criminal offence-based approach for facilitating compliance with fishing quotas and catch balancing (Peacey, 2002). In this system, price-based taxes are levied on non-quota catches, which are set at rates which aim to disincentivise non-quota catches without incentivising discarding (Walker & Townsend, 2008). Finally, institutions will need to be underpinned by trust, local leadership and social capital to ensure legitimacy (M. L. Barnes et al., 2016; Brooks et al., 2012; Gutiérrez et al., 2011; Oyanedel, Gelcich, & Milner-Gulland, 2020), and to guard against elite capture and abuses of power (Adhikari & Boag, 2013; Pascual et al., 2014).

11.3 Gaps and limitations

While this thesis highlights the need for a concerted shift from research on the threats to sharks towards research on responses (Figure 11.2) and outlines several potential methods for doing so, gaps and limitations remain with respect to the methods and results I have presented, which could be improved upon in future research.

In Chapter 3, I outlined the mitigation hierarchy (MH) as a framework for reconciling trade-offs between shark conservation and fisheries objectives. At the time of writing the MH had yet to be applied to a shark conservation context, and so remained somewhat theoretical, however the MH has since been applied to explore management measures for elasmobranch bycatch mitigation in a trawler fishery in India (Gupta et al., 2020). During Gupta et al.'s study, several challenges in applying the MH were identified, including setting a quantitative bycatch reduction target, and quantifying the potential impact and feasibility of different management measures. This was primarily due to limited data on shark population dynamics; critical habitats, and spatio-temporal variation in habitat use; and post-capture survivability (Gupta et al., 2020). Therefore, if the MH is to be applied quantitatively in fisheries, as a way of measuring contributions and progress towards overarching outcome goals, further efforts are needed to fill outstanding gaps in basic fishery and biological data. At a broader scale, full implementation of the MH, and associated instruments such as bycatch levies (Chapter 10), also require accurate and timely data on the impacts of multiple fisheries and supply chains. While there have been innovations in fisheries monitoring and transparency in recent years (e.g., GlobalFishingWatch), there remains a gap in terms of translating data on the spatial footprint of fisheries into data on bycatch liabilities and accountability.

Though there have been some early attempts to translate this data into risk assessments (e.g., Queiroz et al., 2019), current methods have been criticised for their over-simplicity and flawed assumptions (Murua et al., 2021), and have yet to be taken to scale in terms of producing wide-spread real-time data on environmental impacts and heterogeneities across suppliers and supply chains. In the future lessons could be applied from third party monitoring of deforestation and the impacts of agri-food supply chains (e.g., GlobalForestWatch, Hestia, RSPO), and how this data is being leveraged to improve corporate social responsibility and consumer activism (Schouten & Glasbergen, 2011). However, this will require robust methods to translate two-dimensional spatial data into marine biodiversity impacts (e.g., S. Griffiths et al., 2019). Developing methods and systems which can rapidly and reliably estimate these impacts, and which are validated (e.g., by observer data on actual bycatch outcomes) represents an important and impactful research frontier for advancing environmental accountability in the seafood industry.

Some of the theories, methods and predictions offered in this this thesis also require further empirical testing and validation. For example, Chapter 5 outlined the Theory of Planned Behaviour (TPB) as a potential framework for diagnosing the socio-psychological drivers of bycatch-relevant behaviour. However, the reliability of the TPB in terms of predicting actual fisher behaviour, and the relative predictive power of the different belief measures within the TPB, remain untested. In the future, TPB surveys of larger samples of fishers across multiple sites, combined with data on real bycatch outcomes from landings surveys, would enable robust statistical modelling to evaluate the predictive potential of the TPB for diagnosing drivers of bycatch, and the relative influence of attitudes, subjective norms and perceived behavioural control on actual fisher behaviour and bycatch performance (e.g., as previously applied to identify determinants of and predict recycling behaviour (Nigbur et al., 2010; Tonglet et al., 2004). Similarly, Chapter 8 used scenario interviews to predict fisher behaviour under a range of plausible future scenarios. However, the validity of scenario-based methods for predicting pro-conservation behaviour is yet to be fully evaluated. In the future, exploring the extent to which participants' stated responses align with observed outcomes under a real intervention would help to validate the predictive potential of scenario-based methods. Finally, tourists' stated willingness to pay for marine conservation (Chapter 9) may also diverge from revealed preferences, e.g., due to 'cheap talk' (Mahieu et al., 2012). Though I attempted to guard against this in this study design, the results could be supported by, for example, reviewing revealed preferences based on real-world examples of marine conservation levies, or conducting field experiments and comparing real payments with stated WTP (De Martino et al., 2016).

A further limitation of this research is a strong focus on the use of economic valuation as the primary means for understanding costs and cost-effectiveness. For example, chapters 6 and 7 both used data on trip profits to understand hypothetical opportunity costs and socio-economic feasibility of management measures. However, trip profits alone represent a somewhat simplified indicator of the potential wider values of small-scale shark fisheries, and their contributions to holistic well-being (Weeratunge et al.,

2014). These data are also somewhat individualistic, and may fail to account for deeper held values at the group or community level (Kenter et al., 2011). In the future, these relatively narrow assessments of well-being could be supplemented with participatory and deliberative techniques, to further elucidate non-monetary values associated with certain fishing practices, such as social capital and pride (Woodhouse et al., 2015). Chapter 4 – which assessed the impact of a manta ray conservation intervention – also took a relatively narrow focus on impact, with manta ray mortality as the primary outcome measure. However, as emphasised throughout this thesis, a more holistic and equitable approach to shark conservation requires that impacts on coastal communities are explicitly acknowledged and accounted for (Milner-Gulland et al., 2014), and I did not thoroughly assess impacts on human well-being, nor the cost effectiveness of the intervention. A challenge in this regard was a lack of time series data on measures of well-being (though participatory approaches can be adopted where experimental or statistical methods aren't feasible (Woodhouse et al., 2016)), and a lack of data on project costs. Standardised reporting methods – adopted by NGOs and funders – could help to overcome these limitations in the future, and promote greater transparency on the cost-effectiveness of conservation actions and organisations (Baylis et al., 2016; Milner-Gulland et al., 2014; Pienkowski et al., 2021).

11.4 Summary of next steps for science and practice

There are several future directions for research and practice based on the findings of this thesis, including: addressing outstanding gaps, limitations and uncertainties (outlined in Section 11.3); building on and scaling up methods; and implementing actionable findings and testing their impacts.

Firstly, this thesis has highlighted the utility of using interdisciplinary approaches in shark conservation science, and offered some methods which can be further tested and scaled by researchers and practitioners. For example, socio-psychological approaches (Chapter 5) can be used to diagnose the behavioural drivers conservation problems; econometric methods can be used to understand markets, supply and demand, and wider structural issues (Chapter 7); the mitigation hierarchy (Chapter 3) can be used to collate and prioritise potential responses; predictive methods can be used to pro-actively explore the potential impacts of behavioural intervention, and design appropriate intervention mixes (Chapter 8); and robust monitoring and impact assessments can be used to inform adaptive management (Chapter 4). Such research can help to fill gaps in grounded and contextually-rich conservation evidence, which can be tailored to meet specific conservation needs, and co-constructed with those who will be affected by and/or have the agency to facilitate conservation action (Addison et al., 2016; Christie et al., 2020, 2021; Wyborn & Evans, 2021).

Secondly, there are opportunities to implement and test behavioural and structural interventions for shark conservation based on these findings. Together, these interventions could form an integrated strategy for recovery of sharks (and marine megafauna in general), with potential roles for NGOs, civil society and

researchers in driving this strategy (Figure 11.4). The recently proposed mitigation and conservation hierarchy (MCH) offers a potential framework for drawing together the various approaches proposed in this thesis, and structuring an integrated strategy towards an aspirational goal for marine biodiversity (Milner-Gulland, Addison, et al., 2020). The MCH unites the mitigation hierarchy – to address contemporary, attributable biodiversity impacts towards at least NNL of biodiversity, with proactive conservation actions – to address past, indirect and diffuse biodiversity impacts for aspirational net gain (Milner-Gulland, Addison, et al., 2020). In the context of shark conservation, bycatch levies offer a potential instrument for incentivising implementation of the mitigation hierarchy in fisheries, whereby levies incentivise uptake of measures which avoid, minimise and remediate impacts on marine megafauna during fishing activities, and raise revenue for compensatory conservation actions. This could help to internalise bycatch, and drive changes in seafood supply and demand (Chapter 10). In parallel, tourism levies offer a feasible instrument to facilitate pro-active conservation actions, whereby those who value and benefit from marine biodiversity can pro-actively contribute towards marine protection (Chapter 9). When paired with well-designed investments in behavioural interventions in coastal communities and small-scale fisheries – such as funding gear swaps or performance-based payments for marine conservation outcomes (Chapter 8) – this combination of instruments could help to recover marine biodiversity (i.e., achieve >NNL) (Figure 11.4), and facilitate distributive justice by addressing cost-benefit inequities in conservation.

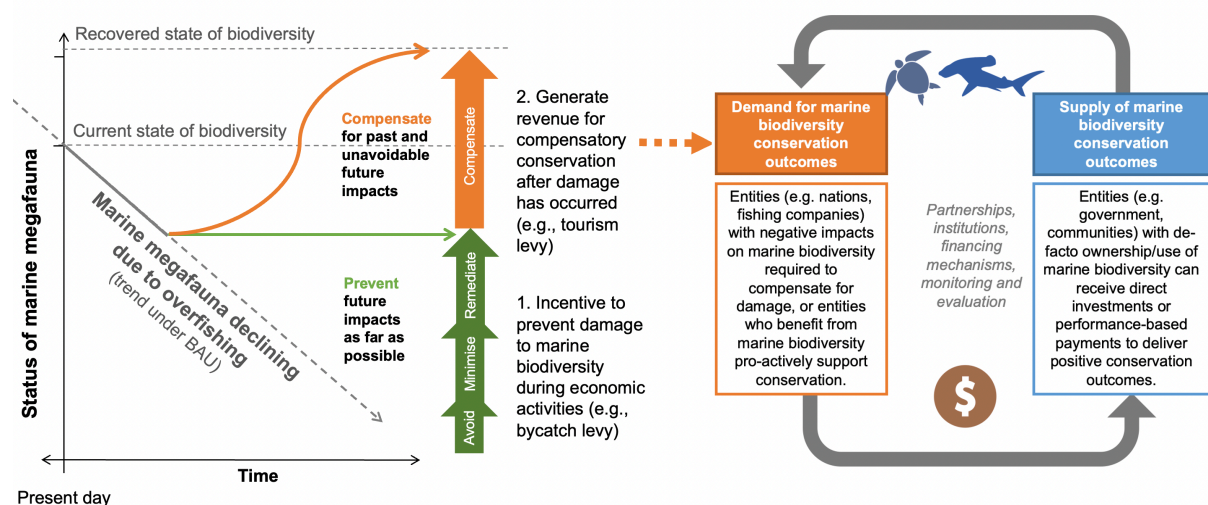


Figure 11.4 A schematic of how bycatch levies, tourism levies and marine conservation payments could support an integrated strategy for recovery of marine biodiversity

Yet several gaps remain – in research and practice – in order to effectively deliver this strategy. In particular, it is predicated on the assumption of emerging markets for marine biodiversity, which remain somewhat theoretical. However, there are several ways in which researchers and practitioners could collaborate, to support the emergence of these markets in the future, for the net benefit of marine biodiversity (Figure 11.5). Firstly, piloting conservation incentives across several SSFs, along with robust

impact and cost-effectiveness assessments that draw on best-practice in field experiments (Duflo et al., 2006; Ferraro, 2009), could provide empirical evidence on if, how and to what degree such approaches can simultaneously deliver conservation outcomes for sharks and well-being outcomes for coastal communities (Figure 11.5). Field experiments could be designed to compare the performance of different payment regimes and institutional arrangements across different contexts (e.g., Wunder et al., 2008), and the real-world outcomes could also be used to evaluate the predictive validity of the methods used in this thesis. At the systemic level, this evidence could provide the basis for investment-ready schemes for “bycatch-neutral” seafood supply chains. Just as investors and consumers are demanding “deforestation-free” terrestrial supply chains (CDP, 2014; Donofrio et al., 2017) they may soon demand “bycatch-neutral” seafood; stimulated by adoption of NNL and net gain policies, growing consumer and shareholder activism, and corporate ocean stewardship (Jouffray et al., 2019; Österblom et al., 2017). This will require that entities which harm marine biodiversity during fishing operations implement the mitigation hierarchy – by first avoiding and minimising bycatch as far as possible, and then remediating and compensating for any residual impacts (Figure 11.4). Since bycatch is difficult to avoid entirely, a supply of marine biodiversity offset projects – which deliver measurable, additional conservation outcomes for a given cost – will be needed to fulfil this demand (Figure 11.4). Future research could also focus on assessing the negative biodiversity impacts of the seafood industry, to calculate biodiversity liabilities; and the positive biodiversity impacts of conservation projects in coastal communities (either actual or hypothetical), to estimate supply and cost per unit of additional biodiversity outcomes and verify costs and conservation outcomes (Figure 11.5).

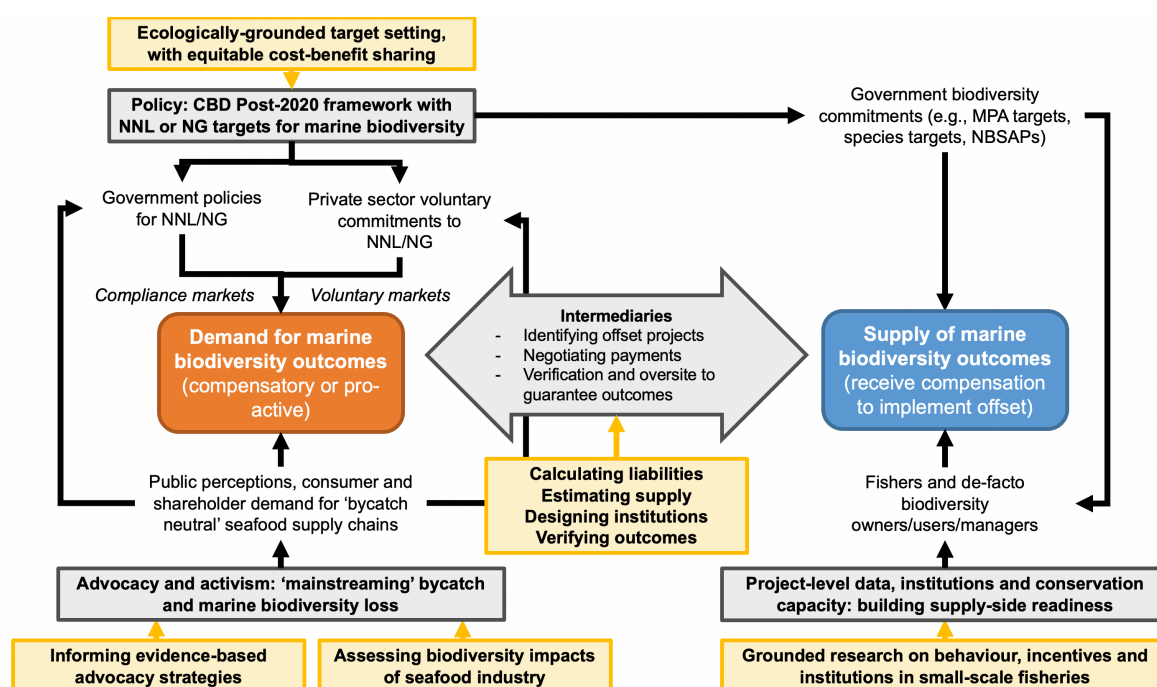


Figure 11.5 A simplified theory of change for emerging marine biodiversity markets, as part of an integrated strategy for recovery of marine biodiversity (Figure 11.3), with potential roles and intervention points for NGOs and civil society (grey boxes) and research opportunities (yellow boxes)

Taking these markets to scale will also require novel methodologies for rapidly and independently estimating bycatch liabilities across multifarious seafood supply chains (e.g., as per those developed for estimating carbon and deforestation liabilities under REDD+ and the RSPO (Estrada, 2011; RSPO, 2020)), and for costing and designing levies and offsets across different types of contexts. Grounded research could also support the design of robust institutional arrangements and agreements, as well as on-going monitoring and verification of outcomes to ensure NNL targets are met (Figure 11.5).

However, despite the potential demonstrated in this thesis, actual demand for marine biodiversity outcomes from the fishing and tourism sectors remains relatively low. It is perhaps only a matter of time until compliance markets emerge following increasing adoption of NNL or net gain commitments (CBD, 2020a; Shumway et al., 2018). However, transparency, advocacy and activism initiatives could help to drive consumer demand for bycatch neutral supply chains, and encourage government and corporate commitments (e.g., by increasing fishing firms' reputational risks of not addressing bycatch (Ruysschaert & Salles, 2018; Schouten & Glasbergen, 2011; Segerson, 2013), or creating value for tourism firms which are actively addressing conservation issues) (Figure 11.5). Social and behavioural research could be used to design evidence-based communications and advocacy strategies, such as targeted online or consumer campaigns, which focus on specific demographics or leverage particular preferences (Doughty et al., 2020, 2021; X. Zhou et al., 2021). In parallel, research which enables 'naming and shaming' of specific entities with respect to their marine biodiversity impacts may promote cooperation, political change and social reform (Jacquet, 2016; Jacquet et al., 2011) (Figure 11.5).

At the policy level, further research is also needed to inform ecologically-sound goal setting for marine biodiversity under the post-2020 Global Biodiversity Framework (Figure 11.5), and to monitor progress towards those goals. Goals, targets and associated metrics or indicators need to capture multiple elements of biodiversity (e.g., species, ecosystems, genetic diversity) and guard against the potential perverse outcomes from taking a net approach; as well as being temporally relevant, and divisible and scalable across jurisdictions, institutions and management-relevant spatial scales (Maron et al., 2021; Mcowen et al., 2016; Milner-Gulland, Addison, et al., 2020). Partnerships between companies, governments and researchers can support development of metrics and indicators which are meaningful, translatable across contexts, and feasible to measure (Addison et al., 2020; Mcowen et al., 2016). The process of goal and target setting should also be founded on principles of cost-effectiveness and equity, with contributions of different entities and nations towards overarching goal tailored towards their biophysical and socio-economic contexts (e.g., Figure 11.1). Future research could focus on methodological innovations for operationalising equity in a decision-making context, to determine which polluters and beneficiaries should pay, and which should be the recipients of financial support.

Finally, for effective and scalable implementation, there is also a need to develop fair partnerships between companies and communities; along with institutions and capacity for long-term transparent allocation of funding. Advocating for change, fulfilling intermediary functions, and building supply-side readiness could all be important roles for NGOs and civil society in the future (Figure 11.5), mirroring NGO involvement in facilitating and establishing REDD+ payments in the terrestrial realm (B. Williams, 2021).

In the future, applied interdisciplinary research can play a critical role in driving transformative change for shark conservation, and marine management in general (Figure 11.5). This will require that researchers think big (i.e., research to inform policy and macro-level structural change) and think small (i.e., grounded research to inform local-level behaviour change) at the same time. Moreover, in order to lead to greater uptake and impact, future research could focus on being contextually-rich and response-oriented and grounded in principles of co-production, supported by productive research-implementation spaces (Christie et al., 2020; Toomey et al., 2017; D. R. Williams et al., 2020; Wyborn & Evans, 2021). This will help to provide change-makers with the evidence they need to drive pro-environmental social change – for sharks, rays and biodiversity in general – which can move humanity towards a pathway of “*living in harmony with nature*” in the coming decade.

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Appendix 1: Supplementary Material for Chapter 4

S4.1. Crime script, framed within the 25 techniques of situational crime prevention

Stages	Steps	Situational Prevention Technique	Response
Manta hunt supported	Vessels, equipment, supplies and crew funded	Deny benefits	Arrests of traders and seizure of products in the national supply chain
		Reduce anonymity	Key facilitators identified and publicly confronted / arrested
Poachers motivated to find and kill manta rays	Sense of pride in being a manta ray hunter	Neutralize peer pressure	Make it shameful to kill manta rays
			Media promotion of fishers reporting and rescuing marine megafauna
			Awards given to fishers who protect marine megafauna
	Financial incentive to killing manta rays	Reduce emotional arousal	Make it shameful to kill manta rays
		Deny benefits	Seizure of manta products and equipment related to manta kills
	Lack of [negative incentives] to killing manta rays	Reduce frustrations and stress	Provision of suitable alternative livelihoods
		Set rules	Socialization and village meetings to raise awareness of the new regulation
		Avoid disputes	Lobbying for significant local political support for manta ray protection
		Assist compliance	Hotline to report megafauna at risk of harm (caught in nets etc.)
Preparation	Prepare large crew and lempara	Assist natural surveillance	Deploying informants in a strategic locations to watch poaching activity
Find manta ray	Depart from Lamakera	Assist natural surveillance	Engagement with community informants
		Screen exits	Patrols stop vessels leaving Lamakera that match the search profile
		Reduce anonymity	Patrols to check fishing permits
	Search hotspot areas	Deflect offenders	Risk-based patrols of spatio-temporal hotspots
		Strengthen formal surveillance	Community monitoring groups equipped with GPS & phones
Kill manta ray	Catch manta with metal-tipped bamboo harpoon	Control tools/weapons	Seising the metal harpoon tips
			Seizure of boats involved in marine crimes

		Extend guardianship	Provision of fast patrol boats for rapid reaction
Transport manta ray	Drag manta onto boat or to shore	Extend guardianship	Community fishing monitoring groups equipped with GPS & phones
Process manta ray	Process manta ray on the village shoreline	Assist natural surveillance	Engagement with community informants
	Separate products	Assist natural surveillance	Engagement with community informants
	Hang gill rakes and [osaphagus] to dry	Deny benefits	Operations to seize manta products in the village
Transport manta products	Manta meat goes to local markets	Disrupt markets	Market surveys and enforcement operations
		Assist natural surveillance	Deploy community monitors
	Manta gill rakes go into local supply chain	Deny benefits	Arrests of traders and seizure of products in the local supply chain
	Gill rakes transported through national supply chains	Identify products	Forensic tools used to identify manta products in trade
	Gill rakes transported to international markets	Deny benefits	Transnational enforcement collaborations

S4.2. Key assumptions for causal inference and attribution, and methods used to test them.

Assumptions		Methods used to test assumptions
A1	The hypothesised ToC is an accurate representation of the world, with true causal links between interim results, objectives, outcomes and impact	Temporal coincidence of observed trends and testing for statistical correlations between datasets
A2	Observed trends in empirical data are reliable. The data are not biased, and provide an accurate representation of reality.	Qualitative contextual information on reliability of data sources to assess inferential weight of evidence, triangulation of multiple sources of evidence
A3	Observed trends are caused by the conservation intervention, as opposed to external confounding factors outside of our field of influence and monitoring.	Counterfactual modelling natural experiments, process tracing, general elimination

S4.3. Detailed description of data sources and methods

NGO Project reports and records

Secondary data gathered from unpublished records, collected by the Wildlife Conservation Society Indonesia Program and Misool Foundation, as indicators for use in external donor reporting. Indicators include:

- Number of business units developed for non-manta livelihoods
- Number of community members benefiting from new non-manta livelihoods
- Number of (ex) manta ray hunters committed to complying to regulations
- Number of community members engaged in institutions for marine monitoring and management
- Frequency of marine monitoring reports from community
- Number of socialisation and training events, and estimated village coverage/attendance

Additional evidence includes:

- Meeting sign-in sheets
- Signed community agreements
- Cooperative records

Time frame of data collection: February 2014-December 2018

Custodian of unpublished records: Wildlife Conservation Society Indonesia Program, Misool Foundation

SMART patrol data

SMART is a Spatial Monitoring and Reporting Tool to measure, evaluate and improve the effectiveness of wildlife law enforcement patrols and site-based conservation activities (<https://smartconservationtools.org/>). Following the commencement of the site-based patrols in Lamakera in 2015, DKP FloTim adopted SMART to collect, store, communicate, and evaluate ranger-based data on: patrol efforts, patrol results, and threat levels.

Time frame of data collection: January 2015-December 2018

Custodian of unpublished records: DKP Flores Timur

Some data is confidential; some data can be made available on request

Intelligence data

Secondary data gathered from unpublished records.

Collected by government agencies and the Wildlife Crimes Unit and collated and analysed by the Wildlife Conservation Society Indonesia Program using IBM i2 software.

Time frame of data collection: February 2014-December 2018

Custodian of unpublished records: Indonesian government

Data is confidential, but a summary can be made available on request.

Law enforcement data

Secondary data gathered from unpublished records

Collected by Indonesian law enforcement agencies and the Wildlife Crime Unit and collated by the Wildlife Conservation Society Indonesia Program

Time frame of data collection: February 2014-December 2018

Custodian of unpublished records: Indonesian government

Some data is confidential; some data is publicly available in Indonesian legal proceedings.

Landings data

Data gathered from published literature and unpublished records.

- Historic data (2013-2015) is sourced from Lewis et al. (2015), which was collected based on local knowledge, via community landings records/interviews with village elders (Lewis et al. 2015)
- Data from May 2015-December 2018 is based on daily landings monitoring by Misool Foundation enumerators, following survey methods from White et al. (2006), and supplemented by additional observation from Misool and WCS-IP field staff

Custodian of unpublished records: Misool Foundation. Some data publicly available in Lewis et al. (2015).

S4.4. Further illustrative evidence for activities and results

Outreach, consultation and awareness raising

Photos from informal community outreach (left) and formal community socialisation events (right)



Community surveillance

Photos illustrating bycatch release of a manta (top left) and whale shark (top right), POKMASWAS training (bottom left) and award for bycatch reporting and release (bottom right)



Livelihood-focused incentives

Photos illustrating the mini-purse seine vessel (left) and cooperative mini-mart (right)



Marine patrols and interception of illegal fishers

Photo of patrol boat in action (left) and a routine inspection with a spear seised (right)



Trader investigations and arrests

Photos from high-profile manta ray trade enforcement cases, with gills being seized

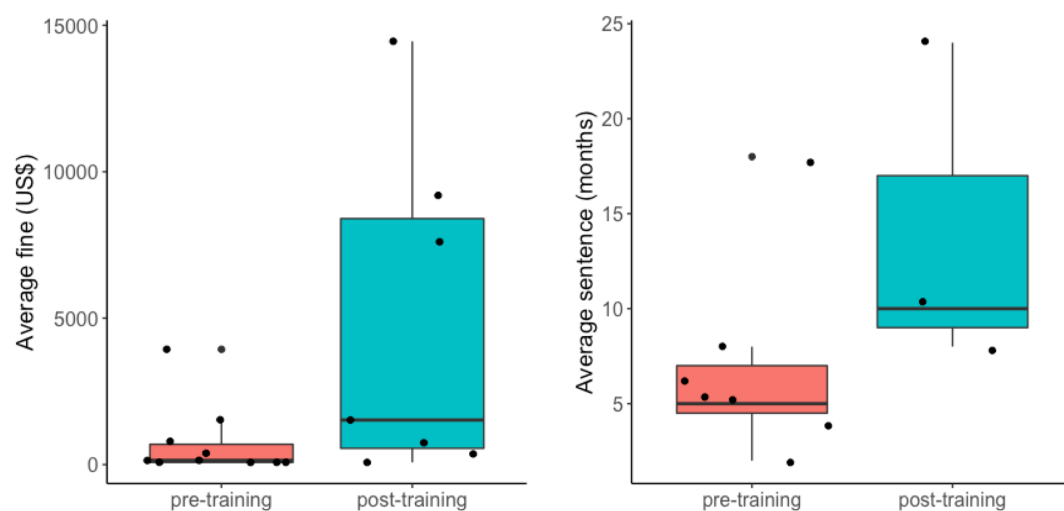


Enforcement training and sanctions

Photo from enforcement officer training

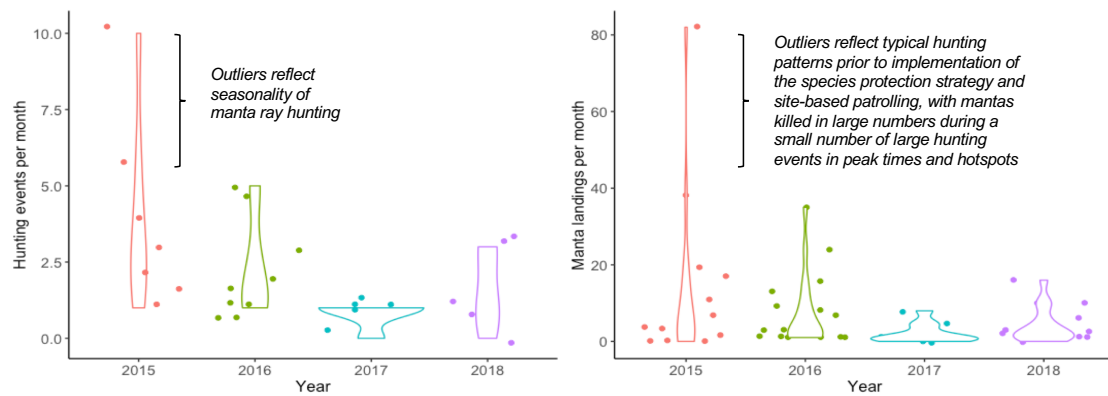


Average fines and sentences for prosecuted illegal elasmobranch traders before and after onset of trainings



Outcomes and impact

Trends in mean observed hunting effort and mortality from 2015 to 2018.



S4.5. Detailed methods

Negative binomial models of manta hunting events, landings and patrol effort

To evaluate whether manta ray catch/hunting events changes through time and, if there is any changes, whether it is related to patrols and patrolling in hotspot areas, we used the following variables and test their correlations/trends using linear regression models, choosing the best model distribution.

Table S4.5a. Variables modelled to explain patterns of the number of manta landing.

Dependent variables	Explanatory variables	Variable type
Daily manta ray catch/hunting events	Year	Integer
Monthly manta ray catch/hunting events	Month	Integer
	Hotspot: Yes (1) or No (0)	Binomial
	Patrol: Yes (1) or No (0)	Binomial

Description of each variables are as following:

1. Daily manta ray catch/hunting events

Total number of manta ray caught based on daily landings pre and post intervention.

2. Monthly manta ray catch/hunting events

Monthly aggregation of daily manta ray catch/hunting events.

3. Year

Year of daily landings record.

4. Patrol focus

Occurrence of adaptive data-driven patrolling strategy to target patrols toward hotspots (Yes = 1, or No = 0).

5. Patrol

Occurrence of patrol. For daily manta ray catch analysis, data used is 1 (Yes to patrol) and 0 (No patrol), while for monthly manta ray catch analysis, data used is total patrol per month. We included an interaction terms between Patrol Focus and Patrol since the two are highly correlated.

6. Month

Month is the month in which the landings occurred.

We divided the analysis into four sets of models:

1. Using daily manta catch, include non-fishing season
2. Using daily manta catch, exclude non-fishing season
3. Using monthly manta catch, include non-fishing season
4. Using monthly manta catch, exclude non-fishing season

Model using daily manta catch, include non-fishing season

We conduct the analysis to see what cause changes of daily landings of manta ray. We first check the data distribution of the daily manta catch, using a simple histogram.

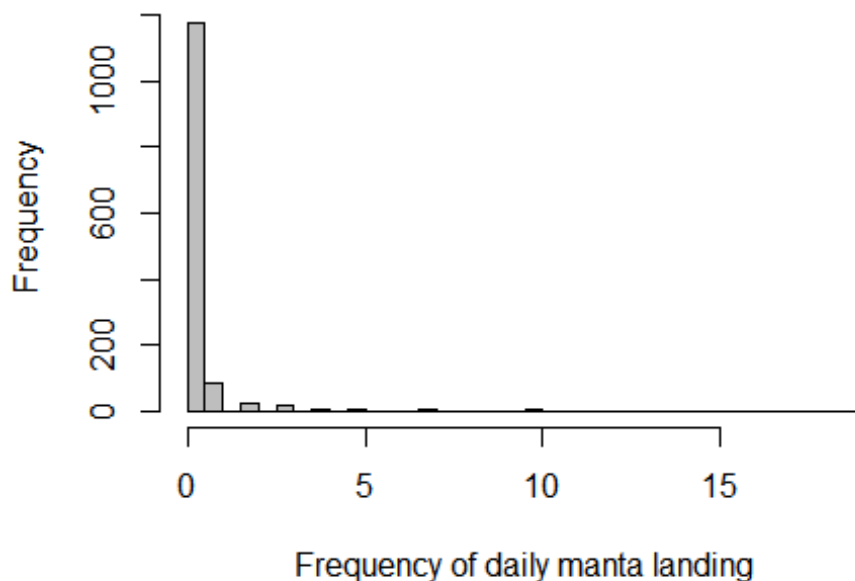


Figure S4.5a. Histogram of daily manta landings.

We detected a huge amount of zero values (>1000). Since the data is integer with high number of zero values, we can use the zero inflated negative binomial model to fit the data (Zuur, et al. 2009). We will nevertheless go through a protocol to evaluate the model using gaussian, poisson, negative binomial (correcting for over dispersion), zero inflated poisson (ZIP) and zero inflated negative binomial (ZINB). The full model is as following:

Table S4.5b. Summary of AIC and %R2 values of models based on different distributions.

Daily manta landing=Year+Patrol focus : Patrol+Month

Model distribution	%R2	AIC	Δ AIC	df
Gaussian	3.1	4565.280	0.000	7
Poisson	20.8	2243.704	2321.576	6
Negative binomial	6.8	1386.512	3178.768	7
Zero-inflated Poisson	4.5	1502.841	3062.439	12
Zero-inflated Negative Binomial	14.4	1334.084	3231.196	13

The zero inflated negative binomial model (mod4 model) gave the lowest AIC value and a relatively good R^2 value. We then use this distribution for our model. Following is the summary of the best model:

```
##
## Call:
## zeroinfl(formula = f4, data = mt, dist = "negbin", link = "logit")
##
## Pearson residuals:
##   Min     1Q  Median     3Q    Max
```

```
## -0.3587 -0.3064 -0.2412 -0.1455 10.8076
##
## Count model coefficients (negbin with log link):
##      Estimate Std. Error z value Pr(> |z|)
## (Intercept) 302.7685      NA      NA      NA
## iyear      -0.1516      NA      NA      NA
## monthnum    0.3041    0.0409  7.435 1.05e-13 ***
## pat        -0.5318    0.6902 -0.771 0.44096
## focus2     -0.8670    0.2387 -3.633 0.00028 ***
## pat:focus2  0.6594    0.7986  0.826 0.40895
## Log(theta) -1.9253    0.1186 -16.228 < 2e-16 ***
##
## Zero-inflation model coefficients (binomial with logit link):
##      Estimate Std. Error z value Pr(> |z|)
## (Intercept) 2122.5287   14.6337 145.044 < 2e-16 ***
## iyear      -1.0721      NA      NA      NA
## monthnum    3.4397    0.9512  3.616 0.000299 ***
## pat        9.9516    3.3678  2.955 0.003128 **
## focus2      6.2537    1.6517  3.786 0.000153 ***
## pat:focus2 -12.9110    4.1238 -3.131 0.001743 **
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## Theta = 0.1458
## Number of iterations in BFGS optimization: 980
## Log-likelihood: -654 on 13 Df
```

We found that:

- Significant explanatory variables (based on the count model) including: patrolling at hotspots (focus2), month (monthnum).
- Significant explanatory variables (based on the binomial model) including: patrolling at hotspots (focus2), patrolling itself (pat), month (monthnum), and interaction between patrolling and patrolling within hotspots area (pat:focus2).

To understand what this means, following is a summary of mean value standard error (SE) of different prediction based on significant explanatory variables, followed by barplots for each significant explanatory variables. The mean value summaries are 1) interaction between patrolling and patrolling within hotspots area, 2) month, and 3) year, respectively.

```
## pat focus2 emmean SE df asymp.LCL asymp.UCL
## 0 0 0.435 0.0614 Inf 0.3145 0.555
## 1 0 0.255 0.1715 Inf -0.0814 0.591
## 0 1 0.183 0.0342 Inf 0.1156 0.250
## 1 1 0.208 0.0590 Inf 0.0919 0.323
##
## Confidence level used: 0.95

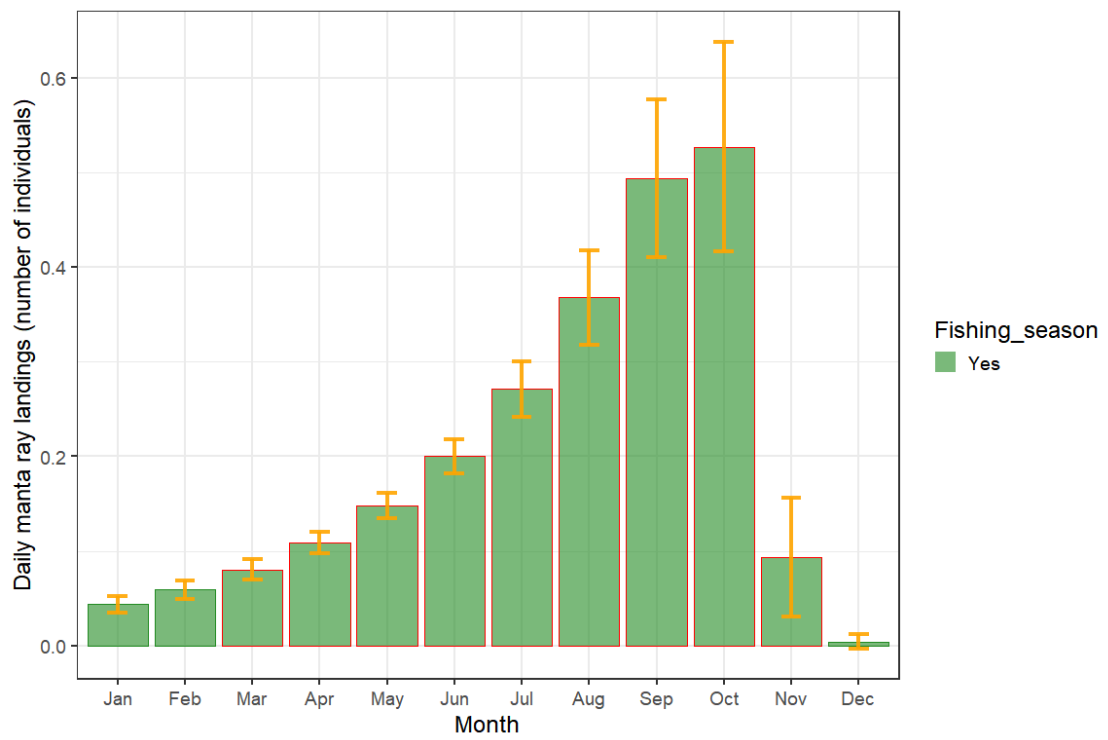
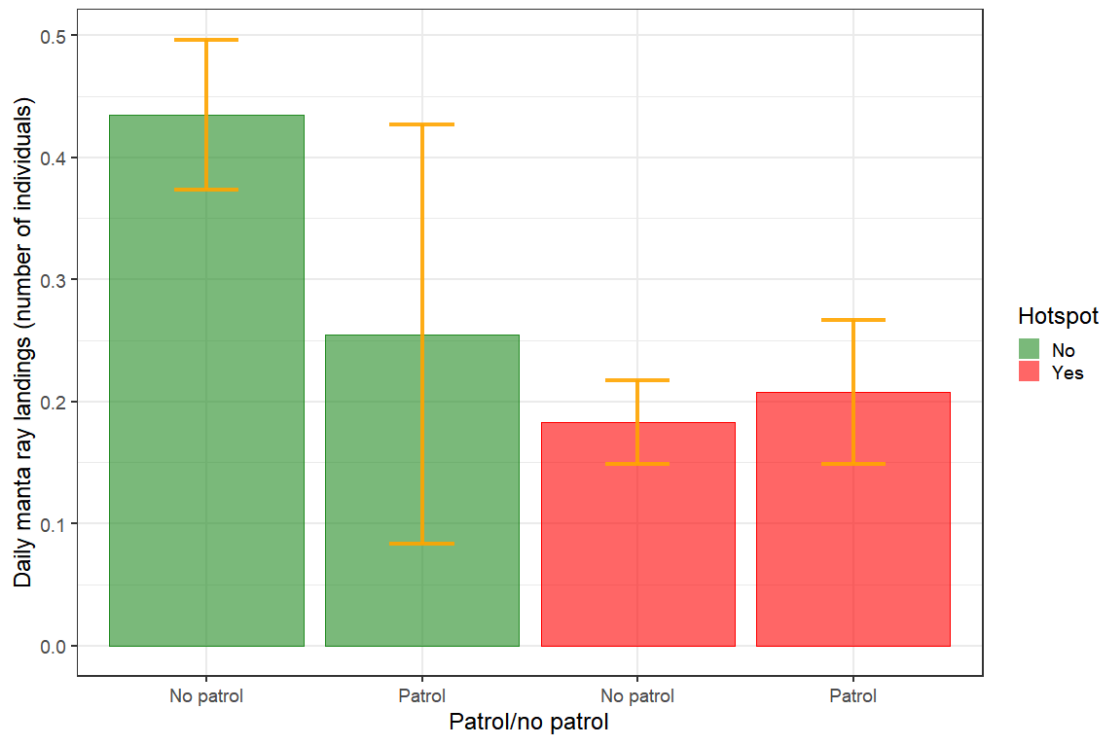
## monthnum emmean SE df asymp.LCL asymp.UCL
## 1 0.04377 0.00884 Inf 0.02645 0.0611
## 2 0.05933 0.00985 Inf 0.04002 0.0786
## 3 0.08041 0.01070 Inf 0.05944 0.1014
## 4 0.10898 0.01153 Inf 0.08639 0.1316
## 5 0.14771 0.01322 Inf 0.12179 0.1736
## 6 0.20020 0.01811 Inf 0.16470 0.2357
## 7 0.27134 0.02935 Inf 0.21381 0.3289
```

```

##      8 0.36766 0.05007 Inf  0.26952  0.4658
##      9 0.49399 0.08329 Inf  0.33074  0.6572
##     10 0.52696 0.11101 Inf  0.30939  0.7445
##     11 0.09350 0.06286 Inf -0.02970  0.2167
##     12 0.00451 0.00734 Inf -0.00987  0.0189
##
## Confidence level used: 0.95

## iyear emmean   SE df asymp.LCL asymp.UCL
## 2015  0.338 0.0360 Inf   0.268   0.409
## 2016  0.291 0.0308 Inf   0.230   0.351
## 2017  0.250 0.0264 Inf   0.198   0.302
## 2018  0.215 0.0226 Inf   0.170   0.259
##
## Confidence level used: 0.95

```



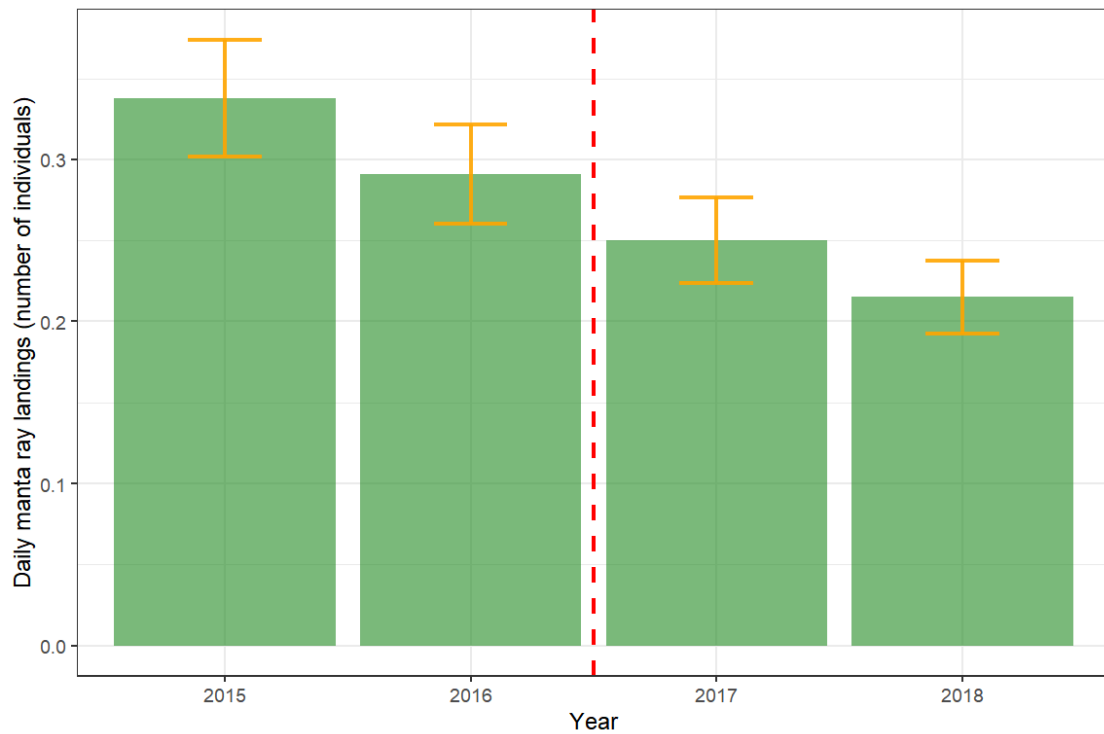


Fig S4.5b. Barplot of mean $\pm$ SE of manta landings based on different explanatory variables.

Top = interaction term of patrol & patrolling on hotspots, middle = month, bottom = year.

Red dashed line shows the start of patrolling with focus on hotspots

Model using daily manta catch, exclude non-fishing season

To detect differences due to analyzing data which includes off-season manta catch, we conducted similar analysis to part A while excluding off-season months. In order to do so, we excluded December, January, and February landings. We first check the data distribution of the daily manta catch, using a simple histogram.

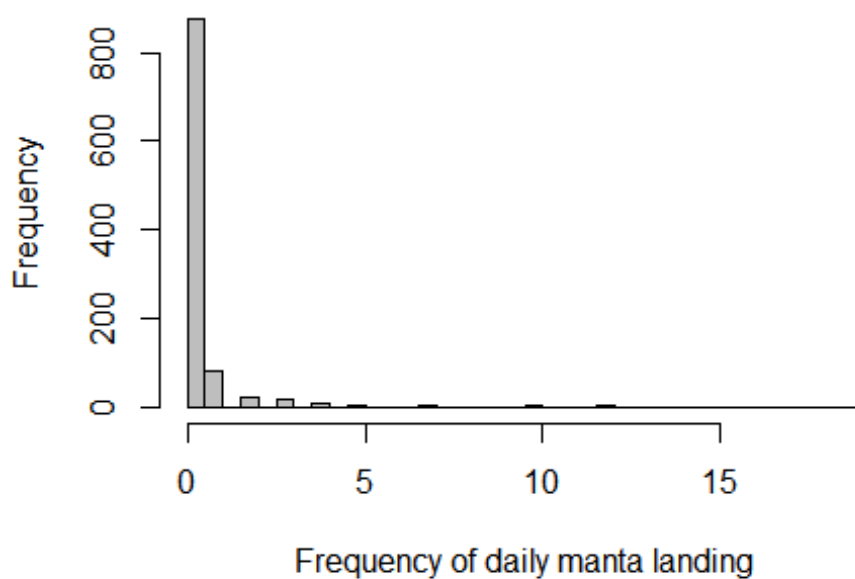


Figure S4.5c. Histogram of daily manta catch.

We detected a huge amount of zero values (>800). Since the data is integer with high number of zero values, we can use the zero inflated negative binomial model to fit the data (Zuur, et al. 2009). We will nevertheless go through a protocol to evaluate the model using gaussian, poisson, negative binomial (correcting for over dispersion), zero inflated poisson (ZIP) and zero inflated negative binomial (ZINB). The full model is as following:

Table S4.5c. Summary of AIC and %R2 values of models based on different distributions.

Daily manta landing=Year+Patrol focus : Patrol+Month

Model distribution	%R2	AIC	Δ AIC	df
Gaussian	3.8	3772.366	0.000	7
Poisson	23.3	2053.827	1718.539	6
Negative binomial	7.0	1306.654	2465.712	7
Zero-inflated Poisson	5.1	1427.539	2344.827	12
Zero-inflated Negative Binomial	10.9	1305.881	2466.485	13

The zero inflated negative binomial model (mod4 model) gave the lowest AIC value and a relatively good R^2 value. We then use this distribution for our model.

```
##
## Call:
## zeroinfl(formula = f4, data = mt1, dist = "negbin", link = "logit")
##
## Pearson residuals:
##   Min    1Q  Median    3Q   Max
## -0.3664 -0.3310 -0.2829 -0.2200  8.9684
##
## Count model coefficients (negbin with log link):
##           Estimate Std. Error z value Pr(>|z|)
## (Intercept) 446.42913      NA      NA      NA
## iyear       -0.22261      NA      NA      NA
## monthnum     0.24649    0.04814   5.120 3.05e-07 ***
## pat         -0.77005    0.63767  -1.208  0.2272
## focus2       -0.62840    0.25921  -2.424  0.0153 *
## pat:focus2   0.79612    0.76012   1.047  0.2949
## Log(theta)  -1.80526    0.12956 -13.934 < 2e-16 ***
##
## Zero-inflation model coefficients (binomial with logit link):
##           Estimate Std. Error z value Pr(>|z|)
## (Intercept) 689.8327      NA      NA      NA
## iyear       -0.3487      NA      NA      NA
## monthnum     1.1389    0.1549   7.351 1.97e-13 ***
## pat         -0.7161      NA      NA      NA
## focus2       3.0813    0.6583   4.681 2.86e-06 ***
## pat:focus2  -0.8089      NA      NA      NA
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## Theta = 0.1644
```

```
## Number of iterations in BFGS optimization: 50
## Log-likelihood: -639.9 on 13 Df
```

We found that:

- Significant explanatory variables (based on the count model) including: **patrolling at hotspots** (focus2), **month** (monthnum) (similar to results from part A).
- Significant explanatory variables (based on the binomial model) including: **patrolling at hotspots** (focus2), and **month**.

To understand what this means, following is a summary of mean value standard error (SE) of different prediction based on significant explanatory variables, followed by barplots for each significant explanatory variables. The mean value summaries are 1) interaction of patrolling itself with patrolling within hotspots area, and 2) month, respectively.

```
## pat focus2 emmean SE df asymp.LCL asymp.UCL
## 0 0 0.494 0.0704 Inf 0.3559 0.632
## 1 0 0.229 0.1430 Inf -0.0508 0.510
## 0 1 0.235 0.0485 Inf 0.1399 0.330
## 1 1 0.265 0.0770 Inf 0.1138 0.415
##
```

```
## Confidence level used: 0.95
```

```
## monthnum emmean SE df asymp.LCL asymp.UCL
## 1 0.0721 0.0171 Inf 0.0386 0.106
## 2 0.0922 0.0178 Inf 0.0573 0.127
## 3 0.1180 0.0178 Inf 0.0830 0.153
## 4 0.1509 0.0173 Inf 0.1171 0.185
## 5 0.1928 0.0172 Inf 0.1590 0.227
## 6 0.2456 0.0214 Inf 0.2037 0.288
## 7 0.3102 0.0336 Inf 0.2443 0.376
## 8 0.3814 0.0525 Inf 0.2785 0.484
## 9 0.4349 0.0613 Inf 0.3146 0.555
## 10 0.4152 NaN Inf NaN NaN
## 11 0.2964 NaN Inf NaN NaN
## 12 0.1593 NaN Inf NaN NaN
##
```

```
## Confidence level used: 0.95
```

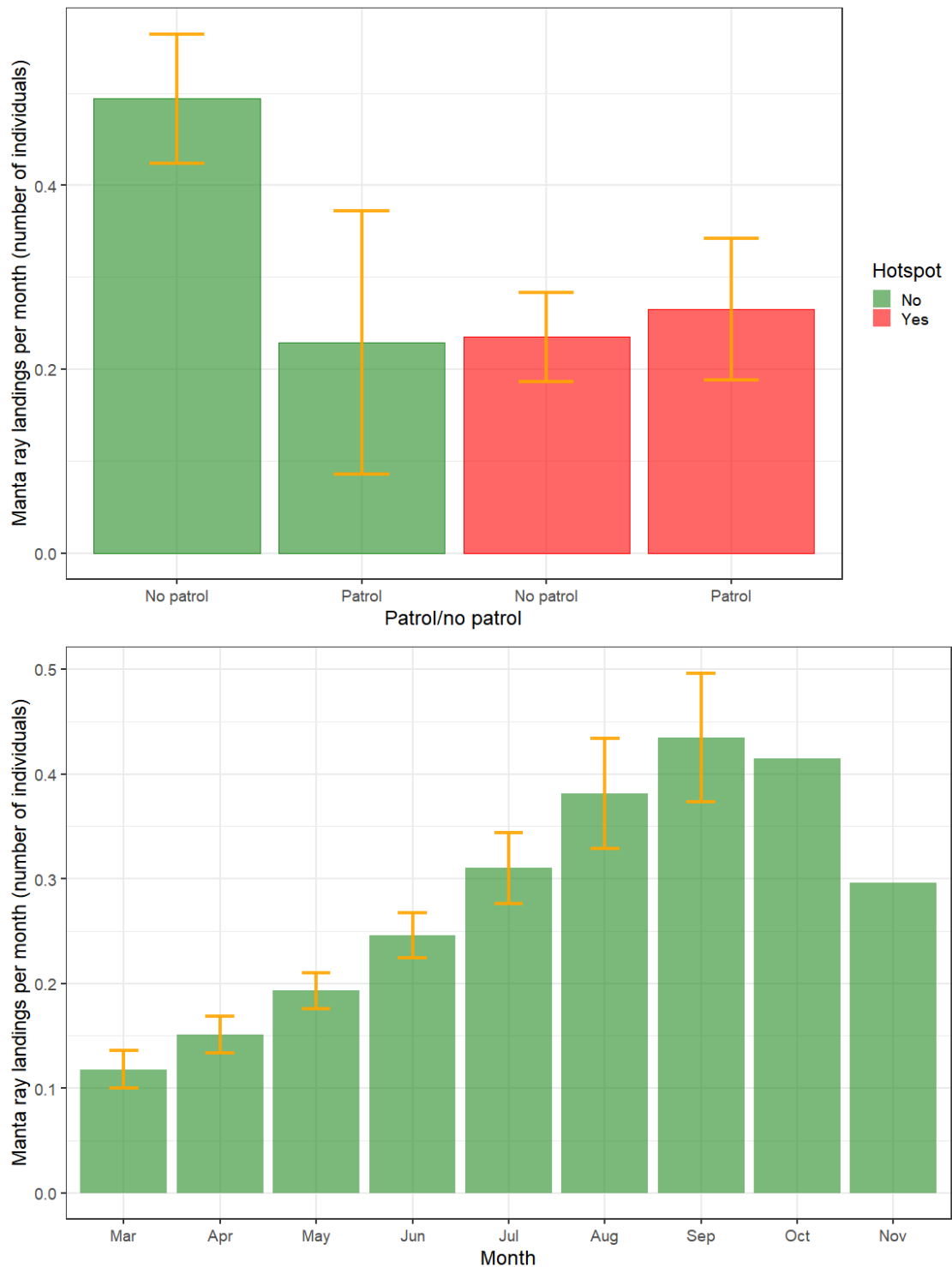



Fig S4.5d. Barplot of mean $\pm$ SE of manta landings based on different explanatory variables.

Top = interaction term of patrol & patrolling on hotspots, bottom = month.

Model using monthly manta catch, include non-fishing season

To have a more general overview of what effect the change of monthly manta landing, we conducted the analysis by aggregating daily landings into monthly landings of manta ray. We first check the data distribution of the monthly manta catch, using a simple histogram.

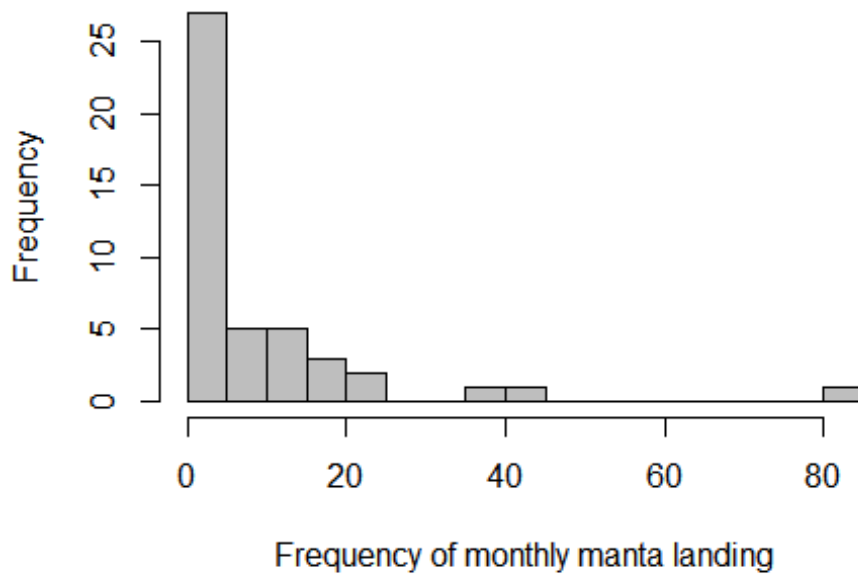


Figure S4.5e. Histogram of monthly manta landing.

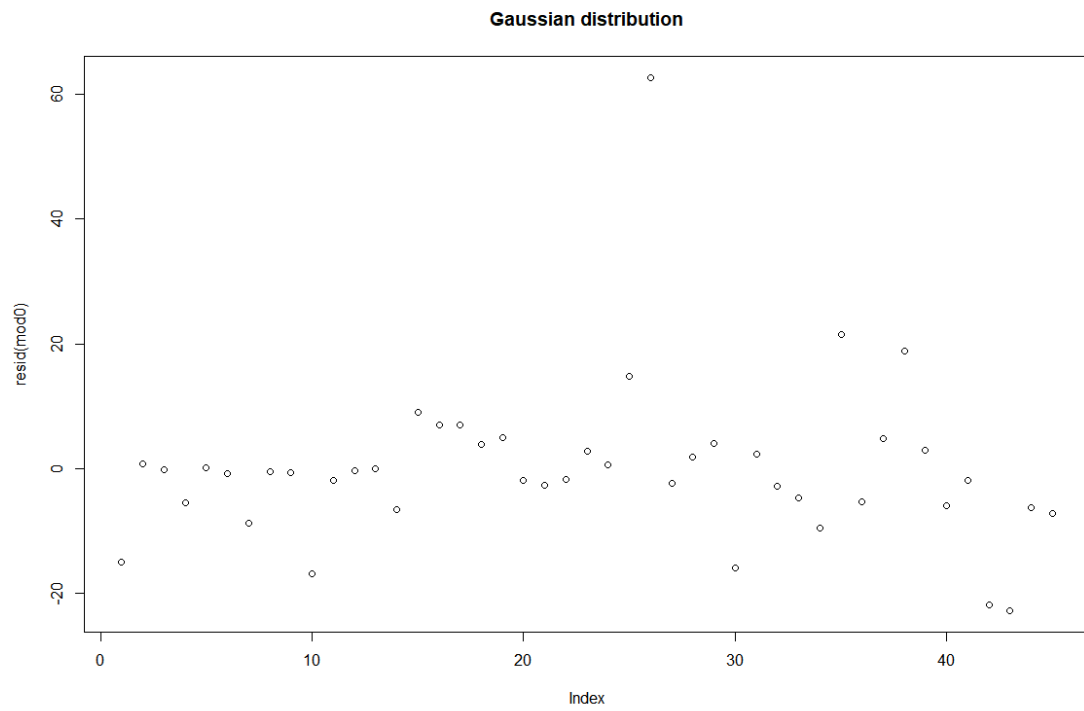
Since the data is integer, we will go through a protocol to evaluate the model using gaussian, poisson, negative binomial (correcting for over dispersion) (Zuur, et al. 2009). The full model is as following:

Table S4.5d. Summary of AIC and %R2 values of models based on different distributions.

Monthly manta landing = *Year* + *Patrol focus* : *Patrol* + *Month*

Model distribution	%R2	AIC	Δ AIC	df
Gaussian	25.14	371.6060	0.0000	7
Poisson	99.80	611.8083	-240.2023	6
Negative binomial	34.80	264.0869	107.5191	7

We also inspect the residual distribution of each model as following:



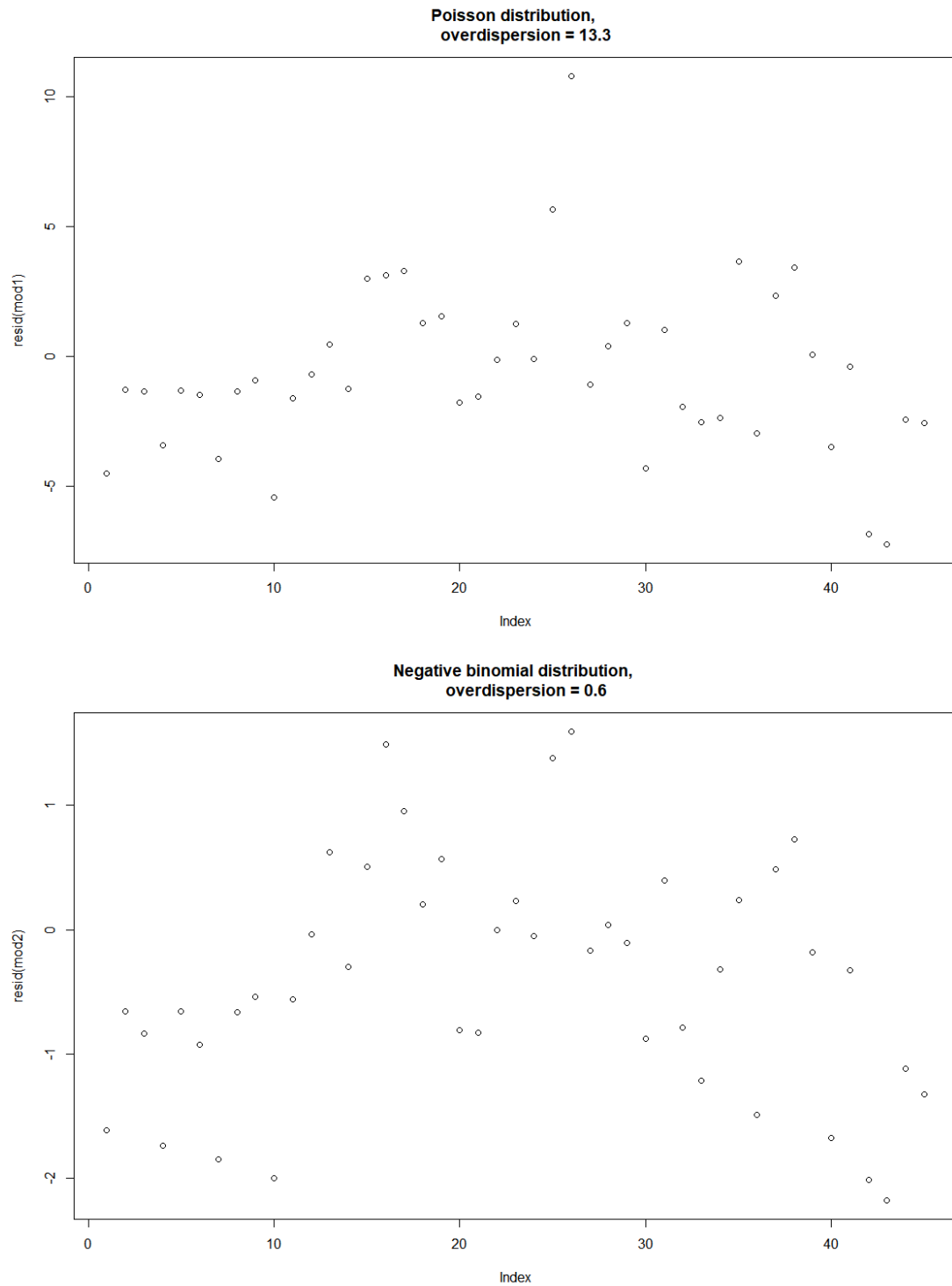


Figure S4.5e. Residual plot of the Gaussian, Poisson, and Negative binomial models (top to bottom).

We found over dispersion within the Poisson model with the lowest AIC found with the negative binomial model. Therefore, we will use the latter to fit our model.

```
##
## Call:
## glm.nb(formula = mantanum ~ monthnum + year + patrol * focus,
## data = ds2, link = "log", init.theta = 0.5874799764)
##
```

```
## Deviance Residuals:
##   Min     1Q   Median     3Q      Max
## -2.1740 -0.9263 -0.3237  0.2291  1.5926
##
## Coefficients:
##             Estimate Std. Error z value Pr(> |z|)
## (Intercept) -1.214e+03  9.787e+02 -1.240 0.214815
## monthnum     6.095e-03  7.748e-02  0.079 0.937296
## year         6.039e-01  4.856e-01  1.243 0.213693
## patrol       -3.648e-01  1.728e-01 -2.112 0.034700 *
## focus        -5.355e+00  1.475e+00 -3.631 0.000282 ***
## patrol:focus  6.414e-01  2.129e-01  3.012 0.002593 **
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## (Dispersion parameter for Negative Binomial(0.5875) family taken to be 1)
##
##   Null deviance: 71.342  on 44  degrees of freedom
## Residual deviance: 47.430  on 39  degrees of freedom
## AIC: 264.09
##
## Number of Fisher Scoring iterations: 1
##
##
##             Theta: 0.587
##             Std. Err.: 0.150
##
## 2 x log-likelihood: -250.087
```

We found only **patrol** (patrol), **patrolling in hotspot** (focus), and **interaction of the two** (patrol:focus) to be the significant variables. Therefore, to find the best fit model, we will conduct a stepwise regression using `drop1` function to ensure what variables need to be dropped.

#Stepwise regression to find best fit model using drop1 function

```
drop1(mod2, test="Chi")

## Single term deletions
##
## Model:
## mantanum ~ monthnum + year + patrol * focus
##           Df Deviance   AIC   LRT Pr(>Chi)
## <none>         47.430 262.09
## monthnum     1  47.433 260.09 0.0025 0.96021
## year         1  48.729 261.39 1.2991 0.25437
## patrol:focus  1  53.247 265.90 5.8167 0.01587 *
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

#drop month

#Re-run model after dropping variable month

```
mod2.b <- glm.nb(mantanum ~ year + patrol*focus, link = "log", data=ds2)
drop1(mod2.b, test="Chi")

## Single term deletions
##
## Model:
```

```
## mantanum ~ year + patrol * focus
##           Df Deviance  AIC   LRT Pr(>Chi)
## <none>           47.407 260.09
## year      1  48.820 259.50 1.4134 0.234494
## patrol:focus 1  56.474 267.16 9.0674 0.002602 **
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

#drop year

#Re-run model after dropping variable year
mod2.c <- glm.nb(mantanum ~ patrol*focus, link = "log", data=ds2)
drop1(mod2.c, test="Chi")

## Single term deletions
##
## Model:
## mantanum ~ patrol * focus
##           Df Deviance  AIC   LRT Pr(>Chi)
## <none>           47.677 259.49
## patrol:focus 1  55.135 264.94 7.4573 0.006318 **
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

#no more variables need to be dropped
```

We found the best fit model to be:

Monthly manta landing = Patrol focus : Patrol

To understand what this means, following is a summary of mean value standard error (SE) of prediction based on significant explanatory variables, followed by barplots for each significant explanatory variables. The mean value summary is based on the only significant variable, interaction of patrolling itself with patrolling within hotspots area.

```
## patrol focus response  SE  df asymp.LCL asymp.UCL
##    0    0  19.943 7.347 Inf   9.6875   41.05
##    1    0  15.423 4.698 Inf   8.4900   28.02
##    0    1   0.379 0.280 Inf   0.0893    1.61
##    1    1   0.503 0.332 Inf   0.1383    1.83
##
## Confidence level used: 0.95
## Intervals are back-transformed from the log scale
```

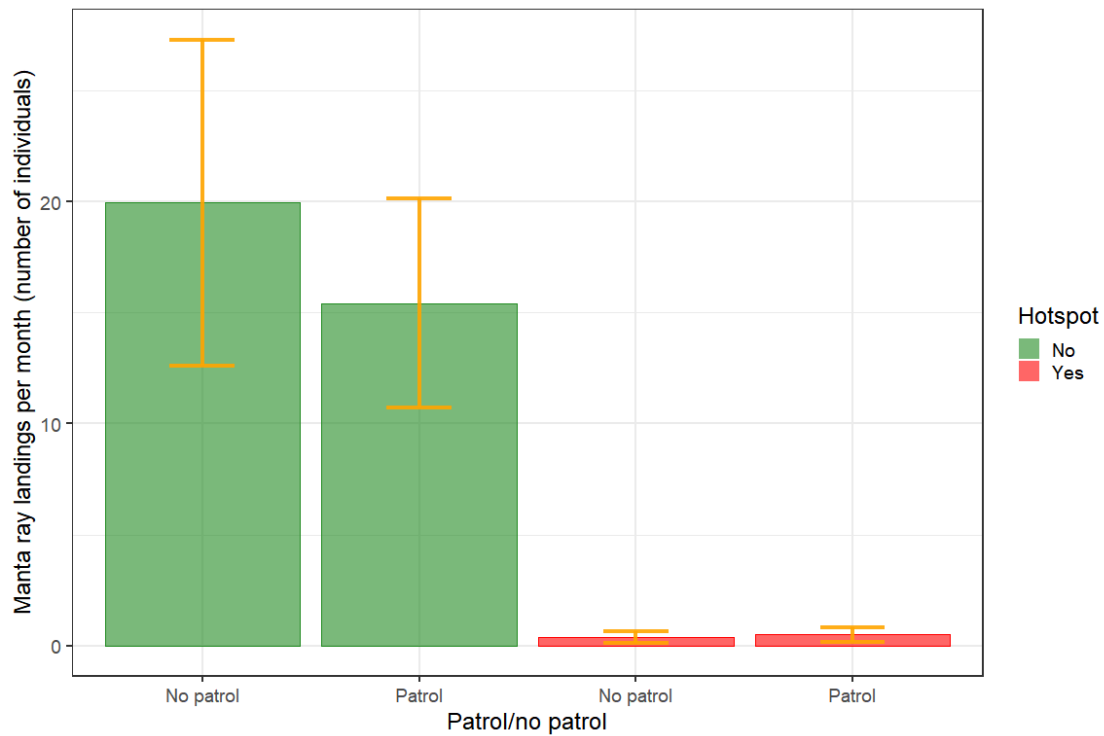


Fig S4.5f. Barplot of mean $\pm$ SE of manta landings based on Interaction term of patrol & patrolling on hotspots.

Model using monthly manta catch, exclude non-fishing season

Similar to part B, we also applied exclusion of non-fishing season within the manta ray monthly landing analysis. In order to do so, we excluded December, January, and February landings. We then conducted the analysis similarly to part C. We first check the data distribution of the monthly manta catch, using a simple histogram.

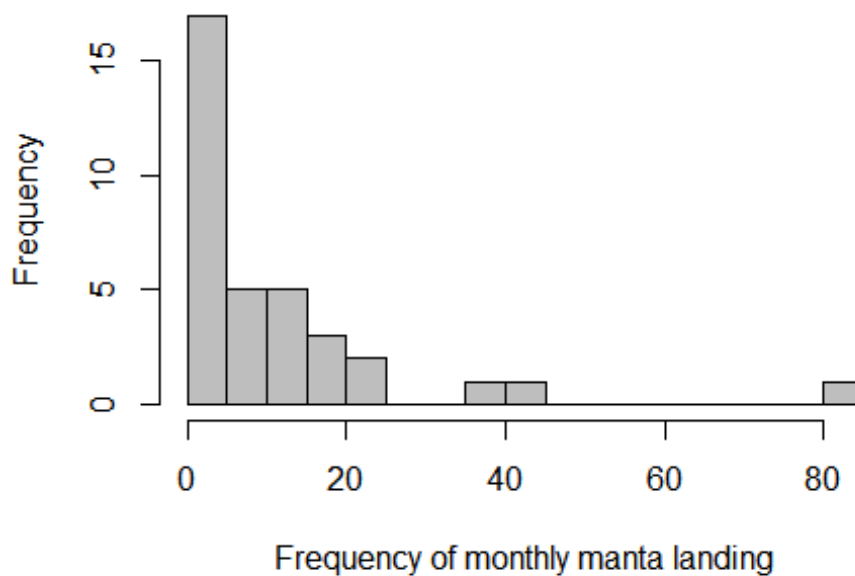


Figure S4.5g. Histogram of monthly manta landing.

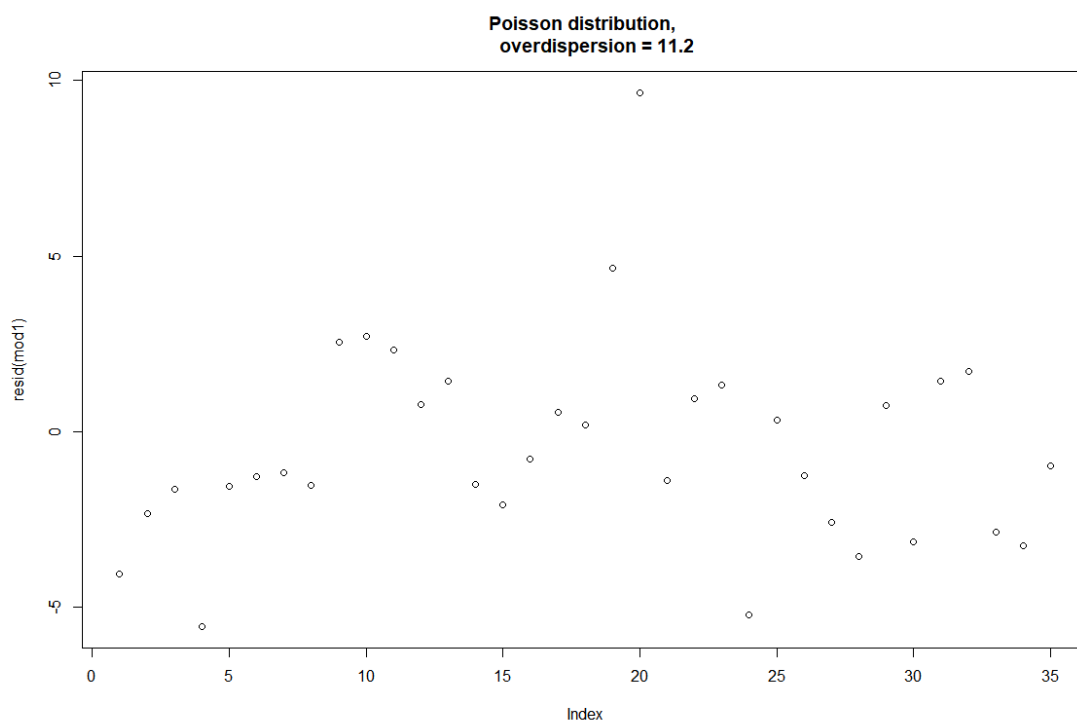
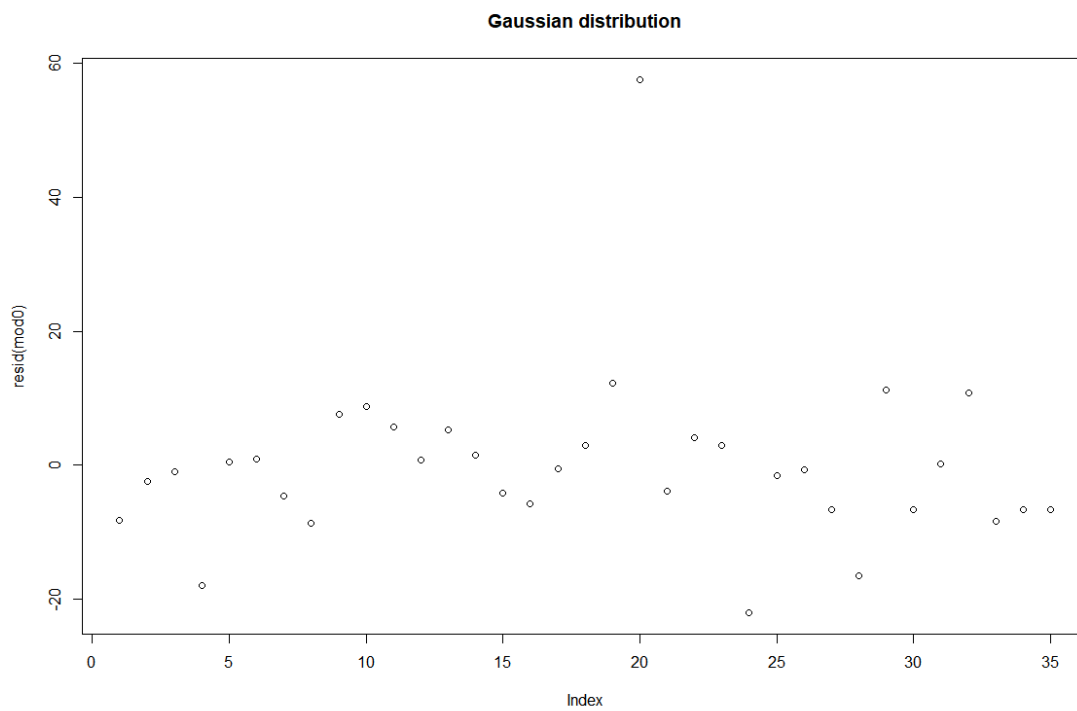
Since the data is integer, we will go through a protocol to evaluate the model using gaussian, poisson, negative binomial (correcting for over dispersion) (Zuur, et al. 2009). The full model is as following:

Table S4.5e. Summary of AIC and %R2 values of models based on different distributions.

$$\text{Monthly manta landing} = \text{Year} + \text{Patrol focus} : \text{Patrol} + \text{Month}$$

Model distribution	%R2	AIC	Δ AIC	df
Gaussian	38.5	290.4101	0.0000	7
Poisson	100.0	421.0082	-130.5981	6
Negative binomial	42.4	233.5535	56.8566	7

We also inspect the residual distribution of each model as following:



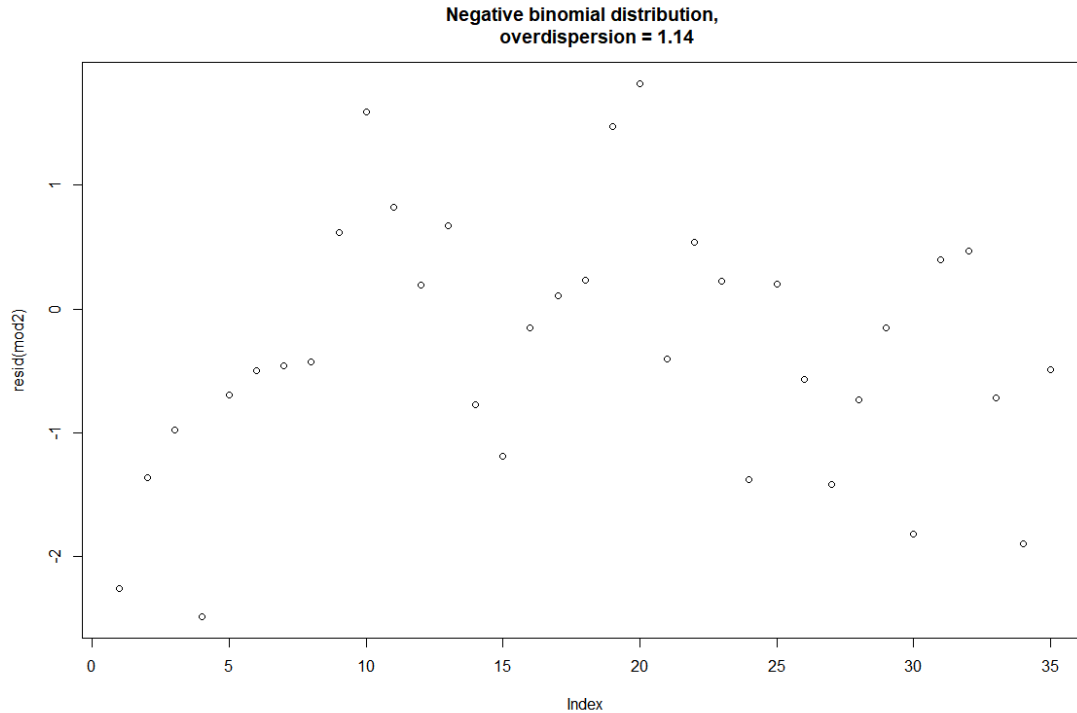


Figure S4.5h. Residual plot of the Gaussian, Poisson, and Negative binomial models (top to bottom).

We found over dispersion within the Poisson model with the lowest AIC found with the negative binomial model. Therefore, we will use the latter to fit our model.

```
##
## Call:
## glm.nb(formula = mantanum ~ monthnum + year + patrol * focus,
## data = ds21, link = "log", init.theta = 1.139908748)
##
## Deviance Residuals:
##   Min       1Q   Median       3Q      Max
## -2.4840 -0.8777 -0.4239  0.3137  1.8198
##
## Coefficients:
##              Estimate Std. Error z value Pr(> |z|)
## (Intercept) -1.432e+03  8.436e+02 -1.697  0.08964 .
## monthnum      7.210e-02  8.716e-02  0.827  0.40815
## year          7.118e-01  4.187e-01  1.700  0.08910 .
## patrol       -3.865e-01  1.495e-01 -2.584  0.00976 **
## focus        -3.756e+00  1.483e+00 -2.533  0.01132 *
## patrol:focus  4.360e-01  2.001e-01  2.178  0.02940 *
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## (Dispersion parameter for Negative Binomial(1.1399) family taken to be 1)
##
##   Null deviance: 66.548  on 34  degrees of freedom
## Residual deviance: 40.243  on 29  degrees of freedom
## AIC: 233.55
##
## Number of Fisher Scoring iterations: 1
```

```

##
##
##      Theta: 1.140
##      Std. Err.: 0.336
##
## 2 x log-likelihood: -219.553

#Stepwise regression to find best fit model using drop1 function
drop1(mod2, test="Chi") #drop month

## Single term deletions
##
## Model:
## mantanum ~ monthnum + year + patrol * focus
##      Df Deviance  AIC  LRT Pr(>Chi)
## <none>      40.243 231.55
## monthnum   1  40.671 229.98 0.4272 0.51337
## year       1  42.795 232.10 2.5513 0.11021
## patrol:focus 1  43.587 232.90 3.3437 0.06746 .
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

#Re-run model after dropping variable month
mod2.b <- glm.nb(mantanum ~ year + patrol*focus, link = "log", data=ds2)
drop1(mod2.b, test="Chi") #drop year

## Single term deletions
##
## Model:
## mantanum ~ year + patrol * focus
##      Df Deviance  AIC  LRT Pr(>Chi)
## <none>      47.407 260.09
## year       1  48.820 259.50 1.4134 0.234494
## patrol:focus 1  56.474 267.16 9.0674 0.002602 **
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

#Re-run model after dropping variable year
mod2.c <- glm.nb(mantanum ~ patrol*focus, link = "log", data=ds2)
drop1(mod2.c, test="Chi") #no more variables need to be dropped

## Single term deletions
##
## Model:
## mantanum ~ patrol * focus
##      Df Deviance  AIC  LRT Pr(>Chi)
## <none>      47.677 259.49
## patrol:focus 1  55.135 264.94 7.4573 0.006318 **
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

```

We found the best fit model to be:

Monthly manta landing = Patrol focus : Patrol

We found only **patrol** (patrol), **patrolling in hotspot** (focus), and **interaction of the two** (patrol:focus) to be the significant variables. Therefore, to find the best fit model, we will conduct a stepwise regression using `drop1` function to ensure what variables need to be dropped.

To understand what this means, following is a summary of mean value standard error (SE) of prediction based on significant explanatory variables, followed by barplots for each significant explanatory variables. The mean value summary is based on the only significant variable, interaction of patrolling itself with patrolling within hotspots area.

```
## patrol focus response SE df asymp.LCL asymp.UCL
## 0 0 19.943 7.347 Inf 9.6875 41.05
## 1 0 15.423 4.698 Inf 8.4900 28.02
## 0 1 0.379 0.280 Inf 0.0893 1.61
## 1 1 0.503 0.332 Inf 0.1383 1.83
##
## Confidence level used: 0.95
## Intervals are back-transformed from the log scale
```

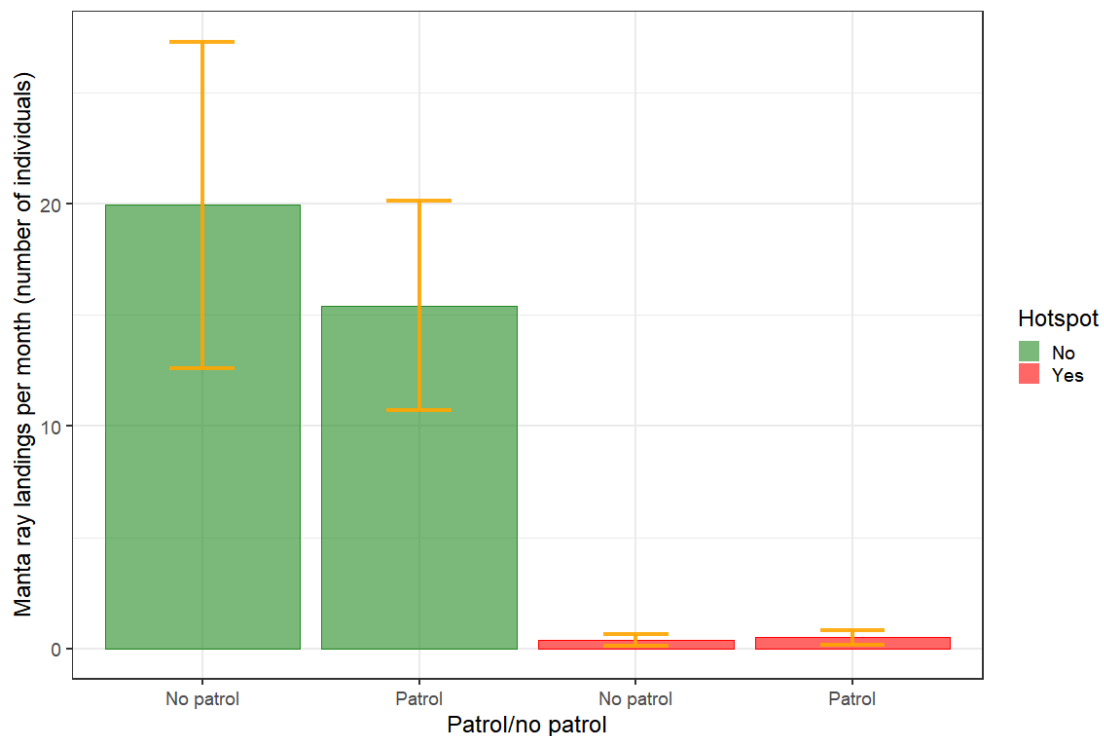


Fig S4.5i. Barplot of mean $\pm$ SE of manta landings based on Interaction term of patrol & patrolling on hotspots.

Events-based analysis

So as not to assume if/when a significant event occurred, we fitted landing occurrences to pre-post models for every year in the time series. We compared AICs of the models with a null model and with each other (Table S7), to identify if/when in the timeline a step-change in landing occurrences may have taken place, assuming that the best fit model provides the best explanation of when a pre-/post- change occurred. The low AIC model for manta ray landing occurrences (2015-2016 (pre) vs 2017-2018 (post)), suggests a step-change in landing occurrences took place at this time with landing occurrences 0.7156 times more likely in pre-2017 than post-2017 ($p = 5.4\text{e-}05$, ***). No similar step change was observed for devil rays, with all models with 3 ΔAIC of each other and the null model (Table S7).

Table S4.5f. Models to identify if/when a step change in manta ray landings occurred

Comparison years	AIC manta		AIC mobula		
Control (landings ~ 1)	938.38		1181		
2015 (pre) vs 2016-2018 (post)	933.61		1180.4		
2015-2016 (pre) vs 2017-2018 (post)	923.64		1183		
2015-2017 (pre) vs 2018 (post)	939.97		1182.7		

Coefficients	Estimate ^b	Std. Error	z value	Pr(> z)	Significance ^a
(Intercept)	-1.639897	0.110947	-14.781	<2.00e-16	***
SpeciesManta	-0.07578	0.15898	-0.477	0.6336	
Pre2017Post2017	0.003924	0.149497	0.026	0.97906	
SpeciesManta:Pre2017Post2017	-0.719478	0.231849	-3.103	0.00191	**

We then modelled all mobulid landings, with species (manta vs. devil ray) and pre- vs post-2017 as explanatory variables (Table S8). The significant negative interaction between species and pre-/post-2017 indicated that there is a significant negative decline for manta ray landings post-2017, which did not occur for devil rays.

Table S4.5g. Model of all mobulid landings comparing pre-2017 vs post-2017 and manta vs =

^a Significance codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

^b Reference level is Pre-2017, SpeciesDevil

Model built with daily landing occurrences as a binary response variable (mobulid landed: 1/0) and intervention (2017) (pre- and post-) and species (manta and devil) as categorical predictor variables

Modelled counterfactual

Table S4.5h. Anova of Generalised additive model for assessing the influence each predictor on manta catch rate estimator. The significance (p-value) of each predictor was based on the Likelihood-ratio test of comparing the full model and the model omitting the respective predictor. Significance predictor assessed by p-value, including marginal significant value (0.1) in *italic* and only significant predictors were included in each model. The proportion of deviance explained (% Dev) represents the importance of each predictor.

Model variable	Targeted catch				Secondary catch			
	Dev (%)	df	χ^2	p (χ^2)	Dev (%)	df	χ^2	p (χ^2)
s(Day)	2.27	3	13.51	<0.01	5.05	3	8.26	<0.05
s(EV)	2.12	2	12.65	<0.01	-	-	-	-
s(Moon)	1.26	3	7.49	<i>0.057</i>	12.43	3	20.30	<0.001
s(SMWS)	4.55	2	27.13	<0.0001	-	-	-	-
s(SPWMP)	1.65	3	9.85	<0.05	5.07	3	8.30	<0.05
s(SSC)	32.91	2	196.1	<0.0001	7.58	2	12.40	<0.01
s(SSHWWS)	3.21	2	19.12	<0.0001	-	-	-	-
s(SSS)	13.23	3	78.82	<0.0001	3.87	3	6.33	<i>0.096</i>
s(SSTA)	3.18	3	18.96	<0.001	-	-	-	-
s(Tide)	2.61	2	15.53	<0.001	-	-	-	-
Deviance explained	67%				34%			

Table S4.5i. Summary of observed and predicted CPUE of mantas from 2015 to 2018.

Targeted catch									Potential change (times)
Year	Raw CPUE (ind.day)				Predicted CPUE (ind.day)				
	Mean	SE	LCL	UCL	Mean	SE	LCL	UCL	
			95%	95%			95%	95%	
2015	0.488	0.113	0.255	0.690	0.485	0.108	0.314	0.749	0.00
2016	0.208	0.056	0.087	0.306	0.108	0.039	0.053	0.221	0.48
2017	0.016	0.007	0.002	0.027	0.460	0.108	0.290	0.730	27.00
2018	0.045	0.014	0.018	0.069	0.836	0.140	0.174	1.256	17.44
Secondary catch									Potential change (times)
Year	Raw CPUE				Predicted CPUE				

2015	0.016	0.007	0.000	0.030	0.228	0.066	0.129	0.402	12.86
2016	0.148	0.031	0.082	0.205	0.114	0.047	0.051	0.258	0.23
2017	0.047	0.012	0.022	0.068	0.143	0.045	0.077	0.266	2.07
2018	0.133	0.028	0.076	0.184	0.133	0.039	0.075	0.236	0.00

List of R packages used for analysis

Knitr:

Yihui Xie (2020). knitr: A General-Purpose Package for Dynamic Report Generation in R. R package version 1.29.

Yihui Xie (2015) Dynamic Documents with R and knitr. 2nd edition. Chapman and Hall/CRC. ISBN 978-1498716963

Yihui Xie (2014) knitr: A Comprehensive Tool for Reproducible Research in R. In Victoria Stodden, Friedrich Leisch and Roger D. Peng, editors, Implementing Reproducible Computational Research. Chapman and Hall/CRC. ISBN 978-1466561595

Dplyr:

Hadley Wickham, Romain François, Lionel Henry and Kirill Müll (2020). dplyr: A Grammar of Data Manipulation. R package version 1.0.0. <https://CRAN.R-project.org/package=dplyr>

DT:

Yihui Xie, Joe Cheng and Xianying Tan (2020). DT: A Wrapper of the JavaScript Library 'DataTables'. R package version 0.14. <https://CRAN.R-project.org/package=DT>

Xtable:

David B. Dahl, David Scott, Charles Roosen, Arni Magnusson and Jonathan Swinton (2019). xtable: Export Tables to LaTeX or HTML. R package version 1.8-4. <https://CRAN.R-project.org/package=xtable>

MASS:

Venables, W. N. & Ripley, B. D. (2002) Modern Applied Statistics with S. Fourth Edition. Springer, New York. ISBN. 0-387-95457-0

pscl/ideal:

Simon Jackman (2020). pscl: Classes and Methods for R Developed in the Political Science Computational Laboratory. United States Studies Centre, University of Sydney. Sydney, New South Wales, Australia. R package version 1.5.5. URL: <https://github.com/atahk/pscl/>

zeroinfl()/hurdle():

Achim Zeileis, Christian Kleiber, Simon Jackman (2008). Regression Models for Count Data in R. Journal of Statistical Software 27(8). URL: <http://www.jstatsoft.org/v27/i08/>.

MuMIn:

Kamil Barton (2020). MuMIn: Multi-Model Inference. R package version 1.43.17. <https://CRAN.R-project.org/package=MuMIn>

lme4:

Douglas Bates, Martin Maechler, Ben Bolker, Steve Walker (2015). Fitting Linear Mixed-Effects Models Using lme4. Journal of Statistical Software, 67(1), 1-48. doi:10.18637/jss.v067.i01.

performance:

Daniel Lüdtke, Dominique Makowski, Philip Waggoner and Indrajeet Patil (2020). performance: Assessment of Regression Models Performance. R package version 0.4.7. <https://CRAN.R-project.org/package=performance>

emmeans:

Russell Lenth (2020). emmeans: Estimated Marginal Means, aka Least-Squares Means. R package version 1.4.8. <https://CRAN.R-project.org/package=emmeans>

vctr:

Hadley Wickham, Lionel Henry and Davis Vaughan (2020). vctr: Vector Helpers. R package version 0.3.1. <https://CRAN.R-project.org/package=vctr>

ggplot2

H. Wickham. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York, 2016.

gridExtra:

Baptiste Auguie (2017). gridExtra: Miscellaneous Functions for "Grid" Graphics. R package version 2.3. <https://CRAN.R-project.org/package=gridExtra>

ggpubr:

Alboukadel Kassambara (2020). ggpubr: 'ggplot2' Based
Publication Ready Plots. R package version 0.4.0.

<https://CRAN.R-project.org/package=ggpubr>

Appendix 2: Supplementary Material for Chapter 5

S5.1 Interview script

Oral consent

[READ BEFORE EACH INTERVIEW & RECORD CONSENT]

“Hello, my name is [REDACTED]. I am an independent research student at [REDACTED] and we are conducting research on how fishers in Indonesia live their lives and make decisions about their fishing practices. We’re particularly interested in fishers’ perceptions of certain species, especially sharks and rays, and we’d like to spend some time discussing this with you. It should only take around an hour of your time.

Would you like to participate? When is a good time to talk?

This is my first visit to collect data, and the information you give us during this meeting will be used to design more specific questionnaires, so that when we come back to village later in the year we can conduct more detailed interviews with other people in the village.

I want to let you know that:

- What is said during this meeting is private.
- The personal information you share with me will not be passed to any third party.
- We won’t directly record your name, or reveal your name to anyone.
- We will not share your answers with other members of the community, the local authorities or any other authorities.
- We will not ask you anything that could get you into trouble.
- Participation is voluntary and you may stop the discussion at any time.

During the interview, I will take notes in my notebook. We may also take photographs, if you give permission. The information that you share with me will be saved in a secure database which can only be accessed with a password. I will analyse the information and the results will be presented as part of my written thesis for my degree qualification. Some of the results may also be published internationally in academic papers, at conferences and on online blogs. At the end of the research, I will return to present my research findings to you. I will not use your name in any reports or publications, unless you insist on the opposite

This study has been reviewed by, and received ethics clearance through, the University of Oxford Central University Research Ethics Committee. If you have a concern about any aspect of this project, you can

speak to me and I will do my best to answer your query. If you remain unhappy or wish to make a formal complaint, I can give you the contact details of the Research Ethics Committee at the University of Oxford.

- Is this ok? (Get verbal agreement from participant)
- Do you have any questions or concerns so far? (Pause here to give participant enough time to think and comment)
- Would you like to continue with the meeting? (Get verbal consent from participant, and tick the interviewee sheet to confirm that verbal consent has been given).
- Do you give me permission to audio record this meeting or take photographs so that we can make sure that we don't miss anything important that you tell us? (Get verbal consent; if anyone objects to being recorded or photographed, do not record the meeting)"

[Once obtained, record consent in the sheet below]

Consent obtained?

Interview number

Interviewer

Village

Opening semi-structured interview

“In this first section, we’re going to start with a general chat about your daily activities and how you make a living...”

SECTION 1. FISHING QUESTIONS:

Firstly, can you tell me a bit of information about yourself...			
1. Which village do you live in?			
2. Are you originally from Aceh Jaya?	Yes	No	If no, where are you from? When did you come to Aceh Jaya?
3. How old are you?			
4. Are you married?	Yes	No	
5. Do you have any children?	Yes	No	If yes how many?
6. When did you leave school/what is your level of education?			
I understand you are a fisher; I'd like to learn more about your fishing activities...			
7. How long have you been a fisher?			
8. Can you remember why you started fishing?			
9. What type of fishing gear do you normally use?			
10. Do you usually fish alone, or as part of a crew?			
11. If as part of a crew, how many people in total?			
12. What is your position in the boat?			
13. Do you own your own boat?			
14. How often do you go fishing?			
15. Are your fishing activities the same all year round, or do you conduct more or less trips to sea at certain			

times of year? Please explain – in which months do you fish more, and which do you fish less?			
16. How long is each trip?			
17. Which species are your main target?			
18. What are your main reasons for fishing?			
19. How much money do you get from fishing?			
20. Do you have any other forms of income?			
21. What is your monthly HHI overall?			
22. Do you enjoy fishing? Why/why not?			
23. Would you like your children to also become fishers?	Yes	No	Why/why not?
24. Have you noticed any changes in fishing or fish catches in recent times? Please explain.			
I'm interested in learning more about some of these species...			
25. Do you know this species? [show picture of wedgefish]	Yes	No	If yes, what do you call this species?
a. Have you ever caught this species?	Yes	No	If yes, how often do you catch this species? [circle one] - Every trip - At least once per week - At least once per month - A few times per year - Less than once per year
b. If you catch them, how many do you usually get per trip?	Min		Max
c. Are there particular places where you're more likely to catch them?			
d. Are there particular times in the year when you're more likely to catch them?			
e. How much do you sell them for?			
26. Do you know this species? [show picture of hammerhead]	Yes	No	If yes, what do you call this species?

a. Have you ever caught this species?	Yes	No	If yes, how often do you catch this species? <u>circle one</u>	
			- Every trip - At least once per week - At least once per month - A few times per year - Less than once per year	
b. If you catch them, how many do you usually get per trip?	Min		Max	
c. Are there particular places where you're more likely to catch them?				
d. Are there particular times in the year when you're more likely to catch them?				
e. How much do you sell them for?				
27. Do you know this species? <u>show picture of whale shark</u>	Yes	No	If yes, what do you call this species?	
a. Have you ever caught this species?	Yes	No	Why/why not?	

SECTION 2. WELL-BEING QUESTIONS:

I'm interested in what it means to have a good life in this village...				
28. Overall, how satisfied are you with your life nowadays?				
Very satisfied	Satisfied	Neutral	Dissatisfied	Very dissatisfied
Please explain...				
a. What makes you satisfied/dissatisfied?				
b. What do you need to be more satisfied with your life?				
I'm interested in your relationship with the ocean...				
29. What comes to mind first when you think of the ocean?				
30. What do you see as positive about the ocean for you?				
31. What do you see as negative about the ocean for you?				
32. To what degree do you agree / disagree with the following statements...				
a. "People have a responsibility to protect the ocean"				
Strongly agree	Agree	Neutral	Disagree	Strongly disagree
Please explain...				

b. "It is my responsibility to protect the ocean"				
Strongly agree	Agree	Neutral	Disagree	Strongly disagree
Please explain...				
c. "It is my responsibility to protect marine animals"				
Strongly agree	Agree	Neutral	Disagree	Strongly disagree
Please explain...				
I'm interested in any local rules or government regulations regarding use of the ocean...				
33. Are you aware of any regulations regarding the ocean? Please describe them				
a. What is your opinion of these regulations?				
I'm interested in your relationship with other fishers in this community...				
34. To what degree do you agree / disagree with the following statements...				
a. "I get along well with other fishers in my community"				
Strongly agree	Agree	Neutral	Disagree	Strongly disagree
Please explain...				
b. "I trust other fishers in my community"				
Strongly agree	Agree	Neutral	Disagree	Strongly disagree
Please explain...				
c. "I talk to other fishers in my community about my fishing practices"				
Strongly agree	Agree	Neutral	Disagree	Strongly disagree
Please explain...				
- If agree, who in particular do you talk to?				
d. "I learn from other fishers in my community about my fishing practices"				
Strongly agree	Agree	Neutral	Disagree	Strongly disagree
Please explain...				
- If agree, who in particular do you learn from?				
e. "I would be more likely to adopt a new fishing practice if other fishers in my community were doing it"				
Strongly agree	Agree	Neutral	Disagree	Strongly disagree
Please explain...				

- If agree, who in particular would you look to?		
35. Are you a member of the Panglima Laot?	Yes	No
a. Please tell me more about it...		

Structured questions about specific behaviour(s) of interest

[READ BEFORE EACH INTERVIEW & ENSURE UNDERSTANDING]

“In this next section, we’re going to talk more about attitudes and behaviours relating to certain species. Many questions in this survey make use of rating scales with 7 places; you are to pick the number that best describes your opinion. For example, if you were asked to rate “The Weather in Aceh Jaya” on such a scale, the 7 places should be interpreted as follows:

good : 1 : 2 : 3 : 4 : 5 : 6 : 7 : bad
 extremely quite slightly neither slightly quite extremely

- If you think the weather in Aceh Jaya is extremely good, then you would select number 1...
- If you think the weather in Aceh Jaya is quite bad, then you would circle select number 6...
- If you think the weather in Aceh Jaya is slightly good, then you would circle the number 3...
- If you think the weather in Aceh Jaya is neither good nor bad, then you would circle the number 4...”

There are no right or wrong responses; we are interested in your personal opinions”

[CHECK UNDERSTANDING BEFORE BEGINNING]

Attitude: instrumental & experiential aspects

Hammerhead (Order: 1 – 2 – 3 (please circle))	
1. If I caught a hammerhead next time I go fishing it would be...	bad: <u> 1 </u> : <u> 2 </u> : <u> 3 </u> : <u> 4 </u> : <u> 5 </u> : <u> 6 </u> : <u> 7 </u> : good
2. If I didn't catch a hammerhead next time I go fishing it would be...	bad: <u> 1 </u> : <u> 2 </u> : <u> 3 </u> : <u> 4 </u> : <u> 5 </u> : <u> 6 </u> : <u> 7 </u> : good
Please explain:	
- What do you see as the advantages? - What do you see at the disadvantages? - What else comes to mind when you think about catching hammerheads?	
3. If I released a hammerhead from my net next time I go fishing it would be...	bad: <u> 1 </u> : <u> 2 </u> : <u> 3 </u> : <u> 4 </u> : <u> 5 </u> : <u> 6 </u> : <u> 7 </u> : good
Please explain:	
- What do you see as the advantages?	

- What do you see at the disadvantages? - What else comes to mind when you think about releasing hammerheads?
Wedgefish (Order: 1 – 2 – 3 (please circle))
4. If I caught a wedgefish next time I go fishing it would be... bad: ___1___: ___2___: ___3___: ___4___: ___5___: ___6___: ___7___: good
5. If I didn't catch a wedgefish next time I go fishing it would be... bad: ___1___: ___2___: ___3___: ___4___: ___5___: ___6___: ___7___: good
Please explain: - What do you see as the advantages? - What do you see at the disadvantages? - What else comes to mind when you think about catching wedgefish?
6. If I released a wedgefish from my net next time I go fishing it would be... bad: ___1___: ___2___: ___3___: ___4___: ___5___: ___6___: ___7___: good
Please explain: - What do you see as the advantages? - What do you see at the disadvantages? - What else comes to mind when you think about releasing wedgefish?
Whale shark (Order: 1 – 2 – 3 (please circle))
7. If I caught a whale shark next time I go fishing it would be... bad: ___1___: ___2___: ___3___: ___4___: ___5___: ___6___: ___7___: good
8. If I didn't catch a whale shark next time I go fishing it would be... bad: ___1___: ___2___: ___3___: ___4___: ___5___: ___6___: ___7___: good
Please explain: - What do you see as the advantages? - What do you see at the disadvantages? - What else comes to mind when you think about catching whale sharks?

9. If I released a whale shark from my net next time I go fishing it would be... bad: ___1___:___2___:___3___:___4___:___5___:___6___:___7___: good
Please explain: - What do you see as the advantages? - What do you see at the disadvantages? - What else comes to mind when you think about releasing whale sharks?

Perceived behavioural control: Capacity and autonomy aspects

Hammerhead (Order: 1 – 2 – 3 (please circle))
10. I am confident that I could catch a hammerhead next time I go fishing, if I wanted to true : ___1___:___2___:___3___:___4___:___5___:___6___:___7___: false
11. It's up to me if I catch a hammerhead next time I go fishing disagree: ___1___:___2___:___3___:___4___:___5___:___6___:___7___: agree
Please explain: - Please list any factors or circumstances that make it easy or enable you to catch hammerheads when you go fishing - Please list any factors or circumstances that make it difficult or prevent you from catching hammerheads when you go fishing
12. I am confident that I could avoid catching hammerheads next time I go fishing, if I wanted to. true : ___1___:___2___:___3___:___4___:___5___:___6___:___7___: false
13. It's up to me if I avoid catching hammerheads next time I go fishing disagree: ___1___:___2___:___3___:___4___:___5___:___6___:___7___: agree
Please explain:

- Please list any factors or circumstances that make it easy or enable you to avoid catching hammerheads - Please list any factors or circumstances that make it difficult or prevent you from avoiding hammerheads
14. I am confident that I could safely release a hammerhead from my net if I wanted to... true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false
15. It is up to me if I safely release a hammerhead from my net next time I catch one... disagree: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : agree
Please explain: - Please list any factors or circumstances that make it easy or enable you to release hammerheads when you go fishing - Please list any factors or circumstances that make it difficult or prevent you from releasing hammerheads when you go fishing

Wedgefish (Order: 1 – 2 – 3 (please circle))
16. I am confident that I could catch a wedgefish next time I go fishing, if I wanted to true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false
17. It's up to me if I catch a wedgefish next time I go fishing disagree: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : agree
Please explain: - Please list any factors or circumstances that make it easy or enable you to catch wedgefish when you go fishing - Please list any factors or circumstances that make it difficult or prevent you from catching wedgefish when you go fishing

18. I am confident that I could **avoid catching wedgefish** next time I go fishing, if I wanted to.

true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false

19. It's up to me if I **avoid catching wedgefish** next time I go fishing

disagree: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : agree

Please explain:

- Please list any factors or circumstances that make it easy or enable you to avoid catching wedgefish
- Please list any factors or circumstances that make it difficult or prevent you from avoiding wedgefish

20. I am confident that I could safely release a **wedgefish** from my net if I wanted to...

true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false

21. It is up to me if I safely release a **wedgefish** from my net next time I catch one...

disagree: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : agree

Please explain:

- Please list any factors or circumstances that make it easy or enable you to release wedgefish when you go fishing
- Please list any factors or circumstances that make it difficult or prevent you from releasing wedgefish when you go fishing

Whale shark (Order: 1 – 2 – 3 (please circle))

22. I am confident that I could catch a **whale shark** next time I go fishing, if I wanted to

true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false

23. It's up to me if I catch a **whale shark** next time I go fishing

disagree: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : agree

Please explain:

- Please list any factors or circumstances that make it easy or enable you to catch whale sharks when you go fishing

- Please list any factors or circumstances that make it difficult or prevent you from catching whale sharks when you go fishing
24. I am confident that I could avoid catching whale shark next time I go fishing, if I wanted to. true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false
25. It's up to me if I avoid catching whale shark next time I go fishing disagree: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : agree
Please explain: - Please list any factors or circumstances that make it easy or enable you to avoid catching whale sharks - Please list any factors or circumstances that make it difficult or prevent you from avoiding whale sharks
26. I am confident that I could safely release a whale shark from my net if I wanted to... true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false
27. It is up to me if I safely release a whale shark from my net next time I catch one... disagree: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : agree
Please explain: - Please list any factors or circumstances that make it easy or enable you to release whale sharks when you go fishing - Please list any factors or circumstances that make it difficult or prevent you from releasing whale sharks when you go fishing

Perceived norm: descriptive aspects

Hammerhead (Order: 1 – 2 – 3 (please circle))
28. Most fishers in this community catch hammerheads true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false

29. Most fishers in this community do not release hammerheads from their net if they catch them
true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false
Wedgefish (Order: 1 – 2 – 3 (please circle))
30. Most fishers in this community catch wedgefish
true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false
31. Most fishers in this community do not release wedgefish from their net if they catch them
true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false
Whale sharks (Order: 1 – 2 – 3 (please circle))
32. Most fishers in this community catch whale sharks
true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false
33. Most fishers in this community do not release whale sharks from their net if they catch them
true : __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : false

Perceived norm: injunctive aspects

Hammerhead (Order: 1 – 2 – 3 (please circle))
34. Most people in this village approve of me catching hammerheads :
Agree: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : disagree
Please explain: - Please list any individuals or groups who approve or think you <i>should</i> catch hammerheads - Please list any individuals or groups who disapprove or think you <i>should not</i> catch hammerheads
35. Most people in this village do not approve of me releasing hammerheads from my net if I catch them
Agree: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : disagree
Please explain: - Please list any individuals or groups who approve or think you <i>should release</i> hammerheads - Please list any individuals or groups who disapprove or think you <i>should not release</i> hammerheads
Wedgefish (Order: 1 – 2 – 3 (please circle))
36. Most people in this village approve of me catching wedgefish :
Agree: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : disagree
Please explain:

- Please list any individuals or groups who approve or think you <i>should</i> catch wedgefish - Please list any individuals or groups who disapprove or think you <i>should not</i> catch wedgefish
37. Most people in this village do not approve of me releasing wedgefish from my net if I catch them Agree: ___1___ : ___2___ : ___3___ : ___4___ : ___5___ : ___6___ : ___7___ : disagree
Please explain: - Please list any individuals or groups who approve or think you <i>should release wedgefish</i> - Please list any individuals or groups who disapprove or think you <i>should not release</i> wedgefish
Whale sharks (Order: 1 – 2 – 3 (please circle))
38. Most people in this village approve of me catching whale sharks: Agree: ___1___ : ___2___ : ___3___ : ___4___ : ___5___ : ___6___ : ___7___ : disagree
Please explain: - Please list any individuals or groups who approve or think you should catch whale sharks - Please list any individuals or groups who disapprove or think you should not catch whale sharks
39. Most people in this village do not approve of me releasing whale sharks from my net if I catch them Agree: ___1___ : ___2___ : ___3___ : ___4___ : ___5___ : ___6___ : ___7___ : disagree
Please explain: - Please list any individuals or groups who approve or think you should release whale sharks - Please list any individuals or groups who disapprove or think you should not release whale sharks

Intention

Hammerhead (Order: 1 – 2 – 3 (please circle))
40. I intend to catch a hammerhead next time I go fishing: Agree: ___1___ : ___2___ : ___3___ : ___4___ : ___5___ : ___6___ : ___7___ : disagree
41. I do not intend to release a hammerhead from my net next time I catch one Agree: ___1___ : ___2___ : ___3___ : ___4___ : ___5___ : ___6___ : ___7___ : disagree
Wedgefish (Order: 1 – 2 – 3 (please circle))

42. I intend to catch a **wedgefish** next time I go fishing:

Agree: __1__: __2__: __3__: __4__: __5__: __6__: __7__: disagree

43. I **do not** intend to **release a wedgefish** from my net next time I catch one

Agree: __1__: __2__: __3__: __4__: __5__: __6__: __7__: disagree

Whale sharks (Order: 1 – 2 – 3 (please circle))

44. I intend to catch a **whale shark** next time I go fishing:

Agree: __1__: __2__: __3__: __4__: __5__: __6__: __7__: disagree

45. I **do not** intend to **release a whale shark** from my net next time I catch one

Agree: __1__: __2__: __3__: __4__: __5__: __6__: __7__: disagree

Past behaviour

Hammerhead (Order: 1 – 2 – 3 (please circle))

46. In the past three months I have caught **hammerhead** whilst fishing:

false: __1__: __2__: __3__: __4__: __5__: __6__: __7__: true

47. In the past three months I **have not** **released any hammerhead** from my net:

false: __1__: __2__: __3__: __4__: __5__: __6__: __7__: true

Wedgefish (Order: 1 – 2 – 3 (please circle))

48. In the past three months I have caught **wedgefish** whilst fishing:

false: __1__: __2__: __3__: __4__: __5__: __6__: __7__: true

49. In the past three months I **have not** **released any wedgefish** from my net:

false: __1__: __2__: __3__: __4__: __5__: __6__: __7__: true

Whale sharks (Order: 1 – 2 – 3 (please circle))

50. In the past three months I have caught **whale shark** whilst fishing:

false: __1__: __2__: __3__: __4__: __5__: __6__: __7__: true

51. In the past three months I **have not** released any whale shark from my net:

false: __1__ : __2__ : __3__ : __4__ : __5__ : __6__ : __7__ : true

Closing statement

Closing statement: Thank you for participating in our research. Do you have any questions for us?

Appendix 3: Supplementary Material for Chapter 6

S6.1 The relative influences of each management-relevant predictor on catch risk

Predictor	Taxa	n_hit	n_miss	mean_CV_ROC	se_CV_ROC	Contribution
Month	Alopias spp.	255	1001	0.788	0.012	32.548
Month	Carcharhinus falciformis	784	472	0.87	0.006	22.71
Month	Carcharhinus obscurus	377	879	0.774	0.019	39.056
Month	Isurus spp.	297	959	0.807	0.019	25.631
Month	Mobula spp.	88	1168	0.718	0.021	21.315
Month	Rhynchobatus australiae	268	988	0.831	0.011	18.138
Month	Sphyrna spp.	686	570	0.632	0.017	32.583
Effort - days	Alopias spp.	255	1001	0.788	0.012	14.001
Effort - days	Carcharhinus falciformis	784	472	0.87	0.006	7.475
Effort - days	Carcharhinus obscurus	377	879	0.774	0.019	8.29
Effort - days	Isurus spp.	297	959	0.807	0.019	8.692
Effort - days	Mobula spp.	88	1168	0.718	0.021	20.368
Effort - days	Rhynchobatus australiae	268	988	0.831	0.011	11.304
Effort - days	Sphyrna spp.	686	570	0.632	0.017	10.902
Effort - hooks	Alopias spp.	255	1001	0.788	0.012	10.813
Effort - hooks	Carcharhinus falciformis	784	472	0.87	0.006	8.465
Effort - hooks	Carcharhinus obscurus	377	879	0.774	0.019	11.587
Effort - hooks	Isurus spp.	297	959	0.807	0.019	15.401
Effort - hooks	Mobula spp.	88	1168	0.718	0.021	27.206
Effort - hooks	Rhynchobatus australiae	268	988	0.831	0.011	16.37
Effort - hooks	Sphyrna spp.	686	570	0.632	0.017	23.357
Gear type	Alopias spp.	255	1001	0.788	0.012	24.484
Gear type	Carcharhinus falciformis	784	472	0.87	0.006	5.826
Gear type	Carcharhinus obscurus	377	879	0.774	0.019	11.615
Gear type	Isurus spp.	297	959	0.807	0.019	32.871
Gear type	Mobula spp.	88	1168	0.718	0.021	0.377
Gear type	Rhynchobatus australiae	268	988	0.831	0.011	38.03
Gear type	Sphyrna spp.	686	570	0.632	0.017	1.923
Fishing depth	Alopias spp.	255	1001	0.788	0.012	14.336
Fishing depth	Carcharhinus falciformis	784	472	0.87	0.006	53.395
Fishing depth	Carcharhinus obscurus	377	879	0.774	0.019	27.583
Fishing depth	Isurus spp.	297	959	0.807	0.019	13.84
Fishing depth	Mobula spp.	88	1168	0.718	0.021	23.006

Fishing depth	Rhynchobatus australiae	268	988	0.831	0.011	11.492
Fishing depth	Sphyrna spp.	686	570	0.632	0.017	24.398
Fishing zone	Alopias spp.	255	1001	0.788	0.012	3.817
Fishing zone	Carcharhinus falciformis	784	472	0.87	0.006	2.129
Fishing zone	Carcharhinus obscurus	377	879	0.774	0.019	1.869
Fishing zone	Isurus spp.	297	959	0.807	0.019	3.564
Fishing zone	Mobula spp.	88	1168	0.718	0.021	7.729
Fishing zone	Rhynchobatus australiae	268	988	0.831	0.011	4.666
Fishing zone	Sphyrna spp.	686	570	0.632	0.017	6.836

S6.2 The relative influences of each management-relevant predictor on profit

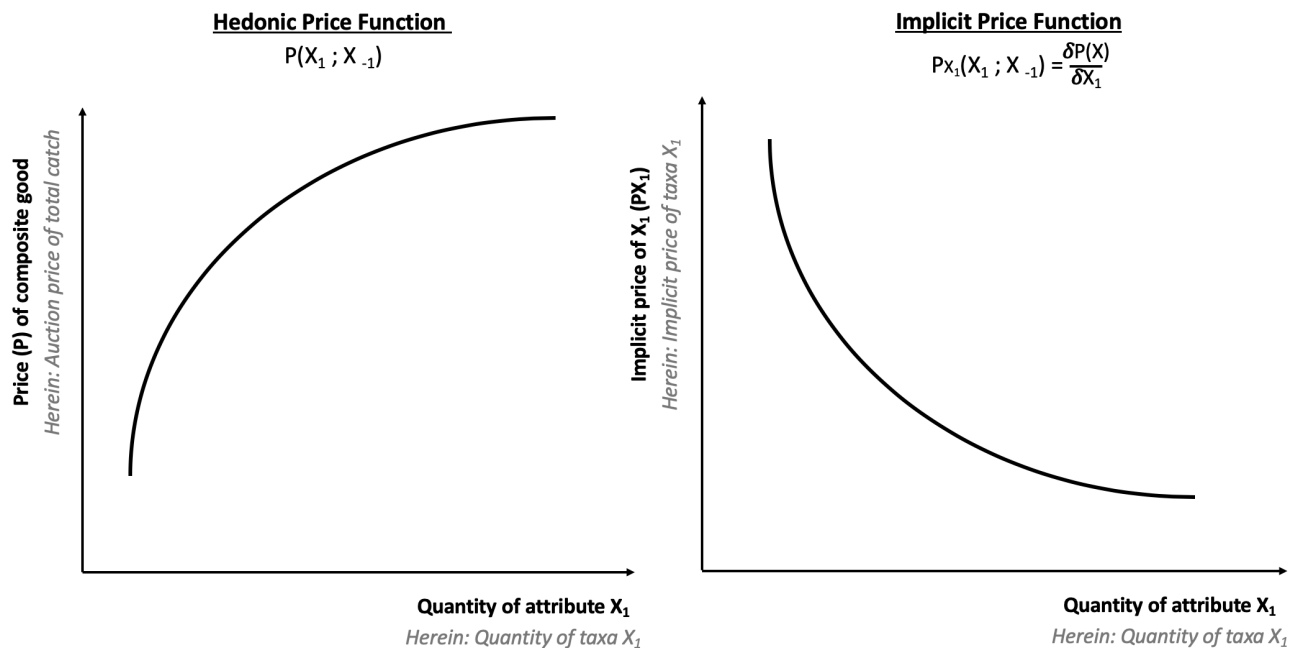
Predictor	mean_CV_cor	se_CV_cor	contribution
Month	0.389	0.041	18.085
Effort - days	0.389	0.041	4.52
Effort - hooks	0.389	0.041	6.136
Gear type	0.389	0.041	27.133
Fishing depth	0.389	0.041	32.725
Fishing zone	0.389	0.041	11.402

Appendix 4: Supplementary Material for Chapter 7

S7.1. Data collected during daily landings monitoring at Tanjung Luar

Type of data	Data	Description
Operational	Vessel ID	Unique name for each vessel
	Date	Date that catch was landed and recorded
	Season	Categorical variable denoting the approximate onset of the seasons (East: June—September, West: December—March, Transition I: April—May, Transition II: October— November)
	Engine power	Engine horsepower of the vessel
	Trip length	Number of days spent at sea
	Fishing ground	Broad descriptive area of fishing grounds visited during trip
	Gear type	Categorical variable denoting whether longline is pelagic (i.e. surface) or demersal (i.e. bottom)
	Sets	Number of times gear was deployed during the trip
	Hook number	Number of hooks used on the gear
	Depth	Depth at which gear was deployed
Catch composition	Species	Visual identification based on White et al. (2006)
	Length	Total length (TL) of the individual (cm), for selachii
	Width	Disc width (DW) of the individual (cm), for batoids
	Weight	Total weight in kg (collected a sample only)
	Sex	Male/Female based on presence/absence of claspers
Economic	Catch auction price	The total final auction price for the whole catch
	Trip operating cost	The cost of the entire fishing trip, including fuel, ice, labour and supplies

S7.2 Schematic of the hedonic price function and implicit price function



Where X_1 is an attribute of a heterogeneous composite good, and X_{-1} is a vector containing the levels of all other attributes apart from X_1 . The semi-colon indicates that these other attributes are held constant. In this study, the composite good is shark catches, sold for price P , while the attributes are the different shark taxa within the catch. The implicit price function for attribute X_1 is the derivative of the hedonic price function with respect to X_1 . As such, $\delta P / \delta X_1$ gives the implicit marginal price of X_1 , all other attributes held equal. Note that in this study we use a linear functional form to estimate the hedonic price function.

S7.3 Summary of modelling process and conclusions

Model	AIC	Adjusted R ²
Total catch volume alone as explanatory variables		
lm(Price ~ TC)	9395.3	0.3017
lm(Price ~ TW)	9418.5	0.2717
Clade and total catch as explanatory variables		
lm(Price ~ Shark(n) + Ray(n) + TW)	9365.9	0.3402
lm(Price ~ Shark(kg) + Ray(kg) + TC)	9374.3	0.3302
lm(Price ~ Shark(n) + Ray(n))	9377.3	0.3253
lm(Price ~ Shark(kg) + Ray(kg))	9419.9	0.2712
<i>Conclusion: Number of individuals (n) is a better unit for explaining the influence of species composition than weight (kg), which also makes intuitive sense since the fins (sharks) or gills (mobulids) are the highest value body parts in commercial trade, their numbers (and therefore the absolute number of valuable sharks/ rays) are likely more important than their total weight. Total weight (TW) is more important as a control variable than total catch (TC).</i>		
Family as explanatory variables		
lm(Price ~ family vector + TW)	8942.8	0.4057
<i>Conclusion: More taxonomic specificity in the explanatory variables improved the explanatory power of the model</i>		
More species-specific models, and control variables		
lm(Price ~ species vector + TEMP + TW)	8444.4	0.6296
lm(Price ~ species vector + TEMP)	8499.4	0.5877
lm(Price ~ species vector + TW)	8528.5	0.5592
lmer(Price ~ species vector + TEMP + TW + (1 V-ID))	8274.1	0.6520
<i>Conclusion: More taxonomic specificity in the explanatory variables improved the explanatory power of the model. Controlling for temporal variation, TW and vessel ID as random effect improves explanatory power of the model</i>		
Null model		
Price ~ 1	9213.9	-
<i>Conclusion: Taxonomic variables explain the data better than no trend/ chance</i>		

Key:

TC = total catch

TW = total weight

(c) = unit as number of individuals (N)

(w) = unit as weight (kg)

Family vector includes Alopidae, Carcharhinidae, Dasyatidae, Hexanidae, Lamnidae, Mobulidae, Rhinidae Sphyrnidae

Species vector includes Alopidae, Silvertip shark, Silky shark, Dusky shark, Tiger shark, Jenkins Whipray, bottlenose wedgefish, Lamnidae, Mobulidae, Sphyrnidae and other carcarhinids (carcharhinus plumbeus, c. sorrah, c. limbatus, c. amblyrhynchos, c. plumbeus, prionace glauca)

lm = linear model

lmer = linear mixed-effects model

S7.4 Summary of diagnostic tests for model assumptions and robustness

Assumption	Diagnostic test	Result
Linearity	Residuals vs fitted plot (visual inference)	✓
Normality of residuals	Q-Q plot (visual inference)	✓
Homogenous variance	Scale – Location plot (visual inference)	✓
	Breusch-Pagan test	✓ (P>0.05)
Influence/sensitivity to data	Residuals vs. Leverage plot visual inference)	✓
Catch/weight is exogenously determined (i.e. not simultaneously determined with price)	Granger Causality	✓ (P>0.05)
	Durban-Wu-Hausman	✓ (P>0.05)
Panel effect (auto-correlation/homogenous variance across entities)	Breusch-Pagan Lagrange multiplier (LM)	✓ (P>0.05) there is no significant panel effect
Fixed vs. random effects for vessel ID	Hausman test	✓ (P>0.05) can use vessel ID as random effect

S7.5 Summary of welfare estimate calculation

Welfare measure estimates followed methods from Taylor (2003) and Day (2001). We used the estimated hedonic price function (i.e. the model in Table 3) to predict prices per trip/transaction, based on the mean values of the explanatory variables for each vessel and season for 2018 alone (average data). We used these price predictions to approximate the ‘current’ total annual welfare across the fishery, by multiplying by the number of trips per vessel per season, and summing across all vessels (current welfare).

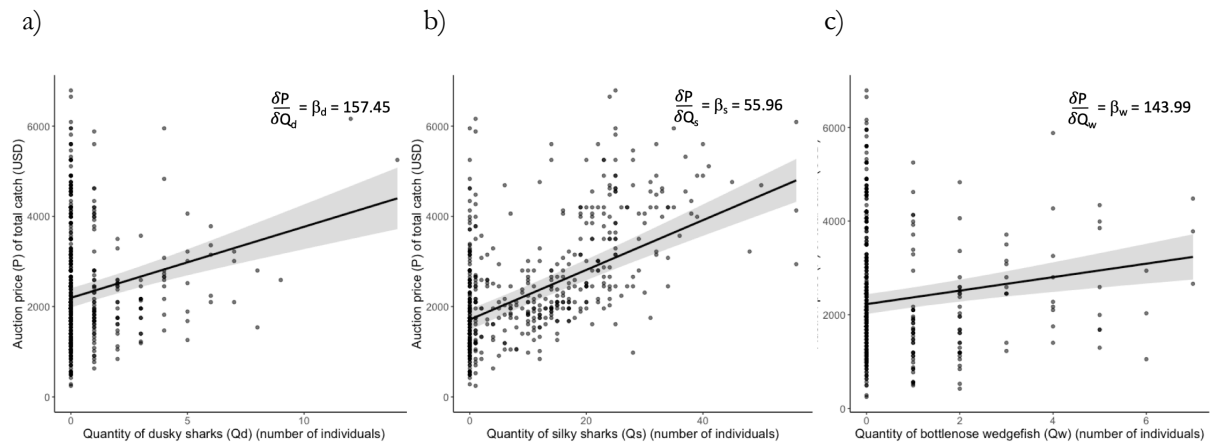
The new states were based on three plausible policy scenarios to restrict supply of bottlenose wedgefish, dusky sharks and silky sharks. We fed dummy data into the estimated hedonic price function (i.e. the model in Table 3) to predict prices per trip/transaction under these scenarios:

1. For the bottlenose wedgefish scenario, we took the average data, made all values for wedgefish catches zero, and reduced the TW by the number caught in the average data multiplied by the average estimated weight of bottlenose wedgefish caught and sold in the fishery.
2. For the dusky shark scenario, we took the average data, made all values for dusky shark catches zero, and reduced the TW by the number caught in the average data multiplied by the average estimated weight of dusky sharks caught and sold in the fishery.
3. Finally, for the silky shark scenario, we took the average data, made reduced all catch values for silky shark by 33%, and reduced the TW by the number of silky sharks caught in the average data multiplied by the average estimated weight of silky sharks caught and sold in the fishery, multiplied by 33%

We used these price predictions to approximate the future total annual welfare under each state, by multiplying by the number of trips per vessel per season, and summing across all vessels. We then estimated ΔW associated with each scenario, in comparison to the current state.

S7.6 Hedonic price functions for selected taxa

Dusky sharks (a), silky sharks (b), and bottlenose wedgefish (c), which indicate how much total catch price changes as the quantity of each taxon changes, holding all other attributes constant. The solid black line indicates the estimate, while the grey bands indicate confidence intervals at 95%. Note these are partial dependency plots, while the raw data points reflect the influence of all variables in the model. Large variation at quantity zero reflects the significant influence of the control variables.



Appendix 5: Supplementary Material for Chapter 8

S8.1 Scenario Interview Scripts

Notes on implementation

- You must adhere to all HEALTH & SAFETY guidelines
- You must adhere to the ETHICS PROCEEDURE and ensure FREE PRIOR & INFORMED CONSENT (see below) before each interview
- We will randomise the order of species, scenarios and CV ladders [detailed instructions in each script]
- We will ensure each fisher knows exactly which species we are discussing, by showing them photographs before each interview

Oral consent

[READ BEFORE EACH INTERVIEW & RECORD CONSENT]

“Hello, my name is _____. I am an independent research student working with Oxford University and IPB and we are conducting research on how fishers in Indonesia live their lives and make decisions about their fishing practices. We’re particularly interested in fishers’ perceptions of certain species, especially sharks and rays, and we’d like to spend some time discussing this with you. It should take around an hour of your time. Would you like to participate? Is now a good time to talk?

I want to let you know that:

- What is said during this meeting is private.
- The personal information you share with me will not be passed to any third party.
- We won’t directly record your name, or reveal your name to anyone.
- We will not share your answers with other members of the community, the local authorities or any other authorities.
- We will not ask you anything that could get you into trouble.
- Participation is voluntary and you may stop the discussion at any time.

During the interview, I will take notes and we may also take photographs, if you give permission. The information that you share with me will be saved in a secure database which can only be accessed with a password. I will analyse the information and the results will be presented as part of a written thesis for a degree qualification. Some of the results may also be published internationally in academic papers, at conferences and on online blogs. At the end of the research, I will return to present my research findings to you. I will not use your name in any reports or publications, unless you insist on the opposite

This study has been reviewed by, and received ethics clearance through, the University of Oxford Central University Research Ethics Committee. If you have a concern about any aspect of this project, you can speak to me and I will do my best to answer your query. If you remain unhappy or wish to make a formal complaint, I can give you the contact details of the Research Ethics Committee at the University of Oxford.

- Is this ok? (Get verbal agreement from participant)

- Do you have any questions or concerns so far? (Pause here to give participant enough time to think and comment)
- Would you like to continue with the meeting? (Get verbal consent from participant, and tick the interviewee sheet to confirm that verbal consent has been given).
- Do you give me permission to audio record this meeting or take photographs so that we can make sure that we don't miss anything important that you tell us? (Get verbal consent; if anyone objects to being recorded or photographed, do not record the meeting)"

[Once obtained, record consent below]

Consent obtained?

Interview number

Interviewer

Tanjung Luar

Pre-survey

Type of shark fisher	Bottom LL		Surface LL
Years as shark fisher			
Current vessel			
Time on that vessel			
Position in vessel			
In general, how 'democratic' is the decision-making on your vessel? For example, is there joint decision-making between everyone in terms of when you go and where you fish, or does the skipper make all decisions?	Not democratic (skipper makes all decisions)	Somewhat democratic (everyone has a say, skipper makes final decision)	Very democratic (everyone has equal say)
Can you personally influence what happens during a fishing trip? Does your opinion make a difference?	I have no influence (someone else makes most/all decisions)	I have some influence (I contribute along with others)	I have a large influence (I make most/all decisions)
When you put the line in the water to soak, to what degree is it possible to choose which species you catch?	We cannot choose (the species composition of our catch is totally random)	We have some ability to choose (we can change the species composition of our catch a little by...)	We can easily choose (we can choose which species we want to catch by...)
If fishers can decide/influence, please explain how			
Species ID verified with photos?	Wedgefish		Hammerheads
How many wedgefish did your vessel land in your last fishing trip? How did that compare to your usual catches during this season?			

How many hammerheads did your vessel land in your last fishing trip? How did that compare to your usual catches during this season?	
---	--

INSTRUCTIONS: Start with WF for 50% of fishers, and HH for 50% of fishers.

We will present fishers with four scenarios for each of the two priority taxa, where Scenario 1 (BAU) should always be presented first, and the order of Scenarios 2 and 3 should be randomised so that 50% of participants receive Scenario 2 second, and Scenario 3 third, while 50% of participants receive Scenario 3 second and Scenario 2 third:

1. BAU (Business as usual)
2. Carrot (positive incentives)
3. Stick (negative incentive)
4. Non-monetary carrot

Within scenarios 2 and 3 there is a contingent valuation question with two options (A & B). For 50% of the respondents, option A should be presented and for 50% of respondents option B should be presented.

Scenario 1. BAU [ALWAYS ASK THIS QUESTION FIRST]

In the next year, I want you to imagine that everything remains the same as it is now. Shark fishing continues as it currently is, there are no further regulations, prices remain the same etc...

1. Under this scenario, what changes, if any, would you expect in the number of wedgefish/ hammerheads *[delete as appropriate]* your vessel lands after each trip?

Decrease

No Change

Increase

2. *[If changes]* Why would this change in landings happen?

3. *[If changes]* Would you expect any changes in your personal fishing behaviour? What changes/ what would you do?

4. *[If changes]* Would you expect any changes in your household income? What changes?

Large negative changes in HHI	Moderate negative changes in HHI	No changes	Moderate positive changes in HHI	Large positive changes in HHI
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5. *How fair are the current policies for you and your household?*

Very unfair	Unfair	Neutral	Fair	Very fair
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Please explain:

Scenario 2. Carrot (positive incentive) [ASK THIS QUESTION SECOND FOR 50% OF PARTICIPANTS, AND THIRD FOR 50% OF PARTICIPANTS]

I want you to imagine that the government introduces a **per trip monetary incentive** for each vessel, so that each vessel receives **a monetary payment if they land zero wedgefish/hammerheads** [delete as appropriate]. You can choose to divide the payment between the crew and boat owner however you wish. Everything else will remain the same, you will still continue shark fishing for most species as usual, prices will remain the same as they are now and there will be no further regulations except there will be a payment to the vessel if you return from trip and land zero wedgefish/hammerheads.

Contingent valuation

Payment value (IDR) per vessel per trip
100,000
125,000
250,000
500,000
750,000
1,000,000
1,500,000
2,000,000
3,000,000
5,000,000
8,000,000
12,000,000
15,000,000
20,000,000

Firstly, I want you to think about how large this payment would need to be in order for your vessel to reduce landings of wedgefish/hammerheads [delete as appropriate] to zero. Let's start with... [for 50% ask a, for 50% ask b]

- a) **IDR 100,000. If your vessel was paid IDR 100,000 per trip** if you came back with zero wedgefish/hammerheads, would you come back with zero, or fish as usual? If *fish as usual*, put a line through the amount and keep going until the respondent says "*we would come back with zero if we were paid this amount*".
- b) **If your vessel was paid IDR 20,000,000 per trip** if you came back with zero wedgefish/hammerheads would you definitely accept IDR 20,000,000 per trip to come back with zero wedgefish/hammerheads? If *yes*, circle the amount and keep going until the respondent says *no*.

Interview

1. To confirm, under this scenario, if your vessel was paid [IDR xxx - minimum amount above] per trip to land zero wedgefish/hammerheads [delete as appropriate] what changes, if any, would you expect in the number of wedgefish/hammerheads landed after each trip?

Decrease

No Change

Increase

2. *[If changes]* Why would this change in landings happen?

3. *If this policy was implemented, and you reduced your landings to zero, how would your vessel ensure that you returned from each trip with zero landings?*

Category	Details of how				
Avoid catching them all together					
Try to minimize catching them					
Throw them in the water after capture					
(If this option, clarify if usually alive or dead)	Almost always dead	Usually dead	Sometimes alive, sometimes dead	Usually alive	Almost always alive
Other					

4. Why would you do this as opposed to ... *[other options]*? Why wouldn't you do the other options?

5. *[If changes]* Would you expect any changes in your household income under this scenario? What changes?

Large negative changes in HHI	Moderate negative changes in HHI	No changes	Moderate positive changes in HHI	Large positive changes in HHI
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6. *How fair is this scenario for you and your household?*

Very unfair	Unfair	Neutral	Fair	Very fair
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Please explain:

Scenario 3. Stick (negative incentive) [ASK THIS QUESTION SECOND FOR 50% OF PARTICIPANTS, AND THIRD FOR 50% OF PARTICIPANTS]

In the next year, I want you to imagine that the government introduces a law to fully protect wedgefish/hammerheads. There is someone at the port monitoring every day. There is an administrative warning and a fine per vessel for each vessel landing a wedgefish/hammerheads [delete as appropriate]. Every vessel will be checked after each trip. Everything else will remain the same, you will still continue shark fishing for most species as usual, prices will remain the same as they are now, and there will be no further regulations, except there will be a fine to the vessel if you return from trip and land a wedgefish/hammerheads. The fine will be taken out of the profit from your trip.

Contingent valuation

Firstly, I want you to think about how large would this fine would need to be, in order for your vessel to reduce your landings of wedgefish/hammerheads to zero.

Payment value (IDR) per vessel per trip
100,000
125,000
250,000
500,000
750,000
1,000,000
1,500,000
2,000,000
3,000,000
5,000,000
8,000,000
12,000,000
15,000,000
20,000,000

how large would this fine would need to be, in order for your vessel to reduce your landings of wedgefish/hammerheads [delete as appropriate] to zero. Let's start with... [for 50% ask a, for 50% ask b]

- a) **20,000,000. If your vessel was fined IDR 20,000,000 per trip** if you came back with zero wedgefish/hammerheads would you **definitely come back with zero wedgefish/hammerheads?** If *yes*, circle the amount and keep going until the respondent says "*no*".
- b) **If your vessel was fined IDR 100,000 per trip if you came back with wedgefish/hammerheads, would you come back with zero, or fish as usual?** If *fish as usual*, put a line through the amount and keep going until the respondent says "*we would come back with zero if we were fined this amount*".

Interview

1. *To confirm, under this scenario, if your vessel was fined [IDR XXX – minimum amount above] per trip, what changes, if any, would you expect in the number of wedgefish/hammerheads your vessel lands after each trip?*

Decrease

No Change

Increase

2. *[If changes] Why would this change in landings happen?*

3. *If this policy was implemented and you reduced your landings to zero, how would this change in landings happen?*

Category	Details of how				
Avoid catching them all together					
Try to minimize catching them					
Throw them in the water after capture					
(If this option, clarify if usually alive or dead)	Almost always dead	Usually dead	Sometimes alive, sometimes dead	Usually alive	Almost always alive
Other					

4. Why would you do this as opposed to ... *[other options]*? Why wouldn't you do the other options?

5. *[If changes] Would you expect any changes in your household income under this scenario? What changes?*

Large negative
changes in HHI

Moderate negative
changes in HHI

No changes

Moderate positive
changes in HHI

Large positive changes
in HHI

6. *How fair is this scenario for you and your household?*

Very unfair

Unfair

Neutral

Fair

Very fair

Please explain:

Scenario 4. Non-monetary carrot (indirect incentive)

Now I want you to imagine that instead of a direct payment to the vessels, there is a community education fund.

Every month there is a lottery, and the winners receive a **school fees scholarship** from the education fund, **worth up IDR 3 million**, for one school-age child in their family, for the **following year**. Only people from vessels that have landed zero wedgefish/hammerheads [delete as appropriate] that month are entered in to the lottery.

1. *Under this scenario, what changes, if any, would you expect in the number of wedgefish/hammerheads your vessel lands after each trip?*

Decrease

No Change

Increase

2. *[If changes] Why would this change in landings happen?*

3. *If this policy was implemented, and you reduced your landings to zero, how would this change in landings happen?*

Category	Details of how				
Avoid catching them all together					
Try to minimize catching them					
Throw them in the water after capture					
(If this option, clarify if usually alive or dead)	Almost always dead	Usually dead	Sometimes alive, sometimes dead	Usually alive	Almost always alive
Other					

4. *Why would you do this as opposed to ... [other options]? Why wouldn't you do the other options?*

5. *[If changes]* Would you expect any changes in your household income under this scenario? What changes?

Large negative changes in HHI	Moderate negative changes in HHI	No changes	Moderate positive changes in HHI	Large positive changes in HHI
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6. How fair is this scenario for you and your household?

Very unfair	Unfair	Neutral	Fair	Very fair
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Please explain:

Scenario 4b. non-monetary carrot with peer pressure

1. Finally, I want you to imagine that the monthly community lottery only takes places if **all vessels** land zero wedgefish/hammerheads that month. That is, if all vessels land zero wedgefish/hammerheads *[delete as appropriate]*, then all fishers are entered in to the lottery, but if just one vessel lands a wedgefish/hammerheads, noone receives any benefits...
- Would this change your answers to the previous question?
 - Would you be...

Much more likely to land zero	More likely to land zero	No changes	Less likely to land zero	Much less likely to land zero
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c. Please explain...

THANK YOU FOR PARTICIPATING IN OUR SURVEY [give a small gift]

Do you have any comments or questions for us?

Lhok Rigaih

Pre-survey

1. Which village do you live in?			
2. At which Lhok do you land your catch?			
3. Are you originally from Aceh Jaya?	Yes	No	If no, where are you from? When did you come to Aceh Jaya?
4. Gear			
5. Target species			
6. Age			
7. Years as fisher			
8. Level of education			
9. Do you own the boat you use for fishing?	Yes	No	
10. Name of vessel you use for fishing			
11. Fish alone or with others	Alone	With others... No. people in total _____	
INSTRUCTIONS: If they fish alone, you can skip to Q17			
12. If with others... Position in vessel	Crew	Captain	
13. If with others... Time working on that vessel			
14. If with others... what share of the profits do you get per trip?			
15. If with others... In general, how 'democratic' is the decision-making on your vessel? For example, is there joint decision-making between everyone in terms of when you go and where you fish, or does the skipper make all decisions?	Not democratic (skipper makes all decisions)	Somewhat democratic (everyone has a say, skipper makes final decision)	Very democratic (everyone has equal say)
16. If with others... Can you personally influence what happens during a fishing trip?	I have some influence (I contribute equally along with others)	I have a large influence (I contribute more than others)	I have ultimate influence (I make all of the decisions)
17. Last week... a. How many times did you go fishing? b. How many days did you spend at sea per trip? c. Do you usually fish this often? i. If last week was more/less than usual, how often do you usually fish per week? ii. Is this the same throughout the year?			
18. Approx. monthly household income a. Is this all from fishing, or do you have other sources of income? If so, please explain.			
19. When you go on a fishing trip, are there any measures you can take which influence which species you will catch?	None - the species composition of our	A little - we can change the species	A lot - we can choose which

	catch is totally random	composition of our catch a little by...)	species we want to catch by...)
a. If fishers can influence, please explain how			
20. WF - Do you know this fish? [show wedgefish picture and confirm]	Yes	No	
a. When was the last time you caught a wedgefish?			
b. How many did you catch?			
c. Was it/were they small or large?			
d. Is this typical... How often do you usually catch small wedgefish?	- Every trip - At least once per week - At least once per month - A few times per year - Less than once per year - Never		
e. Do you usually catch [answer from b] small WF per trip or sometimes more or less?			
f. Do you know how much they (small WF) are sold for?			
g. How about a large wedgefish... when was the last time you caught one?			
h. Is this typical... How often do you usually catch large wedgefish?	- Every trip - At least once per week - At least once per month - A few times per year - Less than once per year - Never		
i. If you do catch a large wedgefish, how many do you catch in a trip?			
j. Do you know how much they (large WF) are sold for?			
k. Based on this, is it accurate to say that your vessel normally makes IDR X per trip from WF catches, if you catch them?			
21. HH - Do you know this fish? [show hammerhead picture and confirm]	Yes	No	
a. When was the last time you caught HH on a fishing trip?			
b. How many did you catch?			
c. Is this typical?... How often do you usually catch hammerheads?	- Every trip - At least once per week - At least once per month - A few times per year - Less than once per year - Never		
d. Do you usually catch [answer from b], or sometimes more or less?	-		
a. Do you know how much the HHs are sold for?			

b. Based on this, is it accurate to say that your vessel normally makes IDR X per trip from HH catches?	
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INSTRUCTIONS: Start with WF for 50% of fishers, and HH for 50% of fishers.

We will present fishers with four scenarios, where Scenario 1 (BAU) should always be presented first, and Scenarios 2 and 3 (voluntary and positive incentive) should always be presented sequentially, but Scenario 4 (negative incentive) should be asked second for 50% of respondents:

1. BAU (Business as usual)
2. Voluntary reduction no compensation
3. Voluntary reduction with compensation/positive incentive (+ CV)
4. Negative incentive (+ CV)

Within the scenarios 3 and 4 there is a contingent valuation question with two options (A & B). For 50% of the respondents, option A should be presented and for 50% of the respondents option B should be presented.

Scenario 1. BAU [ALWAYS ASK THIS QUESTION FIRST]

In the next year, I want you to imagine that everything remains the same as it is now. Fishing continues as it currently is, there are no further regulations, fish prices remain the same...

6. Under this scenario, what changes, if any, would you expect in the number of wedgefish/ hammerheads [delete as appropriate] you land each month?

Decrease

No Change

Increase

- a. [If changes]... By how much would you expect it to change?

- b. [If changes]... Please explain: why would this change happen?

7. Under this scenario, would you expect any changes in how people in your community perceive or participate in ocean stewardship?

Large negative changes in
ocean stewardship

Moderate negative
changes in ocean
stewardship

No changes

Moderate positive
changes in ocean
stewardship

Large positive changes in
ocean stewardship

- a. If so, why would these changes happen?

8. Under this scenario, would you expect any changes in your relationships with other members of your community?

Large deterioration in relationships with other community members	Moderate deterioration in relationships with other community members	No changes	Moderate improvements in relationships with other community members	Large improvements in relationships with other community members
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a. If so, why would these changes happen?

9. Under this scenario, would you expect any changes in your household income?

Large negative changes in HHI	Moderate negative changes in HHI	No changes	Moderate positive changes in HHI	Large positive changes in HHI
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a. If so, why would these changes happen?

10. How fair are these management policies for you and your household?

Very unfair	Unfair	Neutral	Fair	Very fair
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a. Please explain your answer:

Scenario 2. Voluntary release **[Always ask S2 before S3]**

Now I want you to imagine that in the next year that most things remain the same as they are now – there are no new regulations, fishing continues as it currently is, and prices for fish remain the same. However, there is a **voluntary wedgefish/hammerheads** **[delete as appropriate]** **guardians'** scheme established by the government and *Panglima Laot*, in which fishers can **voluntarily pledge** to participate in wedgefish/hammerheads **[delete as appropriate]** monitoring and conservation activities.

Each member of the wedgefish/hammerheads **[delete as appropriate]** guardians commits to **landing zero wedgefish**. They receive training in wedgefish/hammerheads **[delete as appropriate]** monitoring techniques, including how to release wedgefish/hammerheads **[delete as appropriate]** from their nets, and those that perform well will receive additional training in how to tag them and provide monitoring data to support research efforts. All members also receive a t-shirt, training certificates, recognition in local newsletters, and the opportunity to join regular social events with the regency government and *Panglima Laot* each month.

Anyone from the guardians who lands a wedgefish/hammerheads **[delete as appropriate]** will be removed from the group for a year. If members accidentally catch wedgefish/hammerheads **[delete as appropriate]** while fishing it's ok, but they have to be released - they cannot be brought to the harbour and sold.

1. *If this 'wedgefish/ hammerheads* **[delete as appropriate]** *guardians' group was real, would you be interested in joining it?*

Yes

No

I don't know

- a. Why/Why not? Please explain your answer:

2. *Under this scenario, what changes, if any, would you expect in the number of wedgefish/ hammerheads* **[delete as appropriate]** *you land each month?*

Decrease

No Change

Increase

- a. **[If changes]**... By how much would you expect it to change?

- b. Please explain your answer, why would you change/not change?

- c. **[if decrease to zero]**, how would you ensure that you had zero landings after each trip? **[NOTE: only ask this question if not discussed in previous question – no need to repeat]**

Category	Please explain				
i. I would avoid catching wedgefish/hammerheads [delete as appropriate] while at sea					
ii. I would release them back in to the water after catching them					
- If release... Are they usually alive or dead?	Almost always dead	Usually dead	Sometimes alive, sometimes dead	Usually alive	Almost always alive
iii. Other method for ensuring zero landings (please explain)					

3. Under this scenario, would you expect any changes in how people in your community perceive or participate in ocean stewardship?

Large negative changes in ocean stewardship Moderate negative changes in ocean stewardship No changes Moderate positive changes in ocean stewardship Large positive changes in ocean stewardship

a. If so, why would these changes happen?

4. Under this scenario, would you expect any changes in your relationships with other members of your community?

Large deterioration in relationships with other community members Moderate deterioration in relationships with other community members No changes Moderate improvements in relationships with other community members Large improvements in relationships with other community members

a. If so, why would these changes happen?

5. Under this scenario, would you expect any changes in your household income?

Large negative changes in HHI Moderate negative changes in HHI No changes Moderate positive changes in HHI Large positive changes in HHI

- a. If so, why would these changes happen?

6. *How fair is this management policy for you and your household?*

Very unfair

Unfair

Neutral

Fair

Very fair

- a. Please explain your answer:

Scenario 3. Voluntary release with compensation [always ask after S2]

Now I want you to imagine that a **voluntary wedgefish/hammerheads** [delete as appropriate] **guardians' scheme** is established, as described above, but this time with a **small monetary stipend, which acts as compensation for reducing wedgefish/hammerheads catches to zero and providing monitoring data**. As before, fishers/vessels who join the scheme pledge to participate in wedgefish/hammerheads monitoring and conservation, which means they commit landing **zero wedgefish, and to releasing any they accidentally catch while fishing**. All members receive training in wedgefish/hammerheads monitoring techniques, how to release wedgefish/hammerheads from their nets, and a small camera to monitor their on-board fishing activities. After each trip they are expected provide monitoring data on any wedgefish/hammerheads interactions (e.g. if they caught any, how many, where they caught them, and if they were alive or dead when released) and submit video footage of the release as proof.

In return, they receive a t-shirt and certificate, and a **payment after each trip as compensation for any released catches and for providing monitoring data**. The payment is given **after every trip to each participating vessel** which returns with zero wedgefish/hammerheads and provides data and videos. You and your captain/crew can choose to divide the payment between you **as you wish**.

Anyone in the scheme who is found landing/killing a wedgefish/hammerheads will be removed from the group, and won't be able to receive any benefits for a whole year.

Contingent valuation

Firstly, I want you to think about **how large this per vessel per trip payment would need to be** in order for you to join the scheme and reduce your wedgefish/hammerheads [delete as appropriate] landings to zero...

Please bear in mind the amounts we discussed earlier in terms of your usual income from wedgefish/hammerheads [remind them of the amount they stated in the pre-survey], and note that the scheme would be **competitive** amongst other villages in the local area, with limited funding. That means the amount stated would need to be reasonable for compensation, and within budget for the funding to be provided. If the amount is too high the scheme would not be established.

Payment value (IDR) per vessel per trip
20,000
40,000
60,000
80,000

[Option A – 50% of respondents] So - if your vessel was paid IDR 20,000 per trip, would you join the wedgefish/hammerheads guardians, and reduce your wedgefish/hammerheads landings to zero, or fish as usual?

[If 'fish as usual', put a line through the amount and keep going down the ladder until the respondent says "*I would join the wedgefish/hammerheads guardians and land zero wedgefish*"]

[Option B – 50% of respondents] So - if your vessel was paid IDR 2 million per month, would you definitely join the wedgefish/hammerheads guardians, and reduce your wedgefish/hammerheads landings to zero, or fish as usual?

100,000
150,000
200,000
250,000
300,000
350,000
400,000
500,000
600,000
800,000
1,000,000
2,000,000

Interview

1. To confirm, if this 'wedgetfish/hammerheads guardians' group was real, would you be interested in joining it?

Yes

No

I don't know

- a. Why/why not? Please explain your answer:

2. To confirm, under this scenario, if your vessel was paid IDR [stopping point above] per trip, what changes would you expect in the number of wedgetfish/hammerheads you land each month?

Decrease

No Change

Increase

- a. [If decrease] to confirm, you would decrease to zero? Y / N

- b. Please explain your answer - why would you make this change / why would you not change?

- c. [if decrease to zero], how would you ensure that you had zero landings after each trip? **[NOTE: only ask this question if not discussed in previous question – no need to repeat]**

Category	Please explain				
i. I would avoid catching wedgefish/hammerheads while at sea					
ii. I would release them back in to the water after catching them					
- If release... Are they usually alive or dead?	Almost always dead	Usually dead	Sometimes alive, sometimes dead	Usually alive	Almost always alive
iii. Other method for ensuring zero landings (please specify)					

3. Under this scenario, would you expect any changes in how people in your community perceive or participate in ocean stewardship?

Large negative changes in ocean stewardship Moderate negative changes in ocean stewardship No changes Moderate positive changes in ocean stewardship Large positive changes in ocean stewardship

a. If so, why would these changes happen?

4. Under this scenario, would you expect any changes in your relationships with other members of your community?

Large deterioration in relationships with other community members Moderate deterioration in relationships with other community members No changes Moderate improvements in relationships with other community members Large improvements in relationships with other community members

a. If so, why would these changes happen?

5. Under this scenario, would you expect any changes in your household income?

Large negative changes in HHI Moderate negative changes in HHI No changes Moderate positive changes in HHI Large positive changes in HHI

a. If so, why would these changes happen?

6. *How fair is this management policy for you and your household?*

Very unfair Unfair Neutral Fair Very fair

a. Please explain your answer:

Scenario 3b. monetary compensation with peer pressure

Now, I want you to imagine that the payments only occur if **all fishers** in your community land zero wedgefish/hammerheads each day. That is, if all vessels land zero wedgefish, then the whole community receives a payment, but if just one vessel lands a wedgefish, no one receives any compensation...

a. Would this change your answers to the previous question? b. Would you be...

Much more likely to More likely to land No changes Less likely to land Much less likely to
land zero wedgefish zero wedgefish zero wedgefish land zero wedgefish

b. Please explain...

Scenario 4. Negative incentive [ask after BAU for 50% of participants]

Finally, I want you to imagine that in the next year there is a new rule established which protects wedgefish/hammerheads [delete as appropriate]. Everything else remains the same in terms of fish prices and other regulations, but no one is allowed to land or trade wedgefish. Someone is monitoring the harbour every day, and there is a fine for anyone who is found landing a wedgefish/hammerheads at the harbour. The fine will be levied at the vessel level, and would come out of your total trip profits for that trip.

Contingent valuation

Payment value (IDR) per vessel per trip
20,000
40,000
60,000
80,000
100,000
150,000
200,000
250,000
300,000
350,000
400,000
500,000
600,000
800,000
1,000,000
2,000,000

“Firstly, I want you to think about **how large this fine would need to be** in order for you to definitely reduce your landings of wedgefish/hammerheads to zero.”

[Option A – 50% of respondents] “If your vessel was fined IDR 20,000 for landing wedgefish, would you stop catching and landing wedgefish, or fish as usual?”

[If fish as usual, put a line through the amount and keep going until the respondent says “*I would stop catching and landing wedgefish*”]

[Option B – 50% of respondents] “If your vessel was fined IDR 2 million for landing wedgefish, would you stop catching and landing wedgefish, or fish as usual?”

[If ‘yes I would stop’, put a circle around the amount and keep going up the ladder until the respondent says “*I would fish as usual*”]

NOTE IF THE RESPONDENT REFUSES TO ANSWER/GIVE A VALUE

Interview

1. To confirm, under this scenario, if your vessel was fined IDR [stopping point above] for landing wedgefish, what changes would you expect in the number of wedgefish/hammerheads you land each month?

Decrease

No Change

Increase

- a. If decrease, to confirm, you would decrease to zero? Y/N

b. Please explain your answer: why would you change/not change?

c. *[if decrease to zero], how would you ensure that you had zero landings after each trip? [NOTE: only ask this question if not discussed in previous question – no need to repeat]*

Category	Please explain				
iv. I would avoid catching wedgefish/hammerheads while at sea					
v. I would release them back in to the water after catching them					
- Are they usually alive or dead?	Almost always dead	Usually dead	Sometimes alive, sometimes dead	Usually alive	Almost always alive
vi. Other method for ensuring zero landings (please specify)					

2. Under this scenario, would you expect any changes in how people in your community perceive or participate in ocean stewardship?

Large negative changes in ocean stewardship	Moderate negative changes in ocean stewardship	No changes	Moderate positive changes in ocean stewardship	Large positive changes in ocean stewardship
---	--	------------	--	---

a. If so, why would these changes happen?

3. Under this scenario, would you expect any changes in your relationships with other members of your community?

Large deterioration in relationships with other community members	Moderate deterioration in relationships with other community members	No changes	Moderate improvements in relationships with other community members	Large improvements in relationships with other community members
---	--	------------	---	--

a. If so, why would these changes happen?

4. *Under this scenario, would you expect any changes in your household income?*

Large negative changes in HHI	Moderate negative changes in HHI	No changes	Moderate positive changes in HHI	Large positive changes in HHI
----------------------------------	-------------------------------------	------------	-------------------------------------	----------------------------------

b. If so, why would these changes happen?

5. *How fair is this management policy for you and your household?*

Very unfair	Unfair	Neutral	Fair	Very fair
-------------	--------	---------	------	-----------

a. Please explain your answer:

THANK YOU FOR PARTICIPATING IN OUR SURVEY [give a small gift]

Do you have any comments or questions for us?

S8.2 Modelling process for behaviour change model

Mixed-effects logistic regressions constructed using behaviour change as a binary response variable, interviewee as a random effect, and all other variables as fixed effects. Models compared based on delta AIC, as per Table S8.2

Table S8.2. A comparison of the different models considered for explaining willingness to change behaviour

	Models considered								
	M1	M2	M2	M2	M3	M4	M5	M6	Null model
Variables included	Scenario	Scenario	Scenario	Scenario	Scenario	Scenario	Scenario	Scenario	1
	Taxa	Taxa	Taxa	Taxa	Taxa	Taxa	Taxa	Taxa	
	Site	Site	Site	Site	Site	Site			
	Income	Income	Income	Income	Income	Income	Income		
	Last catch	Last catch	Last catch	Last catch	Last catch	Last catch	Last catch	Last catch	
	Age	Age		Age	Age	Age	Age	Age	
			Experience						
	High school	High school	High school	High school	High school				
	Influence score		Influence score	Influence score	Influence score				
		Position							
	Order	Order	Order	Order					
	Interviewee	Interviewee	Interviewee	Interviewee	Interviewee	Interviewee	Interviewee	Interviewee	
AIC	352.3	352.9	354.2	352.4	350.4	348.3	347.6	346.2	788

Full model outputs from Model M6

Generalized linear mixed model fit by maximum likelihood (Laplace Approximation) ['glmerMod']

Family: binomial (logit)

Formula: decrease_catch ~ scenario + species + catch_last + age + (1 | interview)

Data: PES_fish_data_3

AIC	BIC	logLik	deviance	df.resid
346.2	376.6	-166.1	332.2	560

Scaled residuals:

Min	1Q	Median	3Q	Max
-7.7025	-0.1694	0.0436	0.1419	13.2065

Random effects:

Groups	Name	Variance	Std.Dev.
interview	(Intercept)	2.038	1.428

Number of obs: 567, groups: interview, 142

Fixed effects:

	Estimate	Std. Error	z value	Pr(> z)
(Intercept)	-2.44963	0.94307	-2.597	0.00939 **
scenario carrot	9.24450	1.12768	8.198	2.45e-16 ***
scenario stick	4.56381	0.63665	7.168	7.59e-13 ***
species WF	-0.10653	0.40912	-0.260	0.79457
catch_last	-0.09182	0.03587	-2.560	0.01047 *
age	-0.04569	0.02295	-1.991	0.04645 *

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Correlation of Fixed Effects:

	(Intr)	scnr cr	scnr st	spcs WF	ctch_l
scenario crrt	-0.223				
scenario stck	-0.373	0.796			
species WF	-0.003	-0.024	-0.014		
catch_last	-0.053	-0.386	-0.253	0.256	
age	-0.783	-0.280	-0.224	-0.117	0.102

S8.3. Models of WTP & WTA

Methods

We constructed separate models explaining WTP and WTA, acknowledging systematic differences in biases and motivations associated with negative vs. positive incentives (e.g. loss aversion) (Hanemann, 1991; Kahneman & Tversky, 1979)). For these models we only included data where fishers stated they would change their behaviour (i.e. excluding all zeros), and explored linear models of the candidate explanatory variables (Table 2) against $\log(\text{WTP}/\text{WTA})$ as a continuous response variable. Models were constructed in RStudio using the `lm` function in the R Stats package and the `lmer` function in the `lme4` package (Bates et al., 2015; RStudio Team, 2020).

Model 1: WTA, both sites & taxa together

Number of observations = 184, Number of interviewees = 140

Linear model (fixed effects only)

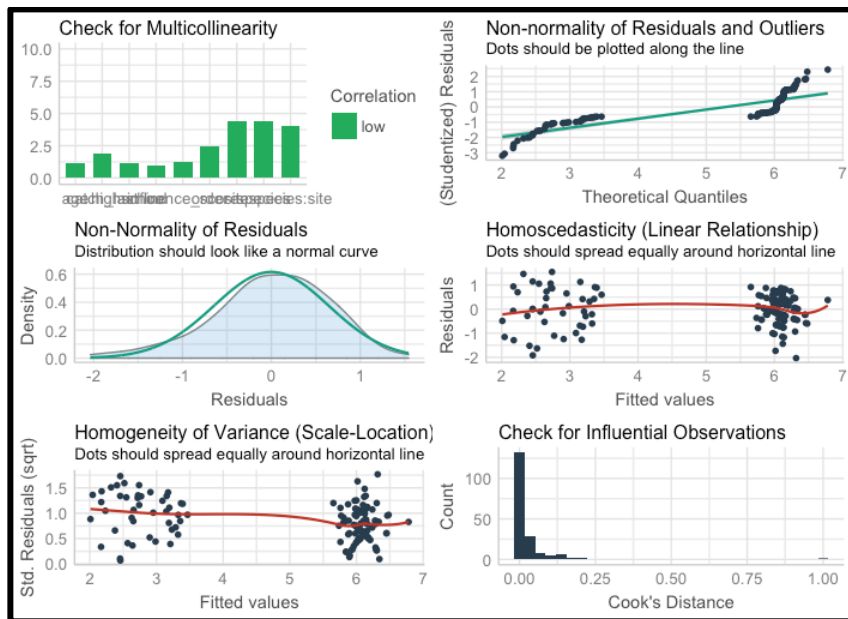
Model as `lm` first: test interaction between site and taxa, and check linear model assumptions

```
Call:
lm(formula = logwta ~ species * site + inc + catch_last + age +
    highschool + influence_score + order, data = s1c)

Residuals:
    Min       1Q   Median       3Q      Max
-2.03501 -0.37402  0.03764  0.43402  1.54391

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept)    2.094e+00  6.143e-01   3.409 0.000809 ***
speciesWF      -7.544e-01  2.395e-01  -3.150 0.001919 **
siteTL         3.060e+00  2.400e-01  12.753 < 2e-16 ***
inc            8.147e-02  4.555e-02   1.789 0.075404 .
catch_last     -4.635e-03  1.154e-02  -0.402 0.688383
age            7.521e-05  5.800e-03   0.013 0.989670
highschool     -2.448e-01  1.716e-01  -1.427 0.155473
influence_score  9.782e-02  6.828e-02   1.433 0.153730
order          2.401e-01  1.291e-01   1.860 0.064556 .
speciesWF:siteTL 1.061e+00  2.980e-01   3.561 0.000477 ***
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.6634 on 174 degrees of freedom
Multiple R-squared:  0.8308,    Adjusted R-squared:  0.822
F-statistic: 94.92 on 9 and 174 DF,  p-value: < 2.2e-16
```



```

Random effects:
Groups      Name          Variance Std.Dev.
interview (Intercept) 0.09575  0.3094
Residual      0.34292  0.5856
Number of obs: 184, groups:  interview, 140

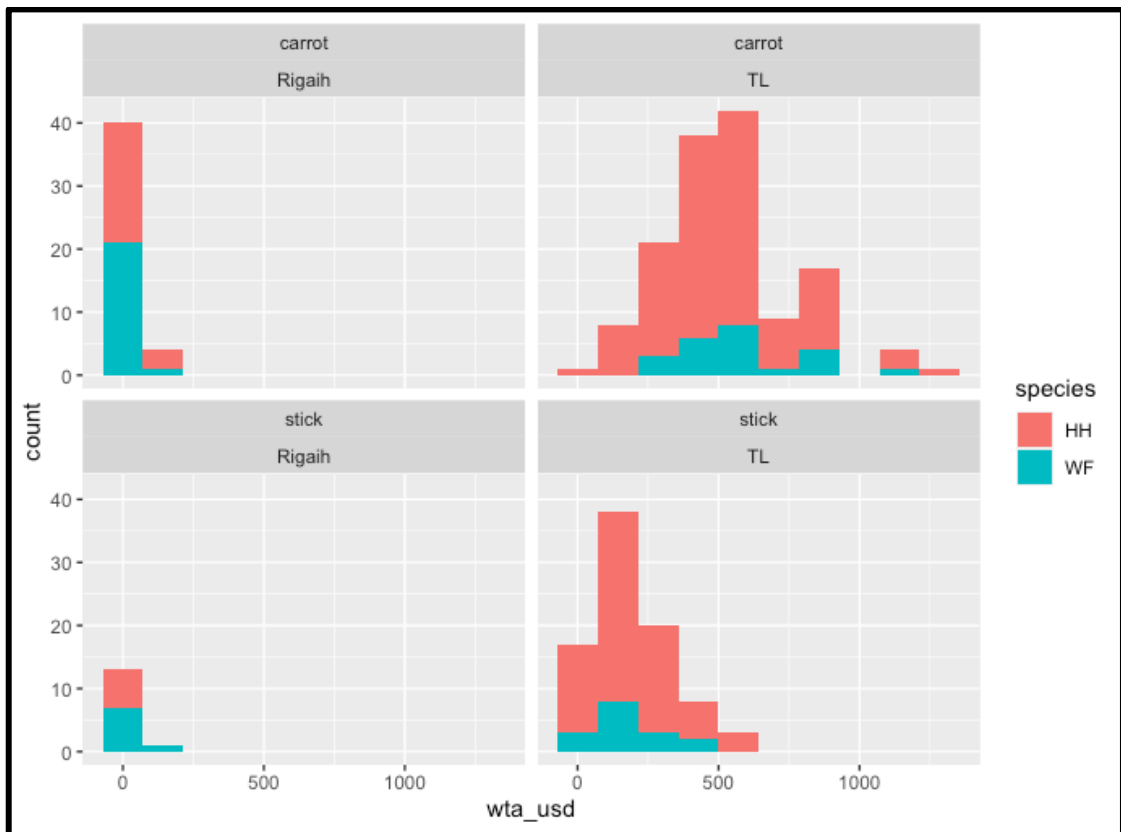
Fixed effects:
              Estimate Std. Error      df t value Pr(>|t|)
(Intercept)    2.125e+00  6.318e-01  1.526e+02   3.363 0.000973 ***
speciesWF      -7.382e-01  2.195e-01  9.593e+01  -3.363 0.001109 **
siteTL         3.068e+00  2.419e-01  1.707e+02  12.686 < 2e-16 ***
inc            7.933e-02  4.745e-02  1.382e+02   1.672 0.096855 .
catch_last    -3.213e-03  1.143e-02  1.686e+02  -0.281 0.778993
age            -2.839e-04  5.976e-03  1.495e+02  -0.048 0.962178
highschool     -2.249e-01  1.805e-01  1.279e+02  -1.246 0.215120
influence_score 8.797e-02  7.156e-02  1.319e+02   1.229 0.221132
order          2.338e-01  1.334e-01  1.463e+02   1.753 0.081689 .
speciesWF:siteTL 1.043e+00  2.709e-01  8.682e+01   3.850 0.000225 ***
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Correlation of Fixed Effects:
              (Intr) spcsWF siteTL inc    ctch_l age    hghsch infln_ order
speciesWF    -0.283
siteTL       -0.752  0.634
inc          -0.168  0.003  0.009
catch_last   -0.288  0.594  0.572  0.005
age          -0.496  0.100  0.264  0.019  0.169
highschool   -0.265 -0.005  0.205 -0.018 -0.008  0.183
influnc_scr  -0.258 -0.024  0.223  0.000 -0.040 -0.141 -0.026
order        -0.807 -0.018  0.442 -0.016 -0.031  0.079  0.183  0.246
spcsWF:stTL  0.010 -0.791 -0.430  0.018 -0.448 -0.074  0.069 -0.021  0.295

```

Conclusions:

- Model is significant
- Site, taxa and the interaction between site and taxa are significant
- However, linear model assumptions are not well met – significant differences WTA in Rigaih and Tanjung Luar are too different – not comparable as single population



Histogram of WTA (top) and WTP (bottom) for Lhok Rigiah (left) and Tanjung Luar (right) and hammerheads (pink) and wedgefish (blue)

Model 2: WTA, separate sites, taxa together

2a Tanjung Luar

Number of observations = 140, Number of interviewees = 118

Model with all variables

```
Call:
lm(formula = logwta ~ species + catch_last + age + highschool +
    influence_score + order, data = s1cT)

Residuals:
    Min       1Q   Median       3Q      Max
-1.87393 -0.27288  0.08863  0.34750  1.09440

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept)   5.619355   0.470221  11.950  <2e-16 ***
speciesWF      0.280339   0.164620   1.703   0.0909 .
catch_last    -0.013452   0.028525  -0.472   0.6380
age            -0.002018   0.005310  -0.380   0.7046
highschool     0.077691   0.195166   0.398   0.6912
influence_score 0.037949   0.070604   0.537   0.5918
order          0.178645   0.137911   1.295   0.1974
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.5495 on 133 degrees of freedom
Multiple R-squared:  0.03115, Adjusted R-squared: -0.01255
F-statistic: 0.7128 on 6 and 133 DF, p-value: 0.6399
```

Simpler model with fewer key variables

```
Call:
lm(formula = logwta ~ species + catch_last + age, data = s1cT)

Residuals:
    Min       1Q   Median       3Q      Max
-1.7999 -0.2564  0.1079  0.3195  1.1486

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept)   6.188793   0.206568  29.960  <2e-16 ***
speciesWF      0.155160   0.128589   1.207   0.230
catch_last    -0.016994   0.027999  -0.607   0.545
age            -0.002872   0.005091  -0.564   0.574
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.5474 on 136 degrees of freedom
Multiple R-squared:  0.01705, Adjusted R-squared: -0.004629
F-statistic: 0.7865 on 3 and 136 DF, p-value: 0.5034
```

- Both models are not significant and R-squared is low
- Wedgefish correlates positively with WTA, and is close to significance threshold

Number of observations = 44, Number of interviewees = 22

Model with all variables

```
Call:
lm(formula = logwta ~ species + catch_last + age + highschool +
    influence_score + order, data = s1cR)

Residuals:
    Min       1Q   Median       3Q      Max
-1.7214 -0.6812 -0.1049  0.7409  1.6014

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept)   2.289246   1.624899   1.409   0.1672
speciesWF     -0.660631   0.364686  -1.812   0.0782 .
catch_last     0.003585   0.019161   0.187   0.8526
age            0.015384   0.021031   0.731   0.4691
highschool    -0.675971   0.425694  -1.588   0.1208
influence_score 0.181434   0.182482   0.994   0.3266
order          0.017694   0.351492   0.050   0.9601
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.9681 on 37 degrees of freedom
Multiple R-squared:  0.2455,    Adjusted R-squared:  0.1232
F-statistic: 2.007 on 6 and 37 DF,  p-value: 0.08946
```

Simpler model with fewer key variables

```
Call:
lm(formula = logwta ~ species + catch_last + age, data = s1cR)

Residuals:
    Min       1Q   Median       3Q      Max
-1.89303 -0.57079  0.09786  0.69614  1.91721

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept)   2.064308   1.044093   1.977   0.0549 .
speciesWF     -0.645492   0.370968  -1.740   0.0895 .
catch_last     0.004912   0.019261   0.255   0.8000
age            0.023541   0.021125   1.114   0.2718
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.9913 on 40 degrees of freedom
Multiple R-squared:  0.1449,    Adjusted R-squared:  0.08077
F-statistic: 2.259 on 3 and 40 DF,  p-value: 0.09627
```


- Both models are not significant and R-squared is low
- Wedgefish correlates negatively with WTA, and is close to significance threshold

Model 3: WTP, separate sites, taxa together

3a Tanjung Luar

- Number of observations = 86, Number of interviewees = 71
- Simple lm is not significant
- WF correlates negatively with WTP

```
Call:
lm(formula = logwta ~ species + catch_last + age, data = s1sT)

Residuals:
    Min       1Q   Median       3Q      Max
-1.8160 -0.5344  0.1026  0.5262  1.5456

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept)  4.695135   0.373555  12.569  <2e-16 ***
speciesWF    -0.195077   0.229835  -0.849   0.398
catch_last   -0.049735   0.055465  -0.897   0.373
age           0.006112   0.009739   0.628   0.532
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.7905 on 82 degrees of freedom
Multiple R-squared:  0.01768,    Adjusted R-squared:  -0.01826
F-statistic: 0.4919 on 3 and 82 DF,  p-value: 0.6889
```

3b Rigaib

- Number of observations = 10, Number of interviewees = 7
- Simple lm is not significant
- WF correlates negatively with WTP

```
Call:
lm(formula = logwta ~ species + catch_last + age, data = s1sR)

Residuals:
    Min       1Q   Median       3Q      Max
-1.39140 -0.86834  0.03812  0.62669  1.50692

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept) -0.03777    2.85948  -0.013   0.990
speciesWF     0.71812    1.31361   0.547   0.604
catch_last   -0.02292    0.22678  -0.101   0.923
age           0.04305    0.03970   1.085   0.320

Residual standard error: 1.255 on 6 degrees of freedom
Multiple R-squared:  0.3187,    Adjusted R-squared:  -0.02193
F-statistic: 0.9356 on 3 and 6 DF,  p-value: 0.4796
```

Model 4: WTA, separate sites, separate taxa

4a Tanjung Luar

Variables	Data included in model							
	WTA Wedgefish (N obs = 23)		WTP Wedgefish (N obs = 16)		WTA Hammerheads (N obs = 117)		WTP Hammerheads (N obs = 70)	
	All variables	Simpler model	All variables	Simpler model	All variables	Simpler model	All variables	Simpler model
Income	x		x		x		x	
Last catch	x	x	x	x	x	x	x	x
Age	x	x	x	x	x		x	x
Fisher experience	x		x		x	x	x	
Education	x		x	x	x		x	
Influence score	x	x	x	x	x	x	x	x
Vessel position	x		x	x	x		x	
Model Significance	0.8287 (NS)	0.8964 (NS)	0.2094 (NS)	0.1039 (NS)	0.3563 (NS)	0.6592 (NS)	0.7855 (NS)	0.462 (NS)
R squared	-0.1945	-0.1228	0.2706	0.3304	0.00714	-0.01217	-0.04675	-0.00575

4b Rigaih

Variables	Data included in model					
	WTA Wedgefish (N obs = 22)		WTP Wedgefish (N obs = 6)	WTA Hammerheads (N obs = 22)		WTP Hammerheads (N obs = 4)
	All variables	Simpler model	No possible to build model – too few data points	All variables	Simpler model	No possible to build model – too few data points
Income	x	x		x	x	
Last catch	x			x	x	
Age	x	x		x	x	
Fisher experience	x			x		
Education	x	x		x		
Influence score	x	x		x		
Vessel position	x	x		x	x	

Model	0.008483	0.003215		0.161	0.366	
Significance				(NS)	(NS)	
R squared	0.5349	0.5301		0.2391	0.02823	

Call:

```
lm(formula = logwta ~ age + highschool + inc + fisher_experience +
    position + +influence_score + catch_last, data = wfcR)
```

Residuals:

Min	1Q	Median	3Q	Max
-0.92575	-0.53987	-0.00396	0.48216	1.33176

Coefficients:

	Estimate	Std. Error	t value	Pr(> t)
(Intercept)	-2.69888	1.73377	-1.557	0.14187
age	0.05746	0.03147	1.826	0.08928 .
highschool	-1.10589	0.40557	-2.727	0.01637 *
inc	0.66310	0.19298	3.436	0.00401 **
fisher_experience	-0.04273	0.02913	-1.467	0.16451
positioncrew	1.40107	0.45390	3.087	0.00804 **
influence_score	1.11812	0.27611	4.049	0.00119 **
catch_last	0.08985	0.12284	0.731	0.47657

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.7564 on 14 degrees of freedom

Multiple R-squared: 0.6899, Adjusted R-squared: 0.5349

F-statistic: 4.451 on 7 and 14 DF, p-value: 0.008483

AIC = 58

Call:

```
lm(formula = logwta ~ income + age + highschool + influence_score +
    position, data = wfcR)
```

Residuals:

Min	1Q	Median	3Q	Max
-1.0420	-0.4315	-0.0893	0.4741	1.6255

Coefficients:

	Estimate	Std. Error	t value	Pr(> t)
(Intercept)	-2.169e+00	1.432e+00	-1.515	0.149391
income	6.783e-07	1.843e-07	3.680	0.002024 **
age	2.833e-02	2.339e-02	1.211	0.243404
highschool	-1.024e+00	4.027e-01	-2.544	0.021668 *
influence_score	1.138e+00	2.639e-01	4.314	0.000535 ***
positioncrew	1.373e+00	4.259e-01	3.224	0.005299 **

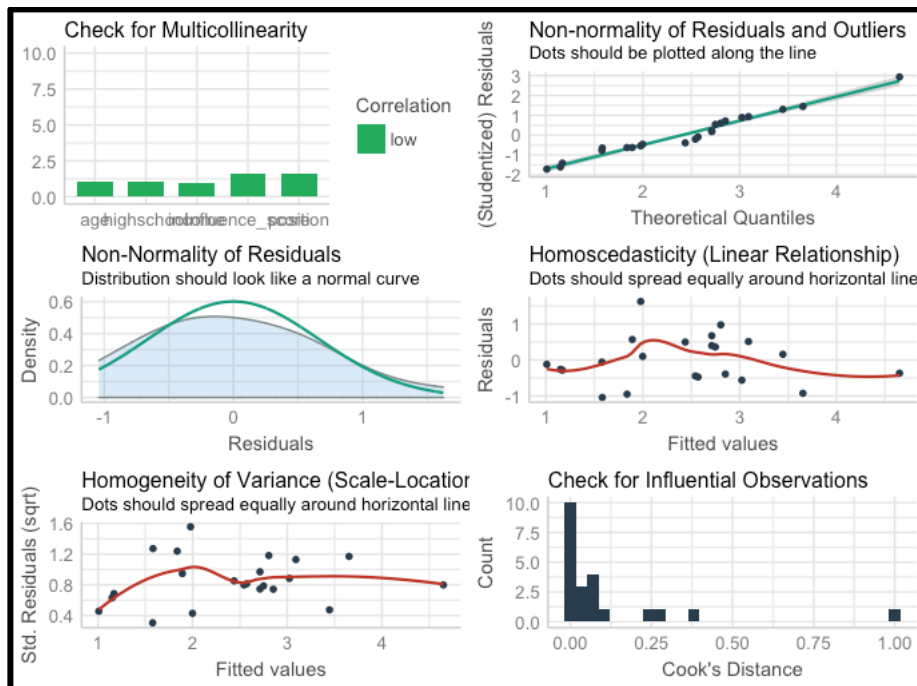
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.7603 on 16 degrees of freedom

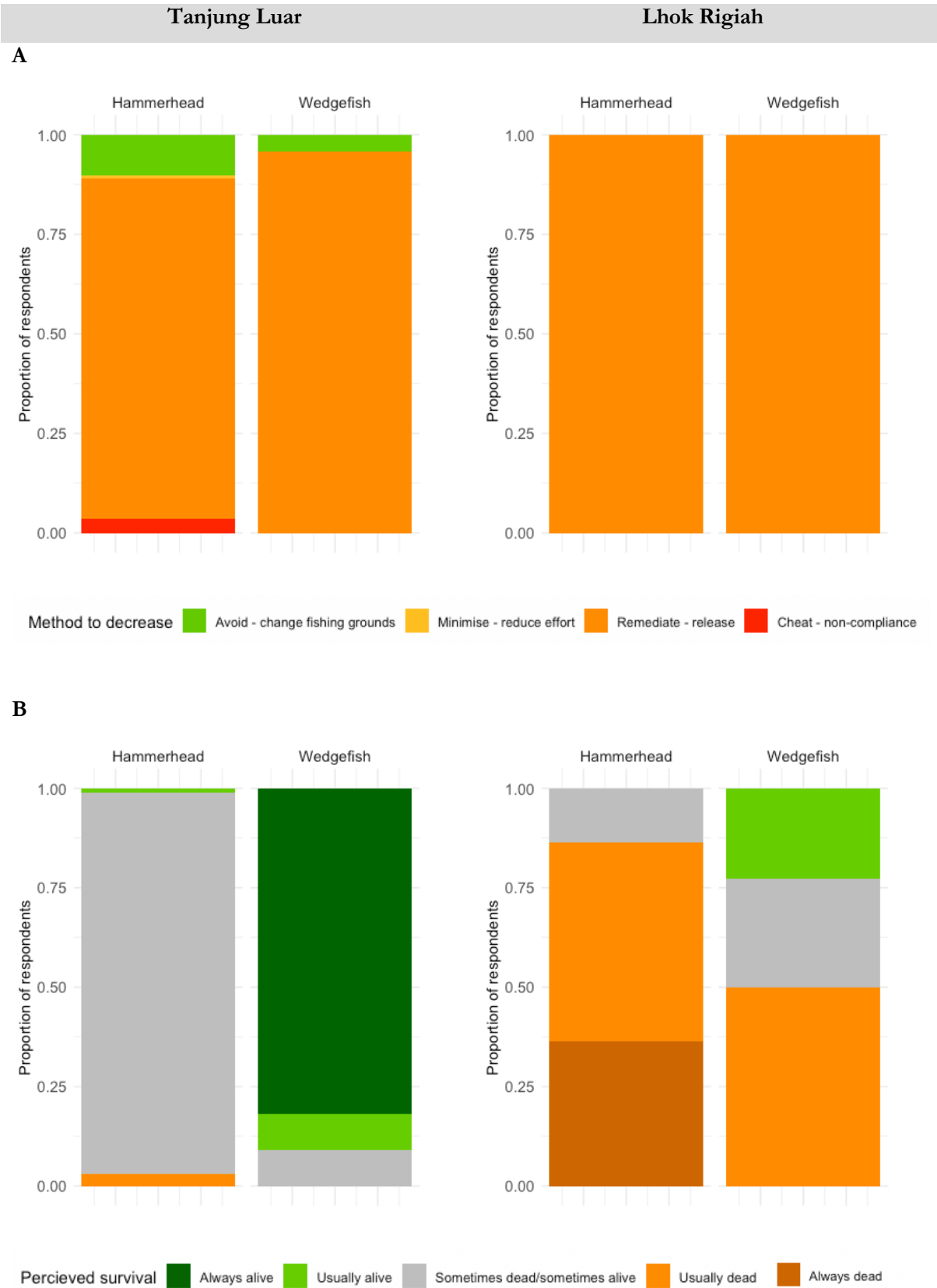
Multiple R-squared: 0.642, Adjusted R-squared: 0.5301

F-statistic: 5.738 on 5 and 16 DF, p-value: 0.003215

AIC = 57



S8.4. Summary of reported techniques for reducing landings, and perceived survivability



Appendix 6: Supplementary Material for Chapter 9

S9.1 Online Marine Tourism Survey

Page 1: Introduction and consent

Thank you for taking our survey! This survey is being led by an independent PhD researcher at Oxford University. It will take **10-15 minutes to complete**, and provide valuable information on marine tourism. At the end of the survey you also have the option to enter a **prize draw**, in which you will have a **1 in 100 chance of winning \$20!**

Before you proceed, please read through the following points and confirm that you understand and agree to take part:

- Participation is voluntary.
- We won't collect or share any personally identifiable information, including IP addresses.
- Jisc hosts the survey platform and states that it is GDPR compliant. Please see their privacy policy for more details.
- The information that you share will be saved in a secure database, and results will be analysed and presented as part of a written thesis for a doctoral degree. Some generalisable results may also be published internationally in academic papers, at conferences and/or in online blogs. The findings may also be used to inform marine tourism policy and practices by governments, NGOs and the private sector.
- You are free to leave the survey at any time.
- Your responses are submitted at the end of the survey. Once submitted, you will not be able to withdraw since we have no way of identifying your response.
- The study has been reviewed and approved by the University of Oxford Central University Research Ethics Committee (CUREC).
- For questions, comments, or complaints, please email the lead researcher, Hollie Booth (hollie.booth@zoo.ox.ac.uk), and we will aim to get back to you within 10 working days.
- If you remain unhappy or wish to make a formal complaint, please email the Medical Sciences Interdivisional Research Ethics Committee (ethics@medsci.ox.ac.uk).

Please confirm you have read and understood the above information, and give your consent to proceed *Required*

I confirm and consent

Page 2: Prolific ID

Please enter your Prolific ID here: *Required*

Page 3: Travel preferences

In the past three years, which of the following regions have you visited on holiday? Please tick all that apply. If you are unsure about which countries fall in to which regions, please refer to [this map](#). *Required*

- Sub-Saharan Africa
- Northern Africa and Western Asia
- Central and Southern Asia
- Eastern and South-Eastern Asia
- Latin America and the Caribbean
- Australia and New Zealand
- Oceania (excluding Australasia)
- Europe and Northern America
- None - I haven't been on holiday in the past three years

What are your main motivations for going on holiday? Please select up to three options *Optional*

- Adventure
- Culture
- Learning and discovery
- Nature, wildlife and the outdoors
- Rest and relaxation
- Sports
- Better weather
- Spend time with family and friends
- Visit family and friends abroad
- To mark a special occasion
- Other

If you selected Other, please specify:

Which of the following options best describes your typical holiday budget? *Required*

- Budget/backpacker
- Mid-range
- High end/luxury

Assuming international travel safely re-opens, which of the following regions are you most likely to visit on holiday within the next three years? Please tick all that apply. If you are unsure about which countries fall in to which regions, please refer to [this map](#). *Optional*

- Sub-Saharan Africa
- Northern Africa and Western Asia
- Central and Southern Asia
- Eastern and South-Eastern Asia
- Latin America and the Caribbean
- Australia and New Zealand
- Oceania (excluding Australasia)
- Europe and Northern America

Page 4: Your last trip

Where did you go on your last holiday? *Required*

How long was your trip? Please provide your answer in **number of days**. *Required*

Please enter a whole number (integer).

Roughly how much money did you spend for **the whole trip**? Don't worry if you cannot remember precisely, please provide an approximate answer. N.B. Complete this box using **numbers only**, and **select the currency below**. *Required*

Please enter a number.

Currency *Required*

How many people did this expenditure cover? *Required*

Page 5: Marine activities

Do you participate in any of the following marine activities or water sports while on holiday? *Required*

	Yes	No
Boat trips		
Scuba-diving		
Snorkelling		
Surfing		
Kayaking		
Other (please specify below)		

Please list any other marine activities or watersports you participate in that are not listed above

If you scuba-dive, do you have a PADI AWARE or SSI Marine Ecology certification?

Yes

No

Page 6: Perceptions of marine animals

In this section, we are interested in understanding your perceptions of the following marine animals...

Reef sharks:



I have seen reef sharks in the wild

Yes
No
I don't know

Regarding **reef sharks**, to what extent do you agree/disagree with the following statements... *Required*

	Strongly agree	Agree	Neutral	Disagree	Strongly disagree	I don't know
I like this animal						
Wild populations of this animal need to be protected						

If you would like to, please provide more details to explain your answers/describe your perceptions of reef sharks:

In this section, we are interested in understanding your perceptions of the following marine animals...

Manta rays:



I have seen manta rays in the wild

Yes
No
I don't know

Regarding **manta rays**, to what extent do you agree/disagree with the following statements... *Required*

	Strongly agree	Agree	Neutral	Disagree	Strongly disagree	I don't know
I like this animal						
Wild populations of this animal need to be protected						

If you would like to, please provide more details to explain your answers/describe your perceptions of manta rays:

In this section, we are interested in understanding your perceptions of the following marine animals...

Wedgefish:



I have seen wedgefish in the wild

Yes
No
I don't know

Regarding **wedgefish**, to what extent do you agree/disagree with the following statements... *Required*

	Strongly agree	Agree	Neutral	Disagree	Strongly disagree	I don't know
I like this animal						
Wild populations of this animal need to be protected						

If you would like to, please provide more details to explain your answers/describe your perceptions of wedgefish:

Background on threats to sharks [informational intervention seen by 50% of participants]

Before you proceed, we would like to give you some background information on shark conservation and marine tourism. Please *either watch* this 90-second video, *or read* the information provided in the text below.



<https://youtu.be/jqsZNYBap5s>

- Many shark and ray species are **threatened with extinction**, primarily due to **fishing**.
- In some tropical countries, such as Indonesia and India, **people in coastal communities catch endangered sharks and rays**, like hammerhead sharks, wedgefish and manta rays.
- They use the sharks for **food and income**, and other economic opportunities are usually limited.
- The average monthly income for a shark fisher in Lombok, Indonesia is just \$300. For an average family of 7, this is less than \$1.50 per person per day.
- These countries are also **popular tourist destinations** for scuba-divers, snorkelers and other marine activities.
- The marine tourism industry is **worth millions of dollars**, and people often pay thousands of dollars for trips to see sharks and rays.
- Shark and ray fishers **cannot usually benefit from this**, because they don't have the skills and resources to run tourism businesses.
- As such, **marine tourism does not typically contribute to protecting shark and ray populations in the wild**.
- Addressing this requires that **local communities are given alternative economic opportunities to shark fishing, or a reason to value live sharks and healthy shark populations**.
- Payments from marine tourists could play a role in **creating incentives for fishers to reduce shark catches**.



Please confirm that you have either watched the video or read the text above, and that you understand the information *Required*

I confirm and understand

Page 9: Marine Conservation Payment Scheme for Tourists

In the following section, we want you to imagine that you have taken a beach holiday to a tropical destination (e.g. Indonesia, Malaysia), and you would like to participate in a **marine tourism activity**. This activity could be boat-based (e.g. a glass-bottom boat) or in the water (e.g. snorkeling or scuba-diving).



Near to your holiday destination there is a small fishing village, where **fishers often catch endangered sharks and rays** for their food and income.

A mandatory **marine conservation fee** has been introduced for all marine tourism activities in the area, which are in addition to the usual price of the activity. The money is used to **support shark and ray conservation** in the local fishing village, as follows:

- Marine tourism operators pool the fees, and use it as **direct compensation** for local fishers to **stop catching** endangered species
- **Compensation payments** are **conditional**, such that they are **only paid to fishers who stop or reduce catches**, and catches are monitored by a local NGO **every day**
- **The more money that is available, the more sharks that can be saved.** For example, it requires around \$8 to save one juvenile wedgefish, or up to \$140 to save a large adult wedgefish.

We are interested in your willingness to pay to such a fee.



- The fees would be paid **per person per day of marine activity**. For example, if you do a marine activity on one day of your holiday you will pay the conservation fee once, but if you do marine activities on five different days you will pay the conservation fee five times.

Page 10: Marine Conservation Payment Scheme for Tourists

Important note: Before you complete the question, we want to ask you to help us with a problem we have in studies like this. Because this is a hypothetical scenario and people don't really have to pay the amount, they often don't pay a lot of attention to the actual cost shown, and respond differently from how they would behave in real life, if they did have to really pay. If people don't pay attention to the actual costs, our analysis will be wrong, and we won't get a true measure of the potential value of tourist payments for protecting sharks. Please help us measure your willingness to pay correctly by considering the prices as if you were really about to pay them. *Required*

I confirm I have read the above information, and will consider the fees as if I was really about to pay them

In the scenario described on the previous page, what is the maximum amount - in USD - you would be prepared to pay per person per day for a **marine conservation fee**? Please scroll through the following amounts and select the **maximum** amount you **would be prepared pay**. *Required*

\$0 - I would not be prepared to pay

\$1 per person per day

\$2 per person per day

\$5 per person per day

\$10 per person per day

\$15 per person per day

\$20 per person per day

\$25 per person per day

\$35 per person per day

\$45 per person per day

\$55 per person per day

\$65 per person per day

\$75 per person per day
 \$100 per person per day
 \$125 per person per day
 \$150 per person per day
 \$200 per person per day
 \$300 per person per day

[If \$0] Please let us know why you would be unwilling to pay a marine conservation fee

I can't afford it
 I am not sufficiently concerned about shark conservation to pay for it
 I don't agree with this program
 I think this program is unrealistic
 I don't believe that the money would be spent effectively
 I would not go on this trip
 Other

If you selected Other, please specify:

Page 11: Perception of the scenario

We are interested in your perceptions of the scenario described on the previous page. To what extent do you agree/disagree with the following statements... *Required*

	Strongly agree	Agree	Neutral	Disagree	Strongly disagree	I don't know
This scenario is realistic and credible - I think it could happen in practice						
I am likely to participate in a marine activity as described in the scenario						

Please explain your answers. Why is this scenario realistic/unrealistic?

Do you actively engage in any of the following causes or activities... *Required*

	Yes	No
I am a member of an environmental or animal welfare organisation		
I donate money towards an animal or environmental charity		
I volunteer for an animal or environmental charity		
I regularly visit aquariums		
I regularly go on fishing trips		

To what extent do you agree/disagree with the following statements... *Required*

	Strongly agree	Agree	Neutral	Disagree	Strongly disagree
It is my responsibility to do something to tackle environmental issues					
It is the tourism industry's responsibility to do something to tackle environmental issues					
I am more likely to purchase					

goods and services from a tourism company if they are playing a role in addressing environmental issues					
---	--	--	--	--	--

Page 13: Background information

What is your nationality? *Required*

Are you currently resident in your country of nationality?

Yes

No

If no, where do you currently live?

Age

Please enter a whole number (integer).

Gender *Required*

Female

Male

Non-binary

Prefer not to say

What is your approximate total household income in USD (all sources, gross per annum)? N.B. As per current exchange rates: 1 GBP ~ USD 1.4 1 EUR ~ USD 1.2 *Required*

Less than \$20,000

\$20,000 to \$34,999

\$35,000 to \$49,999

\$50,000 to \$74,999

\$75,000 to \$99,999

\$100,000 to \$149,999

\$150,000 to \$199,999

\$200,000 and over

Prefer not to say

What is the highest degree or level of school you have completed? (If you're currently enrolled in school, please indicate the highest degree you have *received*.)

Did not complete high school

Completed high school (e.g. GED, GCSE)

Vocational qualification (e.g. NVQs, CTE)

Bachelor's degree (e.g. BA, BSc)

Master's degree (e.g. MA, MSc)

Professional degree (e.g. MD, DDS, DVM)

Doctorate (e.g. PhD, EdD)

How would you best describe your political leanings?

To the left

Centre

To the right

Prefer not to say

Which of the following best describes your dietary preferences?

Omnivore (I eat everything)

Flexitarian (mostly plant-based, while allowing meat and other animal products in moderation)

Pescatarian (fish, no meat)

Vegetarian (diary, no meat and fish)

Vegan (entirely plant-based)

Other

If you selected Other, please specify:

Page 14: Prize draw

Thank you for completing this survey! As advertised, you have a **1 in 100 chance of winning \$20 for participating**. You can choose to donate a proportion of your prize towards [Thresher Shark Project Indonesia](#), which will go directly towards reducing capture of endangered sharks in a small-scale fishery in Eastern Indonesia. If you would like to donate a portion of your prize, please indicate below **how much you would like to donate**, if anything. Please don't feel obliged to donate if you prefer not to, or if you have other causes you like to donate to instead.

\$0.00
\$2.00
\$4.00
\$6.00
\$8.00
\$10.00
\$12.00
\$14.00
\$16.00
\$18.00
\$20.00
Other

If you selected Other, please specify the amount you would like to donate:

Please provide your email address so that we can contact you to arrange payment of your prize, should you be successful:

Please indicate if you are also happy for us to use this email address to contact you about this research in the future

Yes, I am happy to be contacted about this research
No, please just contact me if I have won the prize draw

Page 15: Final comments

Do you have any additional comments or information you would like to share, which has not been covered elsewhere in this survey?

Page 16: End of survey

Thank you so much for taking part in this survey! Please **CLICK THIS LINK** to take you back to the Prolific app and confirm you have completed the study.

Your contributions will help to inform environmentally-friendly marine tourism strategies. If you want to know more about this research, and what you can do to support shark conservation, you can visit

the research profile of the lead researcher, Hollie Booth; follow Hollie on Twitter (@holleiboothie); or send her an email (hollie.booth@zoo.ox.ac.uk).

Key for selection options

7 - Where did you go on your last holiday?

Afghanistan
Albania
Algeria
Andorra
Angola
Antigua and Barbuda
Argentina
Armenia
Australia
Austria
Azerbaijan
Bahamas
Bahrain
Bangladesh
Barbados
Belarus
Belgium
Belize
Benin
Bhutan
Bolivia
Bosnia and Herzegovina
Botswana
Brazil
Brunei
Bulgaria
Burkina Faso
Burundi
Cabo Verde
Cambodia
Cameroon

Canada
Central African Republic
Chad
Chile
China
Colombia
Comoros
Congo
Costa Rica
Côte d'Ivoire
Croatia
Cuba
Cyprus
Czech Republic (Czechia)
Denmark
Djibouti
Dominica
Dominican Republic
DR Congo
Ecuador
Egypt
El Salvador
Equatorial Guinea
Eritrea
Estonia
Eswatini
Ethiopia
Fiji
Finland
France
Gabon
Gambia
Georgia
Germany
Ghana
Greece
Grenada

Guatemala
Guinea
Guinea-Bissau
Guyana
Haiti
Holy See
Honduras
Hong Kong
Hungary
Iceland
India
Indonesia
Iran
Iraq
Ireland
Israel
Italy
Jamaica
Japan
Jordan
Kazakhstan
Kenya
Kiribati
Kuwait
Kyrgyzstan
Laos
Latvia
Lebanon
Lesotho
Liberia
Libya
Liechtenstein
Lithuania
Luxembourg
Madagascar
Malawi
Malaysia

Maldives
Mali
Malta
Marshall Islands
Mauritania
Mauritius
Mexico
Micronesia
Moldova
Monaco
Mongolia
Montenegro
Morocco
Mozambique
Myanmar
Namibia
Nauru
Nepal
Netherlands
New Zealand
Nicaragua
Niger
Nigeria
North Korea
North Macedonia
Norway
Oman
Pakistan
Palau
Panama
Papua New Guinea
Paraguay
Peru
Philippines
Poland
Portugal
Qatar

Romania
Russia
Rwanda
Saint Kitts & Nevis
Saint Lucia
Samoa
San Marino
Sao Tome & Principe
Saudi Arabia
Senegal
Serbia
Seychelles
Sierra Leone
Singapore
Slovakia
Slovenia
Solomon Islands
Somalia
South Africa
South Korea
South Sudan
Spain
Sri Lanka
St. Vincent & Grenadines
State of Palestine
Sudan
Suriname
Sweden
Switzerland
Syria
Taiwan
Tajikistan
Tanzania
Thailand
Timor-Leste
Togo
Tonga

Trinidad and Tobago
Tunisia
Turkey
Turkmenistan
Tuvalu
Uganda
Ukraine
United Arab Emirates
United Kingdom
United States
Uruguay
Uzbekistan
Vanuatu
Venezuela
Vietnam
Yemen
Zambia
Zimbabwe

20 - What is your nationality?

Afghanistan
Albania
Algeria
Andorra
Angola
Antigua and Barbuda
Argentina
Armenia
Australia
Austria
Azerbaijan
Bahamas
Bahrain
Bangladesh
Barbados
Belarus
Belgium
Belize

Benin
Bhutan
Bolivia
Bosnia and Herzegovina
Botswana
Brazil
Brunei
Bulgaria
Burkina Faso
Burundi
Cabo Verde
Cambodia
Cameroon
Canada
Central African Republic
Chad
Chile
China
Colombia
Comoros
Congo
Costa Rica
Côte d'Ivoire
Croatia
Cuba
Cyprus
Czech Republic (Czechia)
Denmark
Djibouti
Dominica
Dominican Republic
DR Congo
Ecuador
Egypt
El Salvador
Equatorial Guinea
Eritrea

Estonia
Eswatini
Ethiopia
Fiji
Finland
France
Gabon
Gambia
Georgia
Germany
Ghana
Greece
Grenada
Guatemala
Guinea
Guinea-Bissau
Guyana
Haiti
Holy See
Honduras
Hong Kong
Hungary
Iceland
India
Indonesia
Iran
Iraq
Ireland
Israel
Italy
Jamaica
Japan
Jordan
Kazakhstan
Kenya
Kiribati
Kuwait

Kyrgyzstan
Laos
Latvia
Lebanon
Lesotho
Liberia
Libya
Liechtenstein
Lithuania
Luxembourg
Madagascar
Malawi
Malaysia
Maldives
Mali
Malta
Marshall Islands
Mauritania
Mauritius
Mexico
Micronesia
Moldova
Monaco
Mongolia
Montenegro
Morocco
Mozambique
Myanmar
Namibia
Nauru
Nepal
Netherlands
New Zealand
Nicaragua
Niger
Nigeria
North Korea

North Macedonia
Norway
Oman
Pakistan
Palau
Panama
Papua New Guinea
Paraguay
Peru
Philippines
Poland
Portugal
Qatar
Romania
Russia
Rwanda
Saint Kitts & Nevis
Saint Lucia
Samoa
San Marino
Sao Tome & Principe
Saudi Arabia
Senegal
Serbia
Seychelles
Sierra Leone
Singapore
Slovakia
Slovenia
Solomon Islands
Somalia
South Africa
South Korea
South Sudan
Spain
Sri Lanka
St. Vincent & Grenadines

State of Palestine
Sudan
Suriname
Sweden
Switzerland
Syria
Taiwan
Tajikistan
Tanzania
Thailand
Timor-Leste
Togo
Tonga
Trinidad and Tobago
Tunisia
Turkey
Turkmenistan
Tuvalu
Uganda
Ukraine
United Arab Emirates
United Kingdom
United States
Uruguay
Uzbekistan
Vanuatu
Venezuela
Vietnam
Yemen
Zambia
Zimbabwe

21.a - If no, where do you currently live?

Afghanistan
Albania
Algeria
Andorra
Angola

Antigua and Barbuda
Argentina
Armenia
Australia
Austria
Azerbaijan
Bahamas
Bahrain
Bangladesh
Barbados
Belarus
Belgium
Belize
Benin
Bhutan
Bolivia
Bosnia and Herzegovina
Botswana
Brazil
Brunei
Bulgaria
Burkina Faso
Burundi
Cabo Verde
Cambodia
Cameroon
Canada
Central African Republic
Chad
Chile
China
Colombia
Comoros
Congo
Costa Rica
Côte d'Ivoire
Croatia

Cuba
Cyprus
Czech Republic (Czechia)
Denmark
Djibouti
Dominica
Dominican Republic
DR Congo
Ecuador
Egypt
El Salvador
Equatorial Guinea
Eritrea
Estonia
Eswatini
Ethiopia
Fiji
Finland
France
Gabon
Gambia
Georgia
Germany
Ghana
Greece
Grenada
Guatemala
Guinea
Guinea-Bissau
Guyana
Haiti
Holy See
Honduras
Hungary
Iceland
India
Indonesia

Iran
Iraq
Ireland
Israel
Italy
Jamaica
Japan
Jordan
Kazakhstan
Kenya
Kiribati
Kuwait
Kyrgyzstan
Laos
Latvia
Lebanon
Lesotho
Liberia
Libya
Liechtenstein
Lithuania
Luxembourg
Madagascar
Malawi
Malaysia
Maldives
Mali
Malta
Marshall Islands
Mauritania
Mauritius
Mexico
Micronesia
Moldova
Monaco
Mongolia
Montenegro

Morocco
Mozambique
Myanmar
Namibia
Nauru
Nepal
Netherlands
New Zealand
Nicaragua
Niger
Nigeria
North Korea
North Macedonia
Norway
Oman
Pakistan
Palau
Panama
Papua New Guinea
Paraguay
Peru
Philippines
Poland
Portugal
Qatar
Romania
Russia
Rwanda
Saint Kitts & Nevis
Saint Lucia
Samoa
San Marino
Sao Tome & Principe
Saudi Arabia
Senegal
Serbia
Seychelles

Sierra Leone
Singapore
Slovakia
Slovenia
Solomon Islands
Somalia
South Africa
South Korea
South Sudan
Spain
Sri Lanka
St. Vincent & Grenadines
State of Palestine
Sudan
Suriname
Sweden
Switzerland
Syria
Tajikistan
Tanzania
Thailand
Timor-Leste
Togo
Tonga
Trinidad and Tobago
Tunisia
Turkey
Turkmenistan
Tuvalu
Uganda
Ukraine
United Arab Emirates
United Kingdom
United States
Uruguay
Uzbekistan
Vanuatu

Venezuela
Vietnam
Yemen
Zambia
Zimbabwe

S9.2 Research instruments for Lombok and Pulau Weh

Tourism operator key informant interviews (in person in Lombok)

Background and oral consent

“Hello, my name is Hollie Booth. I am an independent research student at Oxford University in England and we are looking at the potential role of the tourism sector in marine conservation. Since you’re currently working in marine tourism in Lombok, I’d like to briefly meet with you to discuss my research. It should only take around 20 mins of your time.

Would you like to participate? When is a good time to talk?

In my study, I want to investigate the different ways tourists and tour operators perceive local marine conservation issues, and their potential role in marine conservation. This is my first visit to collect data, and the information you give us during this meeting will be used to design more specific questionnaires, so that when we come back next year we can conduct more detailed interviews.

Before we begin, I should clarify a few things:

- What is said during this meeting is private. The personal information you will share with me will not be passed to any third party. We will not share your answers with anyone else, though will share summarised versions of the overall results.
- We will not ask you anything that could get you into trouble. Participation is voluntary and you may stop the discussion at any time.
- During the interview, I will take some notes and also record the interview, if you give permission. The information that you share with me will be saved in a secure database.
- I will analyse the information and the results will be presented as part of my written thesis for my degree qualification. Some of the results may also be published internationally in academic papers, policy recommendations, at conferences and on online blogs.
- I won’t use your name in any reports or publications, unless you would like me to.
- At the end of the research I will return to present my research findings to tour operators in Lombok, as well as to government and local stakeholders. The findings may be used to inform future conservation efforts by NGOs or government.

This study has been reviewed by, and received ethics clearance through, the University of Oxford Central University Research Ethics Committee. If you have a concern about any aspect of this project, you can speak to me and I will do my best to answer your query. If you remain unhappy or wish to make a formal complaint, I can give you the contact details of the Research Ethics Committee at the University of Oxford.

Is this ok? (Get verbal agreement from participant)

Do you have any questions or concerns so far? (Pause here to give participant enough time to think and comment)

Would you like to continue with the meeting? (Get verbal consent from participant, and tick the interviewee sheet to confirm that verbal consent has been given).

Do you give me permission to audio record this meeting so that I can make sure I don’t miss anything important that you tell us? (Get verbal consent; if anyone objects to being recorded or photographed, do not record the meeting)

Once obtained, consent will be recorded in the interviewee sheet below

Number	Name	Oral information and verbal consent given?

Guiding questions

[most important questions highlighted in yellow]

Name:

Organisation:

How long have you lived/worked in Lombok?

Can you tell me more about your business? What kind of tourism activities do you do?

During these activities, which kinds of marine resources play a role in marine tourism in Lombok?

Are there any that are more important than others?

Are you aware of any threats to these marine resources?

Do these threats impact your business?

I'm interested in your opinion on the impacts of the marine tourism industry on marine resources...

Do you think the marine tourism industry has any **negative impacts** on marine resources?

List them and discuss.

Can you tell me about any activities that the marine tourism industry currently conducts in Lombok which **contribute positively** to conservation? List them and discuss

Do you think the tourism industry should do more?

Why/why not?

What could the tourism industry do?

What could encourage them to do more?

Do you think it could also be beneficial for your business to be engaged in marine conservation?

What kinds of benefits could you perceive?

Are you aware of the Tanjung Luar shark fishery?

Have you ever visited? If so, why?

What are your perceptions of the fishery?

Does this fishery have any role or influence on marine tourism in Lombok? What role?

Does marine tourism have any role or influence on the fishery? What role?

Do people in the tourist industry ever take tourists there?

Do you think something should be done about the fishery? What? How?

Approximately how many tourists per year do you think come to engage in marine tourism in Lombok?

Do you think they would be willing to give financial resources for conservation? What might encourage them to give?

I will be conducting a participatory workshop on this topic with a group of marine tour operators at some point over the next two weeks – would you like to join? Can you specify which times and dates would be most feasible for you to attend?

Can you recommend anyone else I should contact regarding this research?

Do you have any further questions or comments?

Thank you for your time!

Agenda and guiding questions for FGD with tourism sector (in person in Lombok)

Objectives/research questions

1. What are tour operators' understandings of marine/shark conservation issues in the local area?
2. How do they view themselves as actors within the system?
3. What do they see as their potential role in marine/shark conservation in Lombok the future?

Key attendees

1. Dive operators
2. Other hotels and tour operators offering marine-based tourism packages and/or trips to Tanjung Luar market

Agenda

18.30-18.45	Welcome and roundtable introductions
18.45-19.15	Session 1. Introduction - Presentation – background on conservation issues in the local area, and examples of innovative solutions from other locations
10.15-11.00	Session 2. Situation analysis Problem tree of shark and ray fishing - Mapping causes and effects - Understanding the role of the tourism sector as a stakeholder within the system
11.00-11.15	<i>Refreshments break</i>
11.15-12.00	Session 3. Theory of change A vision for the future of marine conservation and tourism in Lombok - What do we want the future to look like? Agree on some priority issues and targets - What can the tourism sector do to achieve this?
12.00-12.45	Session 4. Response Group work and discussion on potential concrete actions from the Lombok tourism sector in contributing to shark conservation
12.45-13.00	Wrap up and close

Guiding questions

Introductions

Name

Organisation

How long have you lived/worked in Lombok

Session 1. Conservation knowledge and motivations

Brief presentation with preliminary results from interviews

List key conservation issues in/around Lombok that came up during interviews

- Here's three stickers each, go and stick a sticker next to the issues that most significantly impact the marine tourism sector. Discuss. Are there any issues missing?
- Here's three stickers each, go and stick a sticker next to the issues you think are most pressing to solve.

Session 2. Shark conservation situation analysis

Map out the current situation using a problem tree

- why shark fishing occurs
- what are the consequences

Which aspects of the tourism sector overlap with this system?

What is the role of the tourism sector as a stakeholder within this system?

Session 3. Theory of change

Think about the future of the marine tourism sector in Lombok. What would you like it to look like?

Everyone draw something together

Do we think the future will look like this? Why/why not?

What can be done to achieve a healthy marine environment (with healthy shark populations) in Lombok?

Agree on a vision

- Map out a results chain
- Indicate areas where the tourism sector can play a role

Session 4. Response

Based on the discussion today, can we map out a timeline of actions that could be taken over the next 3-years to achieve this vision?

Lombok marine tourism survey

Page 1: Introduction and consent

Thank you for visiting our marine tourism survey! It will take around 12-minutes to complete, and could contribute valuable information on improving the role of marine tourism in protecting the marine environment in Indonesia.

This survey is being led by an independent PhD researcher at Oxford University, in collaboration with Bogor Agricultural University, to explore the potential role of the tourism sector in protecting and improving the marine environment in Indonesia.

Please read through the following points before you agree to take part:

- Participation is voluntary.
- We won't collect or share any personally identifiable information, including IP addresses.
- [Jisc](#) hosts the survey platform and [states that it is GDPR compliant](#). Please see their [privacy policy](#) for more details.
- The information that you share will be saved in a secure database, and results will be analysed and presented as part of a written thesis for a doctoral degree. Some generalisable results may also be published internationally in academic papers, at conferences and/or in online blogs. The lead researcher will return to Indonesia to present the research findings to tour operators, government and local stakeholders. The findings may be used to inform future conservation efforts by NGOs or government policy.
- You are free to leave the survey at any time.
- Your responses are submitted at the end of the survey. Once submitted, you will not be able to withdraw since we have no way of identifying your response.
- The study has been reviewed and approved by the [University of Oxford Central University Research Ethics Committee \(CUREC\)](#).

For questions, comments, or complaints, please email the lead researcher, Hollie Booth (hollie.booth@zoo.ox.ac.uk), and we will aim to get back to you within 10 working days.

If you remain unhappy or wish to make a formal complaint, please email the Medical Sciences Interdivisional Research Ethics Committee (ethics@medsci.ox.ac.uk).

1. Please confirm you have read and understood the above information, and give your consent to proceed *Required*

I confirm and consent

Page 2: Participant selection

2. This survey is specifically targeting people who have visited Lombok in Indonesia to participate in marine-based activities. Please confirm that you have visited Lombok for marine tourism (e.g. snorkelling, scuba-diving or surfing). If you have not, you may now exit the survey. *Required*

I confirm I have visited Lombok for marine tourism

Page 3: Participant information

3. Nationality

Please enter a response that only contains letters.

a. Age

18-24

25-34

35-44

45-54

55-64

65 and over

b. Gender

Female

Male

Non-binary

Prefer not to say

4. What were the main reasons you chose to visit Indonesia?

5. How many times have you visited Lombok?

Please enter a whole number (integer).

6. When did you visit Lombok? If you have visited multiple times, please pick the most recent year. Write the year in 20XX format.

Please enter a whole number (integer).

Please make sure the number is between 1970 and 2020.

7. Why did you visit Lombok in particular?

8. How would you categorise your trip budget?

Budget/backpacker

Mid-range

High end/luxury

9. To what extent do you agree/disagree with the following statements

	Strongly agree	Agree	I'm neutral - I neither agree nor disagree	Disagree	Strongly Disagree	I don't know
I am interested in environmental issues						
I am concerned about environmental issues						
It is my responsibility to do something to tackle environmental issues						

Page 4: Marine activities and issues

10. Which types of marine-based activities did you participate in while visiting Lombok? Please tick all that apply.

Scuba-diving

Snorkelling

Surfing

Other

a. If you selected Other, please specify:

b. Which dive shop did you scuba-dive with? Please select

Blue Marlin

Scuba Froggy

Adventure Divers

Two Fish

Seagypsies

Other

i. If you selected Other, please specify:

11. Which place(s) in Lombok did you stay at whilst participating in marine tourism? Please tick all that apply.

Kuta

The Gili islands

Senggigi

Belongas Bay

Ekas

Other

a. If you selected Other, please specify:

b. Please list the specific locations you visited in order to do marine activities (e.g. specific dive sites, snorkelling areas, surf breaks).

12. Do you recall seeing any exciting marine wildlife during your trip to Lombok?

Yes

No

a. Which marine wildlife did you see? Please tick all that apply.

Reef sharks

Whale shark

Manta rays

Other rays

Dolphins

Dugong

Turtles

Other

i. If you selected Other, please specify:

b. Of the marine wildlife you saw during your trip, which were you most excited to see?

Reef sharks

Whale shark

Manta rays

Other rays

Dolphins

Dugong

Turtles

Other

i. If you selected Other, please specify:

13. How did you feel about the state of the marine environment during your trip to Lombok?

Very positive - the marine environment was in pristine condition

Positive - the marine environment was in good condition

Neutral - the marine environment was in neither good or bad condition

Negative - the marine environment was in bad condition

Very negative - the marine environment was in very bad condition

I don't know

a. Given that you didn't feel like the marine environment was in a pristine condition, which types of environmental issues did you notice? Please tick all that apply.

Too much fishing

Too much pollution

The reef is damaged

There are too few fish/marine animals

Other

i. If you selected Other, please specify:

ii. Please describe your experiences of environmental issues in more detail. For example, what did you see and where, how did you feel about it?

iii. Who do you think is responsible for causing these environmental issues? Please tick all that apply

Local people from Lombok

People from other parts of Indonesia

Foreign companies from outside of Indonesia

Tourists

Other

a. If you selected Other, please specify:

Page 5: Background on shark fishing & tourism in Indonesia **seen by 50% of participants**

Before we proceed with the survey, we would like to give you some background information on marine environment issues in Lombok.

Please watch this **1-minute** video, or read the text information below:

Many shark and ray species are threatened with extinction, and Indonesia catches more sharks and rays than anywhere else in the world. Lombok is home to a shark fishing community, where people catch endangered sharks and rays, such as hammerhead sharks, devil rays and thresher sharks. However, people in this village have limited opportunities to make money and feed their families - it is difficult for them to do something other than shark fishing.

The average monthly income for a shark fisher in Lombok is just \$300. For an average family of 7, this works out at less than \$1.50 per person per day. The marine tourism industry in Indonesia is worth millions of dollars, but shark fishers in Lombok cannot usually benefit from this, because they don't have the skills and resources to run tourism businesses. Addressing this requires that local communities in Lombok are given alternative economic opportunities to shark fishing, or a reason to value live sharks and healthy shark populations. Donations from tourists could play a role in creating incentives for fishers to reduce shark fishing in Lombok.

14. Please confirm you have watched the video or read the text above, and that you understand the information *Required*

I confirm and understand

a. Do you have any suggestions for what could be done to address this complex issue?

Page 6: The role of tourism in protecting the marine environment

15. To what degree do you agree or disagree with the following statements:

Please don't select more than 1 answer(s) per row.

	Strongly agree	Agree	Neutral	Disagree	Strongly disagree	I don't know
Marine tourism should play a role in protecting or improving the marine environment in Lombok						
Tourists who visit Lombok should pay a financial contribution towards protecting or improving the marine environment in Lombok						

Tourism companies operating in Lombok, such as hotels or dive shops, should pay a financial contribution towards protecting or improving the marine environment in Lombok						
---	--	--	--	--	--	--

a. Please explain your answers. Why do you agree/disagree that marine tourism should play a role in protecting or improving the marine environment in Lombok?

Page 7: Tourism Payment Scheme

In the following section, we want you to imagine that a marine tourist payment scheme has been established in Kuta, Lombok, such that every dive or surf tourist is requested to **donate towards a marine environment fund**, which would be spent on activities to **protect or improve the marine environment in Lombok**.

We are interested in your **willingness to donate** to such a fund. We want you to imagine that these payments would be made **per person per trip to Lombok**. For example, if you visit Lombok for a 3-day diving trip, you would just make *one* payment to the fund for your *whole holiday to Lombok*, via your dive operator or surf school.

16. Important note: Before you complete the question, we want to ask you to help us with a problem we have in studies like this one. Because people don't really have to pay the donation, they often don't pay a lot of attention to the actual cost shown, and respond differently from how they would behave in real life, if they did have to really pay the donation. If people don't pay attention to the actual costs, our analysis will be wrong, and we won't get a true measure of the potential value of tourist donations for protecting or improving the marine environment. Please help us measure your willingness to pay correctly by considering the donation prices as if you were really about to pay them. Required

I confirm I have read the above information, and will consider the donation prices as if I was really about to pay them

Exchange rates:

We are measuring all values in US\$. At the time of writing this survey, currency conversions were as follows:

US\$ 1.00 = IDR 14,795

US\$ 1.00 = EUR 0.84

US\$ 1.00 = GBP 0.76

US\$ 1.00 = AUD 1.39

US\$ 1.00 = SGD 1.37

Don't see your currency? Please check XE currency converter for the latest exchange rates <https://www.xe.com/currencyconverter/>

17. Would you **definitely pay** US\$ 2.50 **per person per trip** towards a marine environment fund in Lombok?

Yes

No

a. If no, please explain your answer

b. If you would **definitely pay** US\$ 2.50 **per person per trip**, please scroll through the follow amounts until you reach the **maximum** amount you **definitely would pay** towards a marine environment fund in Lombok.

\$2.50 per person per trip

\$5 per person per trip

\$10 per person per trip

\$15 per person per trip

\$20 per person per trip

\$25 per person per trip

\$35 per person per trip

\$45 per person per trip

\$55 per person per trip

\$65 per person per trip

\$75 per person per trip

\$100 per person per trip

\$125 per person per trip

\$150 per person per trip

\$200 per person per trip

\$250 per person per trip

\$500 per person per trip

\$750 per person per trip

\$1000 per person per trip

Page 8: Donation options

19. Tourist donations could be used to support or tackle a range of marine environmental issues in Lombok. To what degree would you be willing/unwilling to donate towards the following issues:

Please don't select more than 1 answer(s) per row.

Please don't select more than 3 answer(s) in any single column.

	Very willing - I would be very happy to support this environmental issue	Willing - I would support this environmental issue	Neutral - I would be neither willing or unwilling to support this issue	Unwilling - I would not support this environmental issue	Very unwilling - I would be strongly against supporting this environmental issue
Reducing shark fishing (e.g. providing alternative income for shark fishers)					
Protecting and restoring coral reefs (e.g. reducing anchor damage, planting corals)					

Preventing plastic pollution (e.g. improving trash collection, regular beach cleans)					
--	--	--	--	--	--

a. Please explain your answers - why would you be more or less willing to pay towards some issues than others?

b. Please mention any other marine environmental issues that you think tourist donations could help to support or alleviate

20. Tourist donations could be given to and managed by a range of different organisations. To what degree would you be willing/unwilling to donate towards the following organisations:

Please don't select more than 1 answer(s) per row.

Please don't select more than 4 answer(s) in any single column.

	Very willing - I would be very happy to support this organisation	Willing - I would support this organisation	Neutral - I would be neither willing or unwilling to support this organisation	Unwilling - I would not support this organisation	Very unwilling - I would be strongly against supporting this organisation
An environmental NGO/charity					
The local government					
The national government					

Directly to a local community					
-------------------------------------	--	--	--	--	--

a. Please explain your answers - why would you be more or less willing to pay towards some organisations than others?

21. Are there any other comments you would like to add regarding this research, and the role of marine tourism in protecting the marine environment in Lombok?

Page 9: End of survey

Thank you so much for taking part in this survey! Your contributions will help to inform better management of the marine environment in Lombok!

S9.3 Summary of tour operator perceptions

Important marine resources

- **Coral reef**
- **Reef fish**
- **Sharks**
 - Reef sharks
 - Hammerhead sharks (Belongas)
 - Whale sharks (occasional)
 - Wobbegong (occasional)
- **Rays**
 - Cownose rays
 - Devil rays
 - Eagle rays
 - Sting rays
 - Mantas (occasional)
 - Guitarfish (occasional)
- **Turtles**
- **Pelagic fish** (i.e. Tuna and barracuda)
- **Marine mammals** (i.e. whales, dolphins, dugongs)
- **Macro** (seahorses, pipefish, scorpionfish, shrimps, mandarin fish, frogfish, nudis etc.)

Participatory ranking of marine resources

	THREAT	YAU	YOUR BUSINESS	HEALTHY ECOSYSTEM
* Squid				
* Coral reef	X	X	X	X
* Turtles		X	X	
* Rays		X		
* Sharks	X	X	X	X
* Rays				
* Pelagic fish	X	X	X	
* Marine mammals	X	X	X	X
* Macro		X	X	X
* Reef fish				X
* Oysters				

Participatory threat identification, ranking and problem tree

Sharks

(Coral Reefs) sharks + bonyfishes

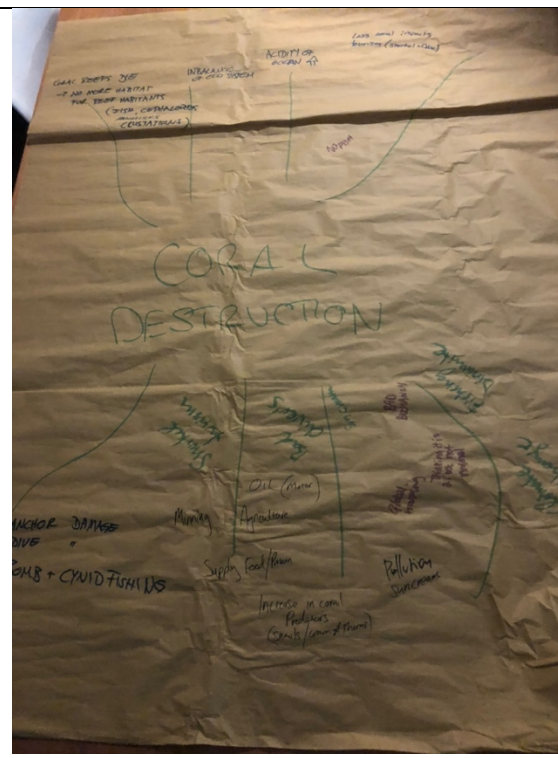
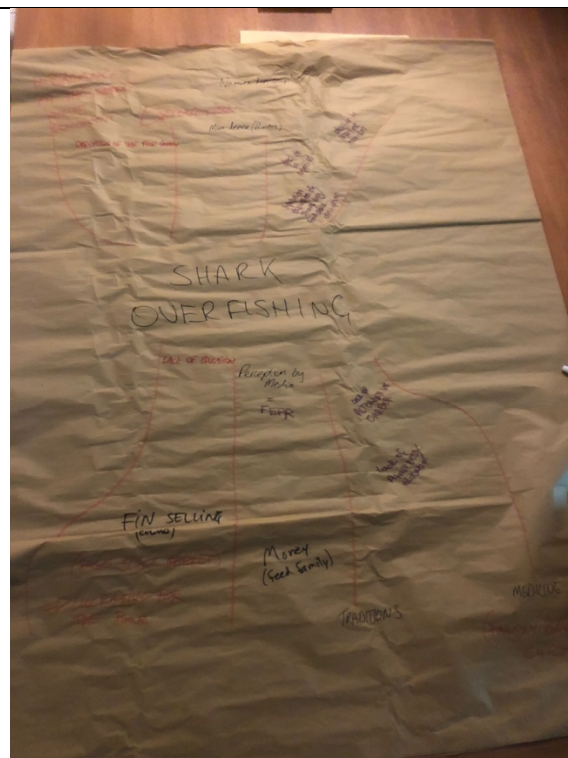
SHARKS

OVERFISHING	4 4 4 4
DESTRUCTIVE FISHING	2 1 2 4
POLLUTION	3 2 3 2 2
HABITAT DEGRADATION	2 3 3 2 4

Coral reefs

CORAL REEFS

OVERFISHING	4 4 4 3 2
ILLEGAL / DESTRUCTIVE FISHING	4 4 4 4 3
POLLUTION	4 4 2 4 4
HABITAT DEGRADATION	4 4 4 4 4



Participatory solution mapping using theory of change

NEXT STEPS /

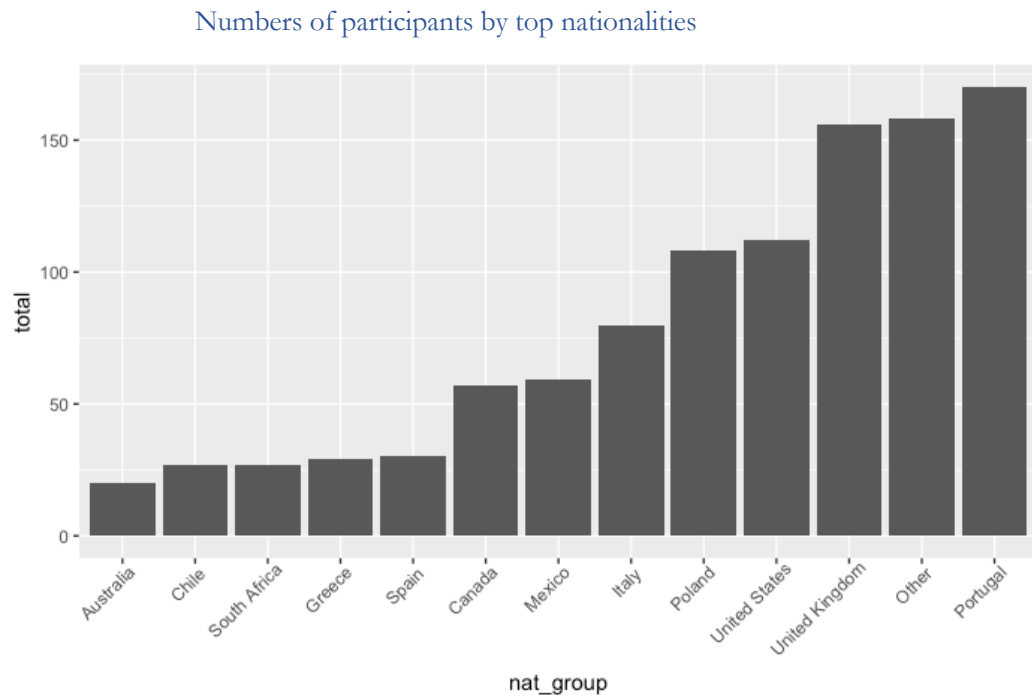
CONCRETE ACTIONS

- * Awareness in dive shops
- * Workshops about issues in surf camps
- * "Standards"
 - "Lombok Green Stamp"
- * Marine protected areas?
 - Enforcement + financing
 - Surveys
- * Tourism donations for marine conservation.

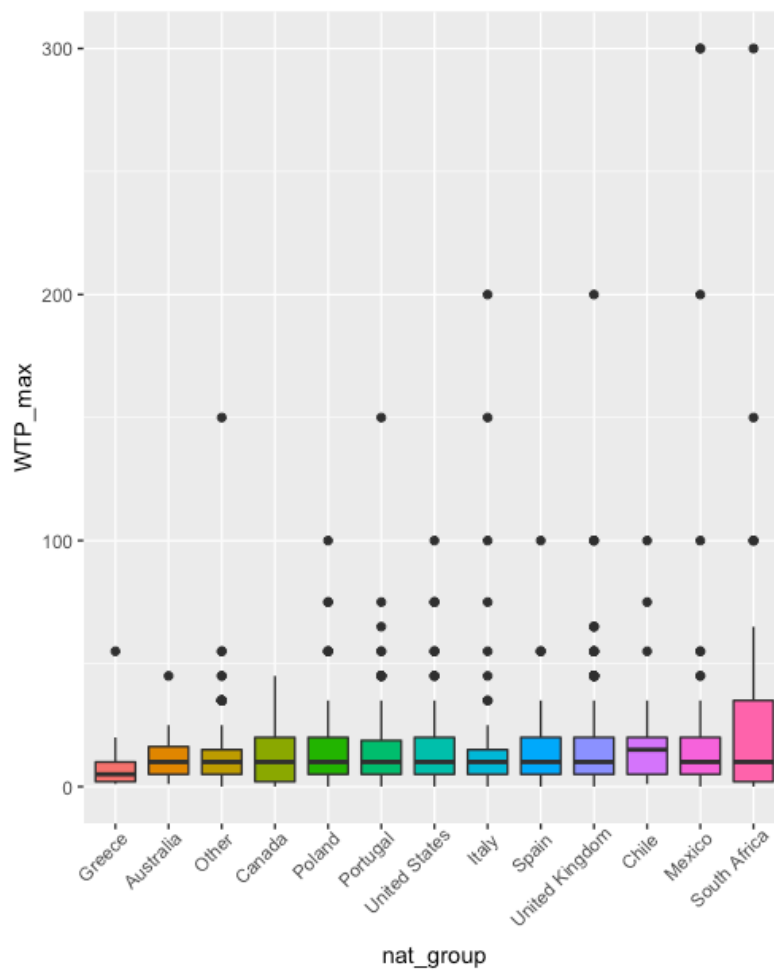
- * Bank sampah?
 - Plastic bank.
- Donation
 - Moving buds
 - Patrols for enforcement
 - School education / tips for school kids
- * Link to Pelita Foundation + other schools
 - environmental lessons
- * Motor GP → growth
 - ["Mandalika vision" → what does this mean?]

S9.4 Demographics and summary statistics for online WTP survey

Nationalities



Boxplot of WTP by nationality



Model output of WTP by nationality, controlling for income

Some significant differences, even when controlling for income

Residuals:

Min	1Q	Median	3Q	Max
-4.6746	-0.5463	0.1134	0.7762	3.5308

Coefficients:

	Estimate	Std. Error	t value	Pr(> t)	
(Intercept)	1.549e+00	2.453e-01	6.316	4e-10	***
nat_groupAustralia	3.295e-01	3.862e-01	0.853	0.39378	
nat_groupOther	2.175e-01	2.662e-01	0.817	0.41408	
nat_groupCanada	5.079e-02	3.053e-01	0.166	0.86790	
nat_groupPoland	1.352e-01	2.754e-01	0.491	0.62365	
nat_groupPortugal	4.100e-01	2.640e-01	1.553	0.12071	
nat_groupUnited States	2.428e-01	2.835e-01	0.856	0.39195	
nat_groupItaly	5.411e-01	2.851e-01	1.898	0.05794	.
nat_groupSpain	3.779e-01	3.426e-01	1.103	0.27022	
nat_groupUnited Kingdom	6.969e-01	2.674e-01	2.606	0.00928	**
nat_groupChile	7.539e-01	3.525e-01	2.139	0.03268	*
nat_groupMexico	6.374e-01	2.983e-01	2.137	0.03284	*
nat_groupSouth Africa	5.892e-01	3.518e-01	1.675	0.09425	.
income_num	3.457e-06	1.105e-06	3.129	0.00181	**

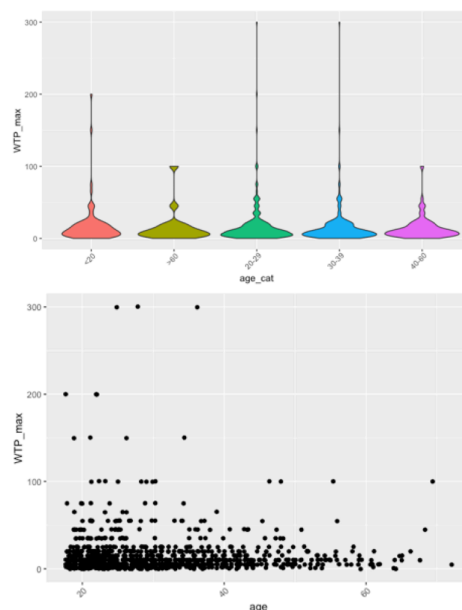
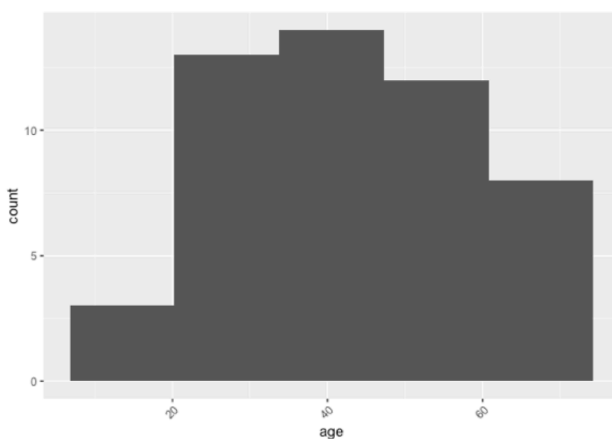
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 1.313 on 1019 degrees of freedom

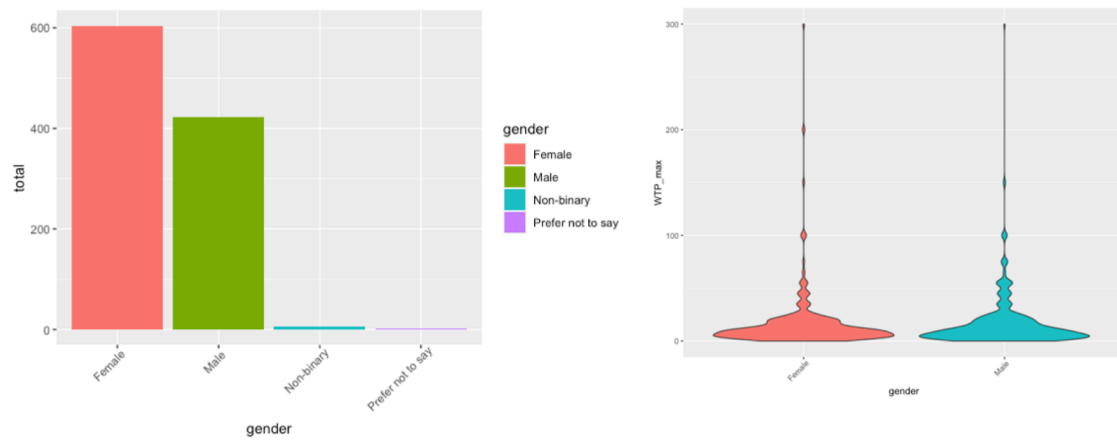
Multiple R-squared: 0.03573, Adjusted R-squared: 0.02343

F-statistic: 2.905 on 13 and 1019 DF, p-value: 0.000375

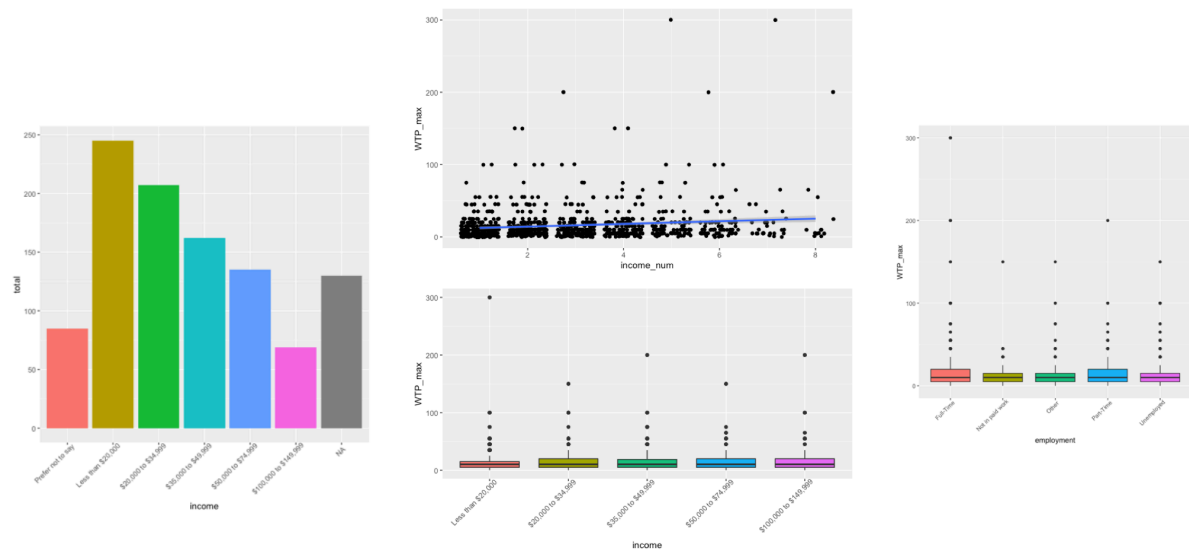
Age



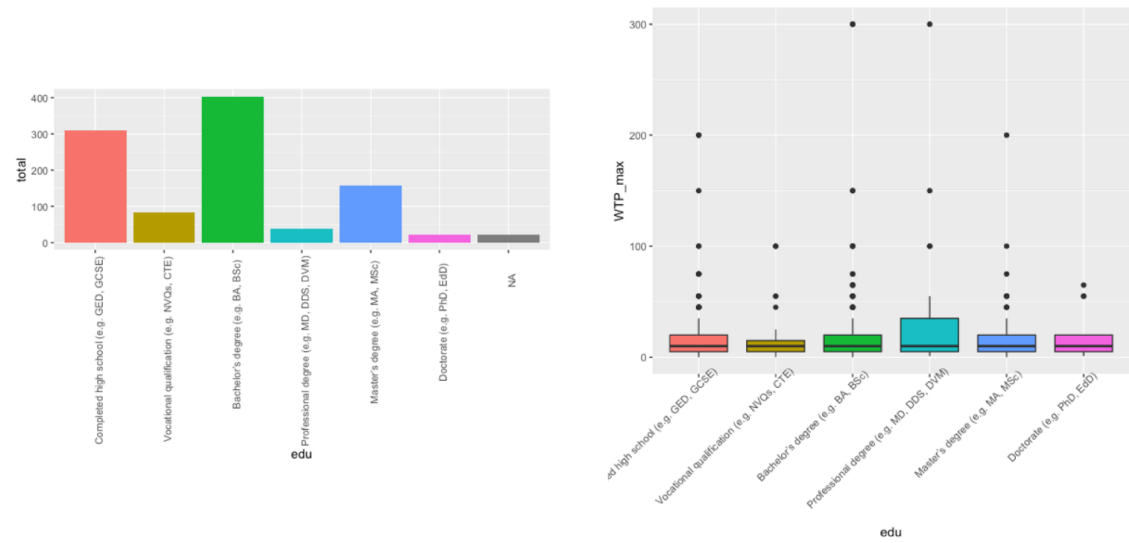
Gender



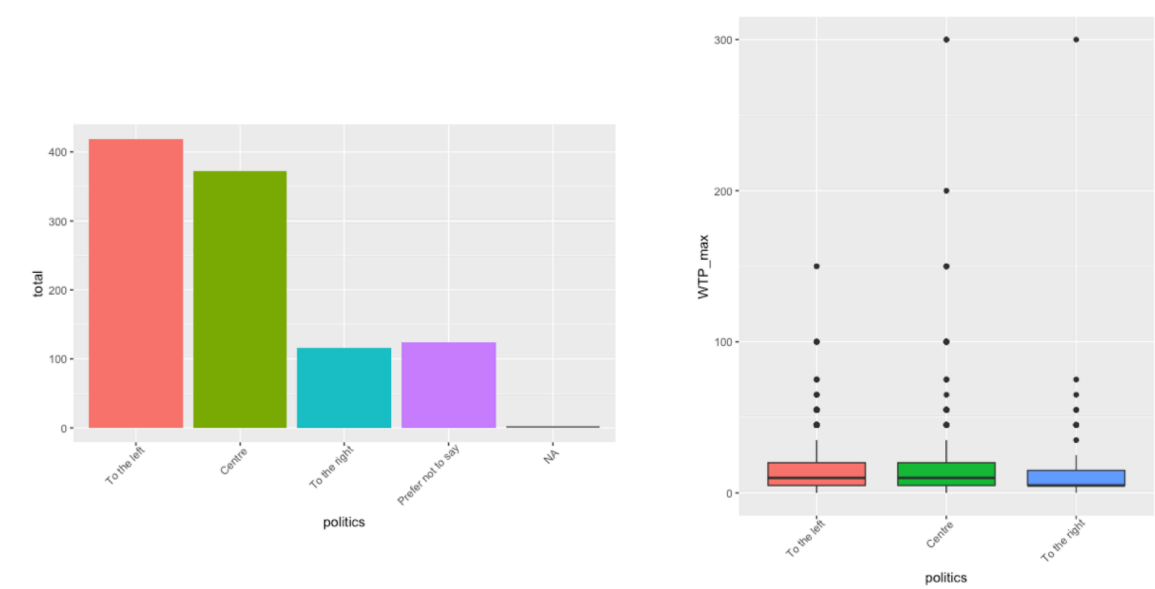
Income and employment



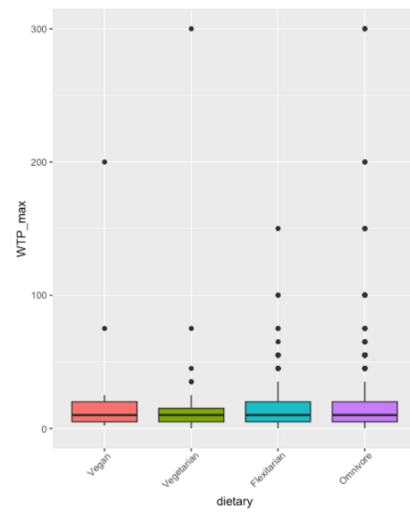
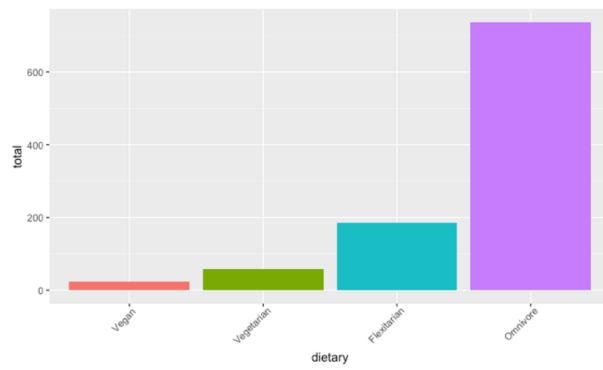
Education



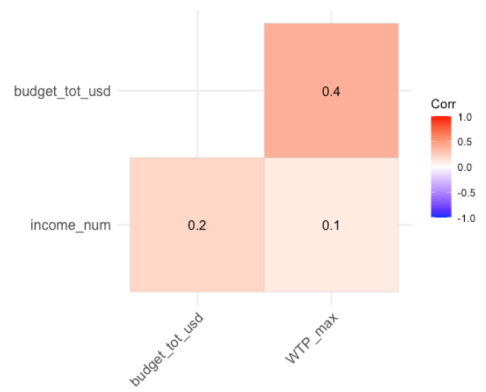
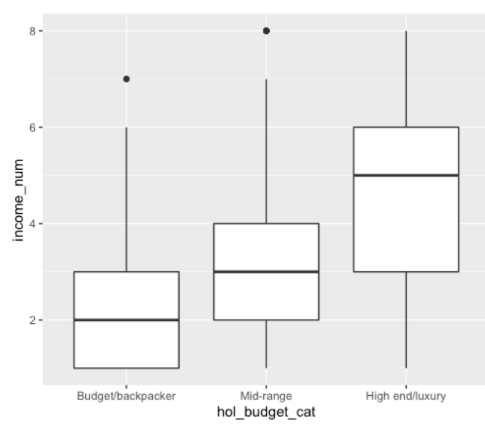
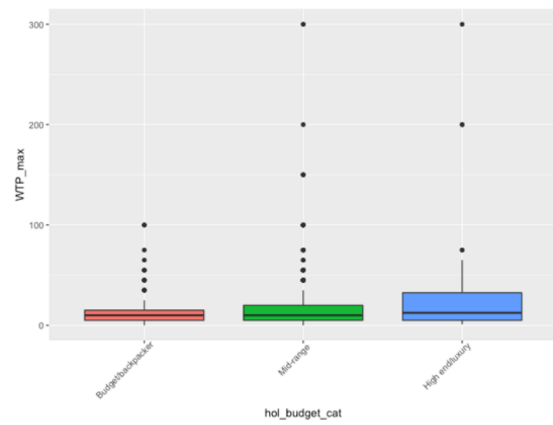
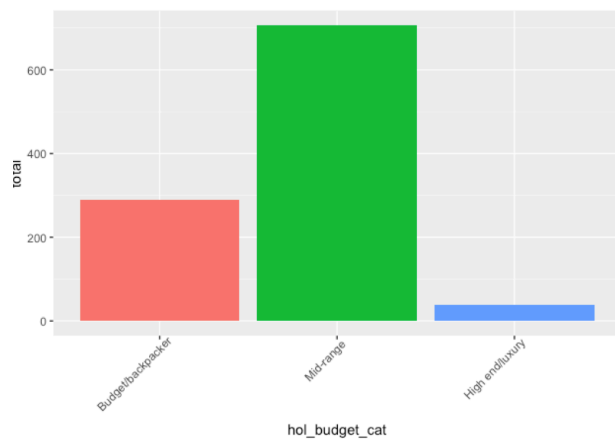
Political leanings



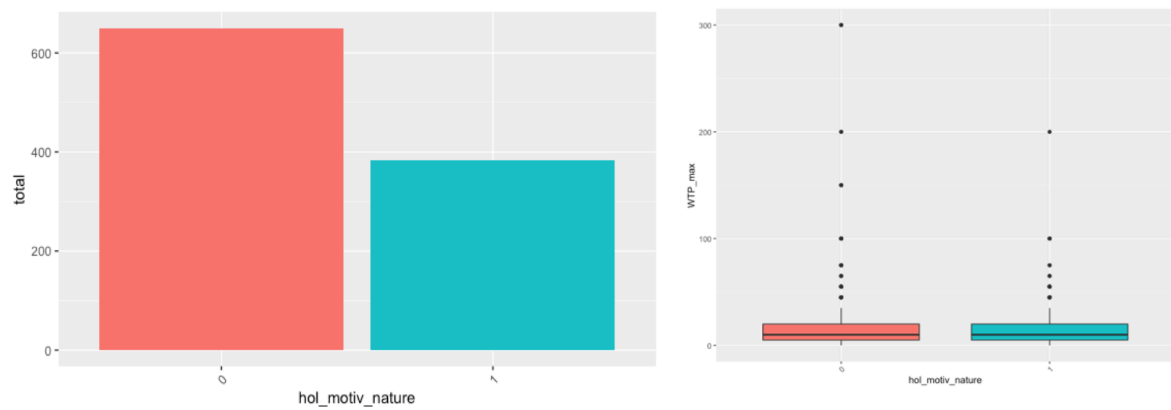
Dietary preferences



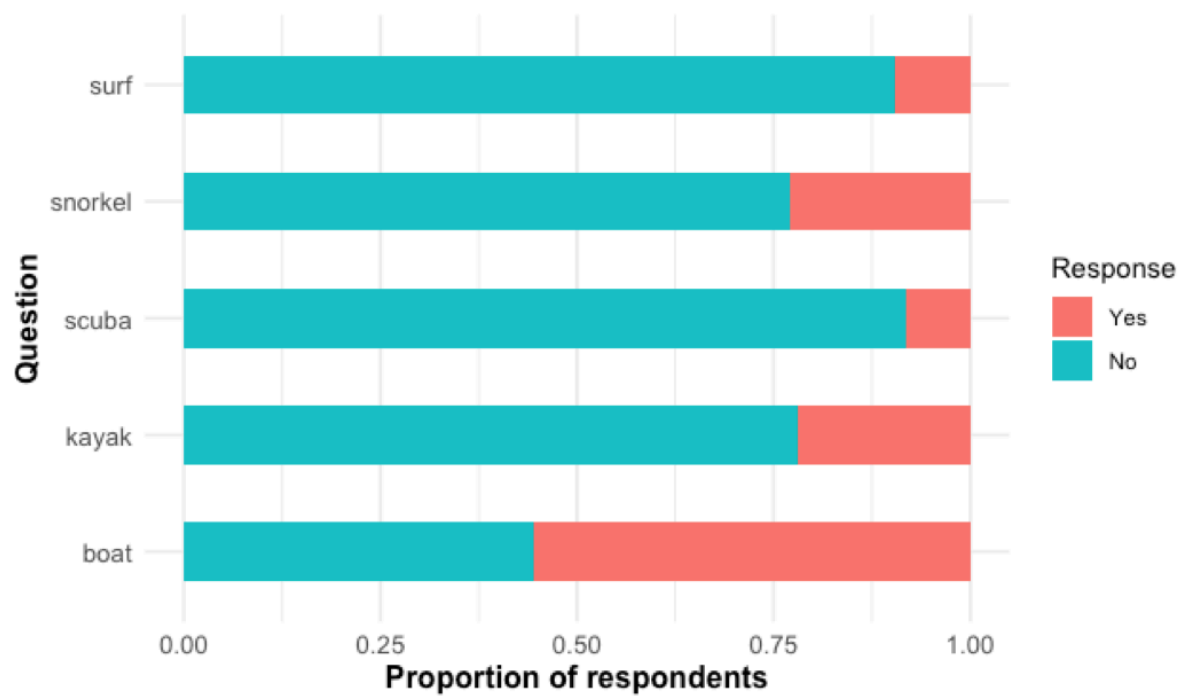
Holiday budget



Holiday motivations and activities

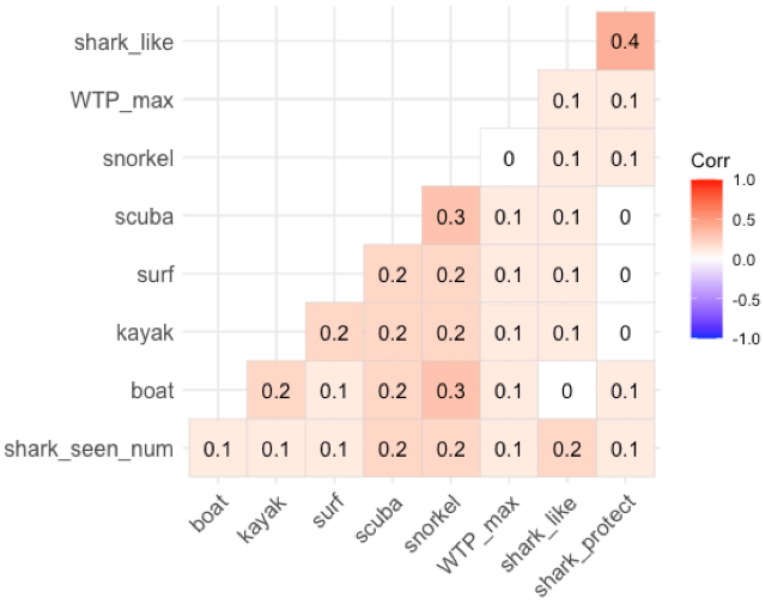
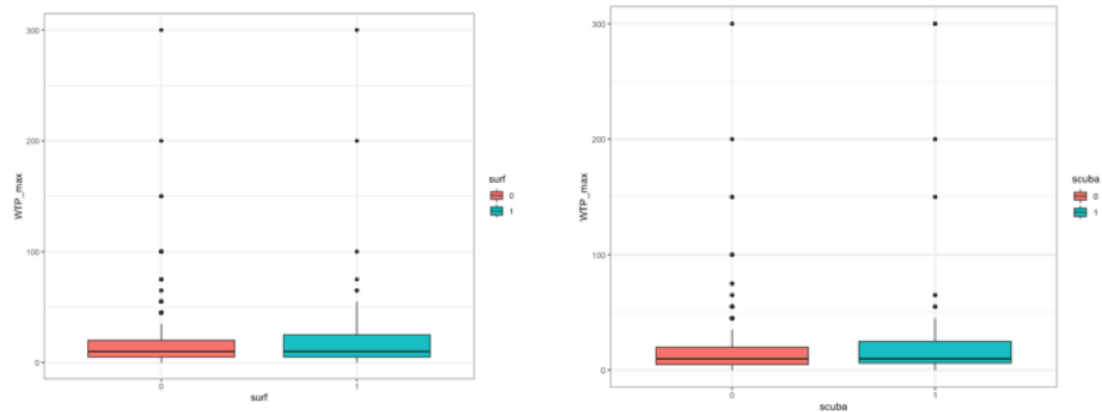


Marine activities



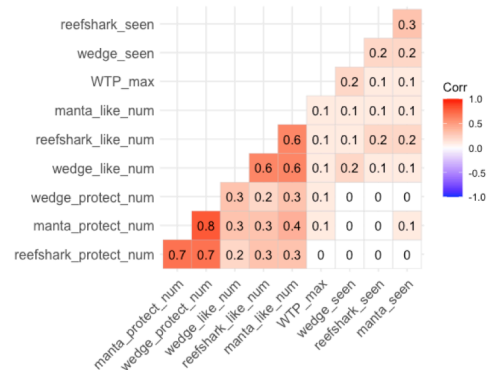
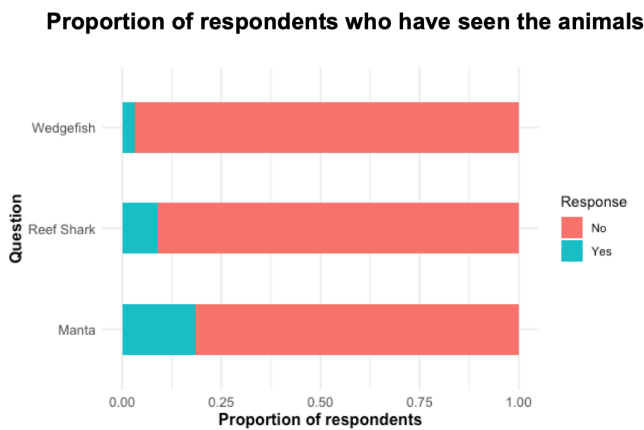
	P-value	Rho
Surf	0.04561	0.062
Snorkel	0.04082	0.064
Scuba	0.001414	0.099
Kayak	ND (0.7975)	
Boat	NS (0.2011)	

Slightly higher WTP for surfers & scuba divers

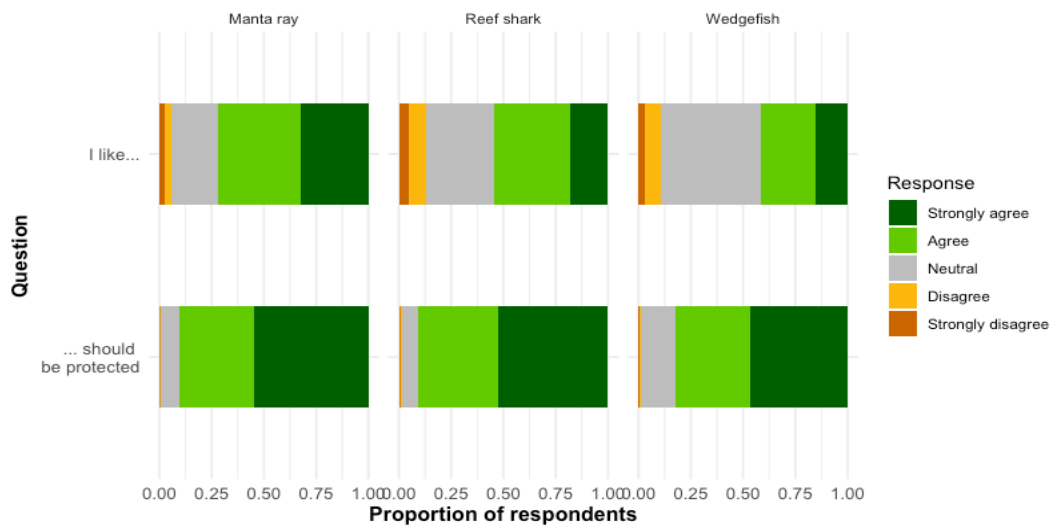


People who scuba and snorkel more likely to have seen sharks

Perceptions of marine animals

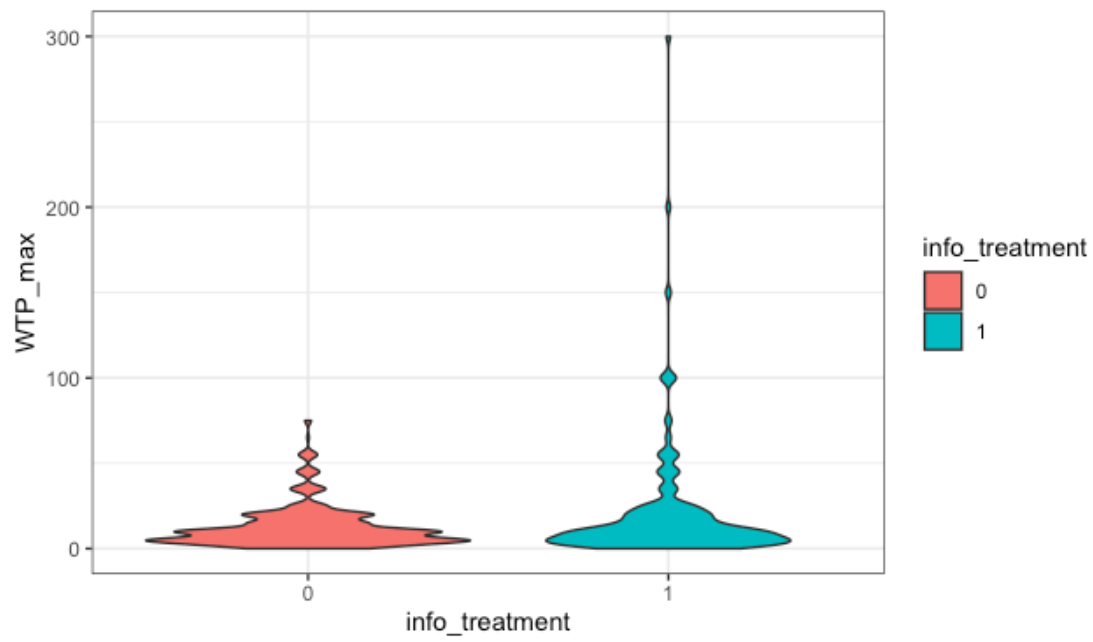


Correlations between preferences for different animals: people who want to protect one typically want to protect all, people who like one typically like all. People who like typically want to protect.

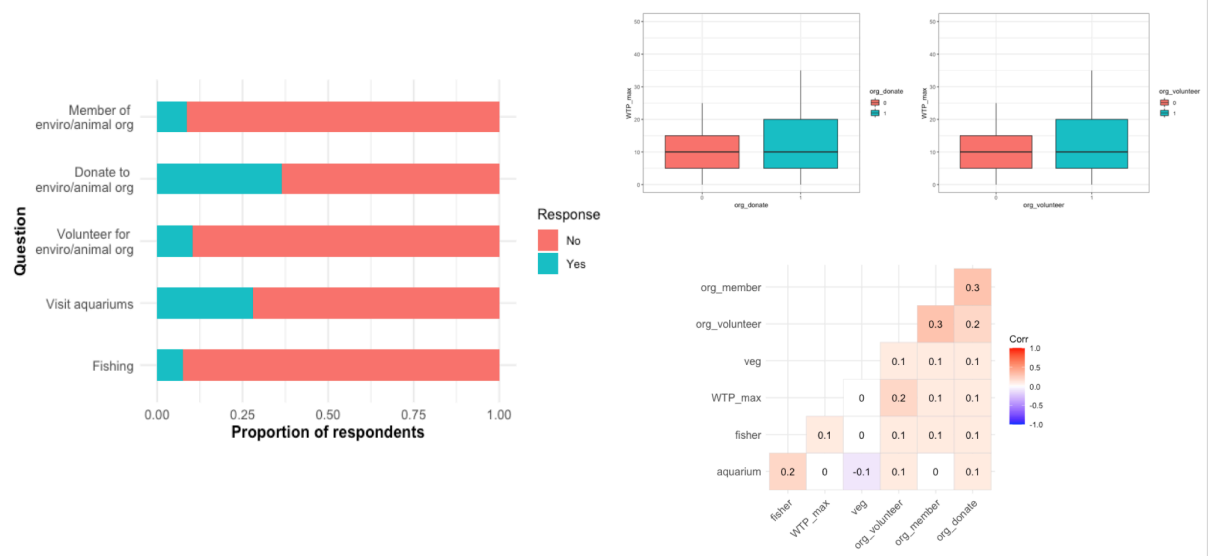


Informational intervention

No significant difference in mean WTP

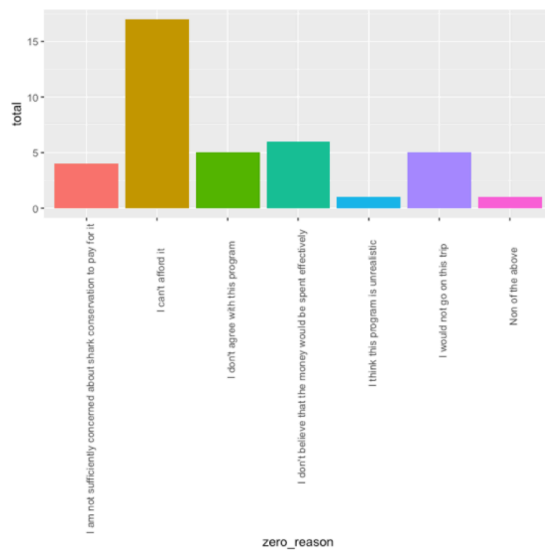


Existing pro-environmental behaviour



Respondents with zero WTP

39 zeros (3.8%)

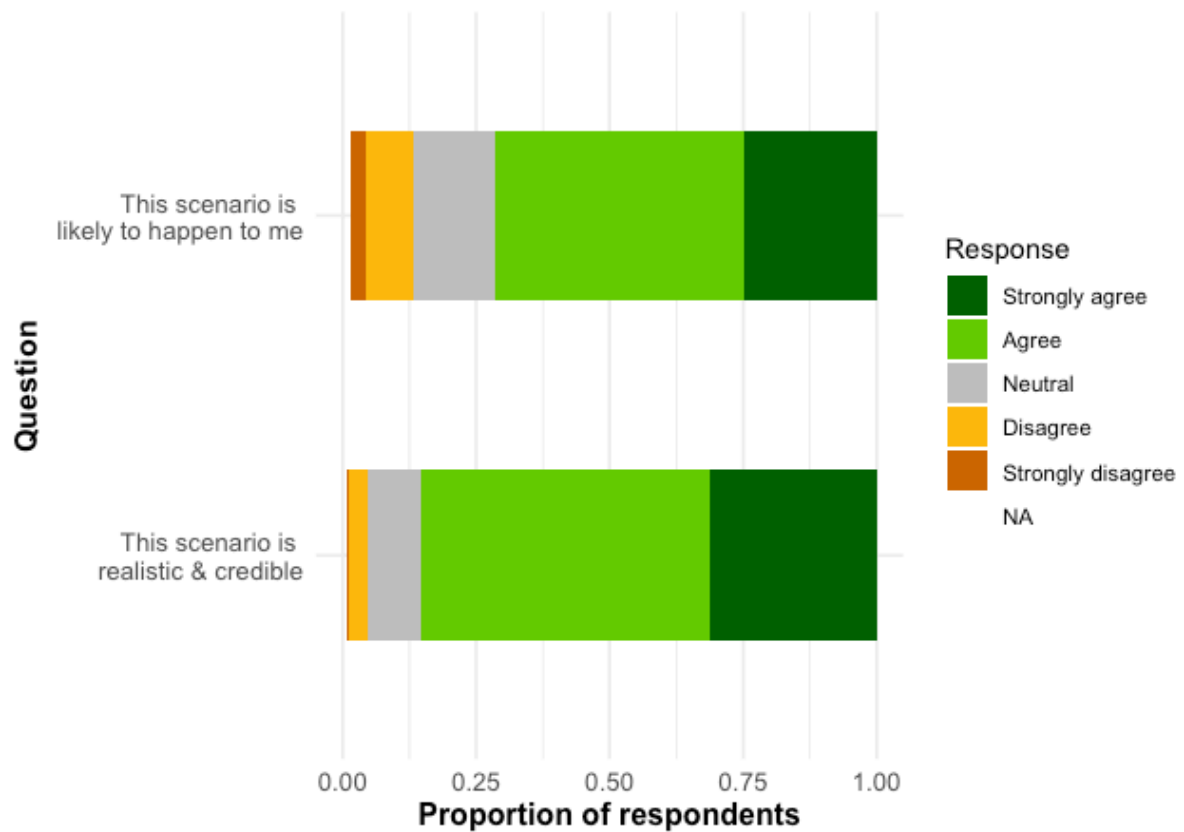


"We travel very economically"
"I am not a rich person"

"I think there would be a lot
of corruption"

"There are far more important problems
in the world"

Scenario consequentiality



S9.5 validation models

Variables considered

Candidate variables	Description
Travel preferences/behaviours	
- Motivations	No significant relationship between being motivated by nature and the outdoors and WTP
- Budget	Strong significant relationship between budget and WTP
- Activities	Small significant relationship for surfers, snorkelers and scuba-divers, though also correlates with sharks seen. Tried in first model but not significant.
Animal attitudes	
- Seen - Like - Protect	Positive relationships between preference/attitudes and WTP (as expected). Correlations between preferences for different animals: people who want to protect one typically want to protect all, people who like one typically like all. Makes sense to group for analysis.
Existing pro-environmental behaviours	
- Volunteer time or money to charities - Go fishing or to aquariums - Vegan or vegetarian	Covariance between members, volunteers, donors. Makes sense to group for analysis. No apparent relationship between people who fish or visit aquariums Vegan higher on average, but not significant in model.
Demographics	
- Income - Age - Gender - Nationality - Education/degree - Politics	As would be expected, significant correlation ($p < 0.001$, $\rho = 0.34$) between income and holiday budget. Tried interaction and tried both separately, budget seems better predictor. Nationality is not significant when other factors are taken into account.
Informational intervention	
- Background info on threats to sharks	Slight average difference but not significant

Modelling process

- Null model: AIC = 2778.906
- Model with everything: AIC = 2154.904
- More parsimonious model (After removing least significant variables)
 - With budget: AIC = 2140.988
 - With income: AIC = 2156.574

Model with everything

Coefficients:

	Estimate	Std. Error	t value	Pr(> t)
(Intercept)	1.526e+00	3.897e-01	3.917	9.92e-05 ***
budget_tot_usd	1.926e-05	8.532e-06	2.257	0.02432 *
income_num	1.649e-06	1.496e-06	1.102	0.27068
age	-2.047e-03	4.549e-03	-0.450	0.65280
genderMale	-2.087e-02	9.977e-02	-0.209	0.83441
genderNon-binary	9.555e-02	6.787e-01	0.141	0.88809
nat_groupCanada	2.227e-02	3.799e-01	0.059	0.95326
nat_groupChile	1.576e-01	4.306e-01	0.366	0.71447
nat_groupGreece	-2.126e-01	4.388e-01	-0.485	0.62819
nat_groupItaly	3.039e-02	3.739e-01	0.081	0.93524
nat_groupMexico	2.582e-01	3.870e-01	0.667	0.50487
nat_groupOther	-6.490e-02	3.525e-01	-0.184	0.85397
nat_groupPoland	5.524e-02	3.682e-01	0.150	0.88079
nat_groupPortugal	4.332e-02	3.557e-01	0.122	0.90312
nat_groupSouth Africa	2.722e-01	4.811e-01	0.566	0.57176
nat_groupSpain	-9.954e-02	4.339e-01	-0.229	0.81861
nat_groupUnited Kingdom	3.338e-01	3.512e-01	0.951	0.34214
nat_groupUnited States	1.238e-01	3.615e-01	0.343	0.73205
degreeY	4.238e-02	9.729e-02	0.436	0.66329
politicsTo the left	4.043e-02	1.013e-01	0.399	0.69009
politicsTo the right	-2.095e-01	1.447e-01	-1.448	0.14815
hol_motiv_nature	4.523e-02	9.295e-02	0.487	0.62669
scuba	2.057e-01	1.790e-01	1.149	0.25095
veg	-4.811e-02	2.904e-01	-0.166	0.86848
org_either	2.507e-01	9.240e-02	2.713	0.00684 **
shark_seenY	1.880e-01	1.140e-01	1.649	0.09954 .
shark_like	1.810e-02	6.409e-02	0.282	0.77771
shark_protect	1.720e-01	7.650e-02	2.248	0.02489 *
info_treatment	3.056e-02	8.982e-02	0.340	0.73381
budget_tot_usd:income_num	1.090e-10	1.414e-10	0.771	0.44110

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 1.14 on 655 degrees of freedom

(199 observations deleted due to missingness)

Multiple R-squared: 0.09743, Adjusted R-squared: 0.05747

F-statistic: 2.438 on 29 and 655 DF, p-value: 4.839e-05

More parsimonious with budget

Coefficients:

	Estimate	Std. Error	t value	Pr(> t)
(Intercept)	1.558e+00	1.857e-01	8.390	2.84e-16 ***
budget_tot_usd	2.671e-05	6.817e-06	3.918	9.84e-05 ***
age	1.863e-03	4.145e-03	0.449	0.65324
genderMale	-1.444e-02	9.231e-02	-0.156	0.87578
genderNon-binary	-7.033e-02	6.612e-01	-0.106	0.91532
degreeY	4.034e-02	9.337e-02	0.432	0.66581
politicsTo the left	7.097e-02	9.643e-02	0.736	0.46198
politicsTo the right	-2.085e-01	1.417e-01	-1.472	0.14150
org_either	2.510e-01	8.982e-02	2.794	0.00535 **
shark_seenY	2.670e-01	1.056e-01	2.528	0.01169 *
shark_protect	1.842e-01	6.831e-02	2.697	0.00717 **
info_treatment	6.410e-02	8.798e-02	0.729	0.46648

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 1.138 on 675 degrees of freedom

(197 observations deleted due to missingness)

Multiple R-squared: 0.07381, Adjusted R-squared: 0.05872

F-statistic: 4.89 on 11 and 675 DF, p-value: 2.616e-07

More parsimonious with income

Coefficients:

	Estimate	Std. Error	t value	Pr(> t)
(Intercept)	1.496e+00	1.865e-01	8.023	4.56e-15 ***
income_num	2.886e-06	1.126e-06	2.562	0.01062 *
age	7.691e-04	4.254e-03	0.181	0.85657
genderMale	-1.286e-02	9.258e-02	-0.139	0.88959
genderNon-binary	2.027e-02	6.647e-01	0.030	0.97568
degreeY	3.429e-02	9.396e-02	0.365	0.71525
politicsTo the left	3.905e-02	9.740e-02	0.401	0.68863
politicsTo the right	-1.577e-01	1.415e-01	-1.115	0.26543
org_either	2.835e-01	8.991e-02	3.153	0.00169 **
shark_seenY	2.645e-01	1.068e-01	2.477	0.01348 *
shark_protect	1.937e-01	6.848e-02	2.828	0.00482 **
info_treatment	7.280e-02	8.810e-02	0.826	0.40891

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 1.143 on 678 degrees of freedom

(194 observations deleted due to missingness)

Multiple R-squared: 0.06187, Adjusted R-squared: 0.04665

F-statistic: 4.065 on 11 and 678 DF, p-value: 8.576e-06

Interval model (using same variables as final OLS model)

Call:

```
survreg(formula = Y ~ budget_tot_usd + age + gender + degree +
  politics + org_either + shark_seen + shark_protect + info_treatment,
  data = cor_dem, dist = "gaussian")
```

	Value	Std. Error	z	p
(Intercept)	2.09e+00	1.42e-01	14.75	< 2e-16
budget_tot_usd	1.88e-05	1.13e-05	1.67	0.0955
age	-5.13e-04	3.19e-03	-0.16	0.8722
genderMale	2.39e-02	7.07e-02	0.34	0.7348
degreeY	1.89e-02	7.12e-02	0.26	0.7912
politicsCentre	-7.25e-02	7.32e-02	-0.99	0.3221
politicsTo the right	-1.83e-01	1.09e-01	-1.69	0.0915
org_either	1.46e-01	6.80e-02	2.15	0.0317
shark_seenY	2.10e-01	8.00e-02	2.62	0.0088
shark_protect	1.60e-01	5.21e-02	3.07	0.0021
info_treatment	8.38e-02	6.69e-02	1.25	0.2102
Log(scale)	-1.73e-01	2.84e-02	-6.09	1.1e-09

Scale= 0.841

Gaussian distribution

Loglik(model)= -1550.6 Loglik(intercept only)= -1568

Chisq= 34.75 on 10 degrees of freedom, p= 0.00014

Number of Newton-Raphson Iterations: 3

n=667 (89 observations deleted due to missingness)

Final OLS model with interpretation

- As expected, holiday budget and existing charitable behaviours are significantly correlated with WTP, as are stated positive attitudes re. shark protection
- People who have seen sharks in the wild also have significantly higher WTP – may be more motivated to protect something they have seen - has more value to them.
- Positive NS co-efficients for older, left-leaning university educated people, and for people who received the informational intervention
- Negative NS co-efficients for male participants, and right-leaning people

Coefficients:	Estimate	Std.Error	t value	p-value	Sig levels
(Intercept)	1.58	0.18	8.56	< 2e-16	***
Last holiday budget	2.70E-05	0.00	3.96	0.00	***
Age	0.002	0.004	0.54	0.59	
Gender - male	-0.01	0.09	-0.13	0.90	
Gender – non-binary	-0.06	0.66	-0.09	0.93	
University education	0.04	0.09	0.40	0.69	
Politics - left	0.07	0.10	0.75	0.46	
Politics - right	-0.20	0.14	-1.45	0.15	
Charitable	0.25	0.09	2.82	0.00	**

Seen sharks	0.27	0.11	2.52	0.01	*
Attitude - shark protection	0.19	0.07	2.73	0.01	**
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1 Residual standard error: 1.138 on 675 degrees of freedom Multiple R-squared: 0.07381, Adjusted R-squared: 0.05872 F-statistic: 4.89 on 11 and 675 DF, p-value: 2.616e-07					

Final interval model

	Value	Std.Error	z	p-value	Sig levels
(Intercept)	2.040	0.142	14.360	< 2e-16	
Last holiday budget	2.41E-05	5.14E-06	4.680	0.000	***
Age	-0.001	0.003	-0.280	0.780	
Gender - male	0.025	0.071	0.350	0.726	
University education	0.025	0.072	0.350	0.724	
Politics - left	0.053	0.074	0.710	0.476	
Politics - right	-0.131	0.109	-1.200	0.231	
Charitable	0.163	0.069	2.370	0.018	*
Seen sharks	0.234	0.081	2.900	0.004	**
Attitude - shark protection	0.172	0.053	3.250	0.001	**
Log(scale)	-0.159	0.028	-5.610	2.00E-08	
Scale=0.852					
Gaussian distribution Loglik(model)= -1565.9 Loglik(intercept only)= -1596 Chisq= 60.16 on 10 degrees of freedom, p= 3.4e-09 Number of Newton-Raphson Iterations: 3 n=670 (89 observations deleted due to missingness)					

Strong significant positive correlation between stated WTP & donation ($p < 0.001$)

