

Mitigating marine megafauna captures in fisheries



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Declaration

I declare that this thesis is entirely my own work, and except where
otherwise stated, describes my own research.

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Dedication

This thesis is dedicated to R. B. Arlidge.

Abstract

The most widely applied decision-making process for balancing the trade-offs between conservation and development activities is the biodiversity mitigation process, implemented using environmental impact assessment supported by a conceptual ‘mitigation hierarchy’ framework. Yet to date, the exploration of the biodiversity mitigation process to the primary resource sectors has not been widely investigated as a subject of study in conservation. In this thesis, I explore mitigating impacts from fisheries on marine megafauna, linking system-wide approaches with individual-level incentives in a unified framework.

The majority of this thesis focusses on a case study of sea turtle captures and mortalities in a coastal fishing community in Peru. A linked, but separate case study explores the application of the framework to all human impact on biodiversity more broadly. I begin at the broadest scale, by exploring challenges and solutions for a global mitigation hierarchy for nature conservation that could enable tracking of progress towards an agreed overarching objective, based on net conservation outcomes. The global framework research precedes an exploration of the biodiversity mitigation process in the case study coastal fishing system.

Throughout the fishery case study, I draw on established decision-making processes to better understand the conservation issue at hand and to develop an understanding of what is necessary to empirically calculate net outcomes in data-poor fishing systems using the proposed framework. The decision-making processes I employ include qualitative ecological risk assessment theory to assess the efficacy of current management systems, and a qualitative management strategy evaluation process to support consideration of trade-offs. I seek to further improve data gathering processes in data-poor fishing systems by applying the IDEA (“Investigate”, “Discuss”, “Estimate” and “Aggregate”) structured elicitation protocol to control for personal bias and heuristics when drawing on stakeholder knowledge. Finally, I characterise the social network of fishing-related information-sharing between fishers to inform understanding of social influences on decision making using network null models.

As humanity seeks to deliver nature conservation alongside development, broader perspectives on human impacts, and how best to mitigate them are needed. This research contributes to an important and timely dialogue that seeks to shift emphasis away from piecemeal actions that prevent biodiversity loss, and instead adopt a strategic and proactive approach to restoring nature that links broad scale concepts to locally tailored solutions.

Resumen

El proceso de toma de decisiones más ampliamente aplicado para balancear las compensaciones entre conservación y actividades de desarrollo es el proceso de mitigación de la biodiversidad, implementado mediante el uso de la evaluación del impacto ambiental apoyado por un marco conceptual de "jerarquía de mitigación". Aún, a la fecha, la exploración del proceso de mitigación de la biodiversidad para los sectores de recursos primarios no ha sido extensamente investigada como objeto de estudio en conservación. En esta tesis exploro la mitigación de los impactos de la pesquería en megafauna marina, relacionando enfoques a nivel sistémico con incentivos a nivel individual en un marco de referencia unificado.

La mayoría de esta tesis se enfoca en un caso de estudio de capturas y mortalidad de tortugas marinas en una comunidad pesquera de la costa peruana. Un caso asociado, pero separado, explora la aplicación del marco referencial a todo el impacto humano en la biodiversidad de manera más amplia. Empiezo en la escala más amplia, mediante la exploración de desafíos y soluciones de una jerarquía de mitigación global para la conservación que pudiese permitir el monitoreo del progreso hacia un objetivo general acordado, basado en resultados netos de conservación. La investigación del marco conceptual global precede a la exploración del proceso de mitigación de la biodiversidad en el caso de estudio del sistema de pesquería costero.

A lo largo del estudio de caso de pesca, me baso en procesos de toma de decisiones establecidos para entender mejor el problema de conservación en mano, y para desarrollar un entendimiento de lo que es necesario para calcular empíricamente resultados netos en sistemas pesqueros deficientes de datos utilizando el marco conceptual propuesto. El proceso de toma de decisiones que empleo incluye teoría de la evaluación cualitativa del riesgo ecológico para evaluar la eficacia de los sistemas de gestión actuales, y un proceso de evaluación cualitativa de la estrategia de gestión para sustentar la consideración de compensaciones. Busco mejorar aun más el proceso de recopilación de datos en sistemas pesqueros deficientes de datos mediante la aplicación del protocolo de obtención estructurado IDEA ("Investiga", "Discute", "Estima", y "Agrega") para controlar el sesgo personal y heurística de basarse en el conocimiento de los actores involucrados. Finalmente, caracterizo la red social de intercambio de información relacionada a la pesca entre pescadores para

informar entendimiento de influencias sociales en la toma de decisiones utilizando una red de modelos nulos.

Mientras que la humanidad busca lograr la conservación de la naturaleza de la mano con el desarrollo, se hace necesaria una perspectiva más amplia de los impactos humanos y la mejor forma para mitigarlos. Esta investigación contribuye a un importante y oportuno diálogo que busca alejar el énfasis en acciones poco sistemáticas que previenen la pérdida de biodiversidad, y en su lugar, adoptar un enfoque estratégico y proactivo para la restauración de la naturaleza que enlace conceptos de gran escala a soluciones adaptadas a escala local.

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List of acronyms and abbreviations

AIC	Akaike Information Criterion
CBD	Convention on Biological Diversity
CPUE	Catch Per Unit Effort
EBM	Ecosystem-based management
EBFM	Ecosystem-based fisheries management
EEZ	Exclusive Economic Zone
EIA	Environmental Impact Assessment
ERA	Ecological Risk Assessment
ERAEF	Ecological Risk Assessment for the Effects of Fishing
ESIA	Environmental Social Impact Assessment
FAO	Food and Agriculture Organisation
FGD	Focus Group Discussion
GLMM	Generalised Linear Mixed Model
GRT	Gross Registered Tonnage
OECMs	Other effective area-based conservation measures
IG	[San Jose] inshore gillnet fleet
IMG	[San Jose] inshore-midwater gillnet fleet
IUCN	International union for the conservation of nature
LOSC	[United Nations] Law of the Sea Convention
MPA	Marine Protected Area
MSY	Maximum Sustainable Yield
NEPA	National Environmental Policy Act
NG	Net Gain
NNL	No Net Loss
NOAA	[USA] National oceanographic and atmospheric organisation
PBR	Potential Biological Removal
PES	Payments for Ecosystem Services
RFMO	Regional Fisheries Management Organisation
RMU	Regional Management Unit
PA	Protected Area
SDGs	Sustainable Development Goals
SD	Standard deviation
TAC	Total allowable catch
USA	United States of America
UN	United Nations

Chapter 1

Introduction

1.1 Background

Preventing the global degradation of ecosystems and the extinction of the species that comprise them is essential for maintaining Earth's system processes (Worm et al. 2006; Cardinale et al. 2012). The conservation of biological diversity (herein 'biodiversity') is the core tenet of the United Nations (UN) Convention on Biological Diversity (CBD)'s 2050 vision – to sustain a healthy planet that delivers benefits such as food, energy, medicines, and a variety of materials fundamental for people's physical and cultural wellbeing (CBD 2018). Despite large-scale conservation efforts, biodiversity loss is now occurring at the fastest rate in human history. Currently, there are an estimated one million animal and plant species threatened with extinction across terrestrial and marine systems (IPBES [Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services] 2019).

Primary pressures causing biodiversity loss include the over-exploitation of species, habitat modification, invasive alien species and disease, pollution, and climate change – with the main human activities currently driving these pressures in marine system including fishing, coastal development, shipping, and energy production (Halpern et al. 2015), and farming, logging, and infrastructure development in terrestrial systems (Maxwell et al. 2016). These pressures can lead to the loss of biodiversity in the form of global species extinctions, or regional or local extirpations of species that results in a loss of genetic diversity, or the loss of ecosystem functionality – the latter through the reduction of species abundances to the point where they no longer play a meaningful role in the energy flow or structuring of their community (McCauley et al. 2015; Akçakaya et al. 2019).

Overfishing is currently the primary human disturbance to marine biodiversity (Jackson et al. 2001; Maxwell et al. 2016). The harvesting of marine life in the pelagic environment began at least 42,000 years ago (O'Connor et al. 2011). Nevertheless, it is only in the last century, with the advent of industrial fishing and the rapid expansion of coastal populations, that the rate of marine defaunation has intensified to such a level as to result in large-scale loss of species abundance (Jackson et al. 2001). This recent and rapid expansion of

pressures on the marine realm has left only ~13% of seascapes that are biologically and ecologically intact and that are mostly free of human disturbance (Jones et al. 2018). Though humans have caused fewer global marine extinctions than terrestrial extinctions (19–24 out of a total of >850 recorded extinctions; Webb & Mindel 2015), we have profoundly affected marine wildlife. Humans have altered the functioning and provisioning of ecosystem services, which are defined as the benefits people obtain from ecosystems, in every ocean (Carpenter et al. 2006; McCauley et al. 2015). The defaunation of marine organisms has also caused numerous cases of species extirpations (i.e., local extinctions resulting in range contractions). The lower rates of global species extinction in marine systems are also partly explained by lower rates of conservation assessment. In the best-studied taxonomic groups, 20%–25% of species are threatened with extinction, regardless of whether they are marine or terrestrial (Webb & Mindel 2015).

1.1.1 International management of biodiversity loss

There has been global recognition of the importance of conserving biodiversity for decades, with international agreements such as the Ramsar Convention on Wetlands (Carp 1972), the Convention on International Trade in Endangered Species (IUCN 1973), the Convention on the Conservation of Migratory Species (UNEP 1979), the Convention on the Law of the Sea (United Nations 1982), and the Convention on Biological Diversity (CBD; United Nations 1992).

In 2002, 193 parties to the CBD agreed that by the end of the decade, humanity would achieve a significant reduction of the current rate of biodiversity loss (CBD 2002). In 2010, in the face of continued biodiversity decline (Butchart et al. 2010), the Parties of the CBD adopted 20 “Aichi Targets” to be met by 2020, which set more specific biodiversity conservation aims (CBD [Convention on Biological Diversity] 2010). In 2015, global ambitions to conserve biodiversity further developed as the UN agreed on 169 targets grouped into 17 high-level Sustainable Development Goals (SDGs; United Nations 2015b). The SDGs focus on solving some of the major challenges humanity currently faces, including those related to poverty, inequality, climate change, environmental degradation, peace, and justice.

The SDGs integrated many of the Aichi Targets. For instance, Aichi Target 11, by 2020, commits governments to conserve $\geq 17\%$ of terrestrial and $\geq 10\%$ of marine environments globally, especially areas of particular importance for biodiversity, through

ecologically representative protected area systems, or ‘other effective area-based conservation measures’ (OECMs). SDG 14 and 15 have incorporated these area-based global protected area targets. Goal 14: Life Below Water aims to “conserve and sustainably use the oceans, seas and marine resources for sustainable development”. Targets within SDG 14 range across marine area protection (e.g., protect $\geq 10\%$ of marine environments, globally), fisheries management, pollution control, and knowledge transfer. Goal 15: Life on Land aims to “protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, halt and reverse land degradation, and halt biodiversity loss”. Together, SDG 14 and 15 can be considered the primary goals for biodiversity conservation. However, the interconnected nature of the natural environment means that all the SDGs are directly or indirectly related to one another in some way. To illustrate, when considering how to achieve ecologically and socioeconomically sustainable fisheries (SDG 14.4), other SDGs feed into its achievement, including zero hunger (SDG 2), good health and wellbeing (SDG 3), decent work and economic growth (SDG 8), responsible consumption and production (SDG 12), and climate action (SDG 13).

As we near the 2020 deadline for achieving the CBD Aichi Targets, negotiations are underway for a more ambitious post-2020 Biodiversity Framework (e.g., CBD 2019). A series of calls to amplify commitments to protect, retain, and restore biodiversity highlight the urgency of the rapid and large-scale reform necessary to sustainably manage human-induced harmful impacts to the natural world (Wilson 2016; Watson & Venter 2017; Mace et al. 2018; Maron et al. 2018b; Visconti et al. 2019). These calls for a stronger ‘deal for nature’ are underscored by the recent release of the global assessment by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, which highlights biodiversity’s swift decline (IPBES [Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services] 2019).

To achieve the scale of conservation required to arrest the rapid degradation of biodiversity and deliver positive outcomes for nature, there is increasing recognition that as well as strengthening the planet’s Protected Area estate (Allan et al. 2019; Jones et al. 2020), a diverse array of actions beyond these formally protected areas is necessary (Boyd et al. 2008; Kok et al. 2018). For example, recognising and managing OECMs for protection such as urban green spaces, abandoned farmland, human-made marine structures, and indigenous lands and coastal zones (Poore 2016; Lepczyk et al. 2017; Renwick et al. 2017; Sommer et al. 2018).

Critically, humanity must focus on the central role the renewable resource sectors play in driving biodiversity loss (Jackson et al. 2001; Maxwell et al. 2016). If the current trends of the rising global population and increasing wealth continue, by 2050, the global demand for food, wood, water and energy has a projected increase of 1.5–2 times the global demand in 2010 (OECD 2012; van Vuuren et al. 2015; Riahi et al. 2017). Thus, biodiversity conservation and restoration, as well as sustainable resource use, must form an integral part of sustainable development strategies in the renewable resource sectors (Friedman et al. 2018; Kok et al. 2018).

1.1.2 Trade-offs and environmental impact assessment

Further integrating biodiversity conservation into sustainable economic development frontiers and ensuring their fair distribution can yield gross benefits (Turner et al. 2012; Kremen & Merenlender 2018), yet biodiversity conservation often takes place in areas involving multiple stakeholders with varying objectives, where trade-offs are inevitable (Sunderland et al. 2007; Henle et al. 2008; Sandker et al. 2009; Halpern et al. 2013; Costello et al. 2016a). To support the implementation of conservation interventions, researchers and natural resource managers use structured decision-making processes to evaluate the effects likely to arise from an action significantly affecting the environment (Schultz 2011).

The Environmental Impact Assessment (EIA) decision-making process attempts to make trade-offs explicit by quantifying when environmental impacts occur and measuring that loss against a pre-determined baseline. The emphasis of EIA is on anticipation and prevention of environmental impacts (Jay et al. 2007). Thus, the mitigation of environmental impacts is a key stage of the EIA process (Pritchard 1993).

EIA gained prominence in the United States at the start of the 1970s as a result of its legislation in the National Environmental Policy Act of 1969 (NEPA) following public concerns over pollution (NEPA; CEQ [Council on Environmental Quality] 2000). Since the early 1970s, mitigation (of wetlands) was initially prescribed a mitigation hierarchy of three action steps comprising avoidance, minimisation, and compensation (LaRoe 1986; Morgan & Hough 2016). The legislation gave rise to biodiversity offsetting and other ecological conservation market mechanisms to compensate for biodiversity loss, facilitated by the US 1972 Clean Water Act §404 permit program. This legislation allowed building permits to be issued provided developers created a certain amount of wetland to compensate for the loss of

the original ones (LaRoe 1986). The now commonly used umbrella term ‘biodiversity offsets’ was only coined in 2003 (ten Kate et al. 2004), inspired by carbon offset markets, made possible through the Kyoto Protocol on Climate Change (United Nations 1997).

The sound legislative basis of EIA has, in part, resulted in its legal and institutional force in nearly every country; currently, 191 of the 193 member nations of the UN either have national legislation or have signed some form of an international legal instrument that refers to the use of EIA (Morgan 2012). The EIA approach has expanded to mitigate adverse impacts on biodiversity and ecosystem services beyond wetlands (Jay et al. 2007). So much so, that the CBD and SDGs cite the EIA process as critical to achieving their goals (United Nations 1992; Slootweg et al. 2009; United Nations 2015b).

1.1.3 The mitigation hierarchy

EIA practitioners seek to mitigate impacts using a conceptual framework known as the mitigation hierarchy (CEQ [Council on Environmental Quality] 2000; Hough & Robertson 2009). Practitioners must first avoid negative impacts where possible, then minimise negative impacts when an action is taking place. Avoidance and minimisation actions are followed by ensuring that the losses that do occur are more than fully compensated for through remedial actions that take place at the impact site, and finally, accounting for any residual impacts by implementing biodiversity offsetting actions away from the impact site (Figure 1.1). This prescribed sequential framework is structured so that overall nature is retained or restored in net terms (CEQ [Council on Environmental Quality] 2000; ten Kate et al. 2004). If implemented correctly, the EIA process has been demonstrated to result in an overall ‘no net loss’ (NNL) of biodiversity following the environmental impacts of a project or activity (zu Ermgassen et al. 2019a).

1.1.4 Challenges and opportunities for net outcome policies

Many elements of biodiversity mitigation present theoretical and practical challenges and the process can bring difficult questions concerning conservation, human development, and sustainability to the fore (Bull et al. 2013). Of all the steps of the mitigation hierarchy, biodiversity offsetting is arguably the most controversial, because offsets require an acceptance of biodiversity loss for uncertain gains in the future (Bull et al. 2013).

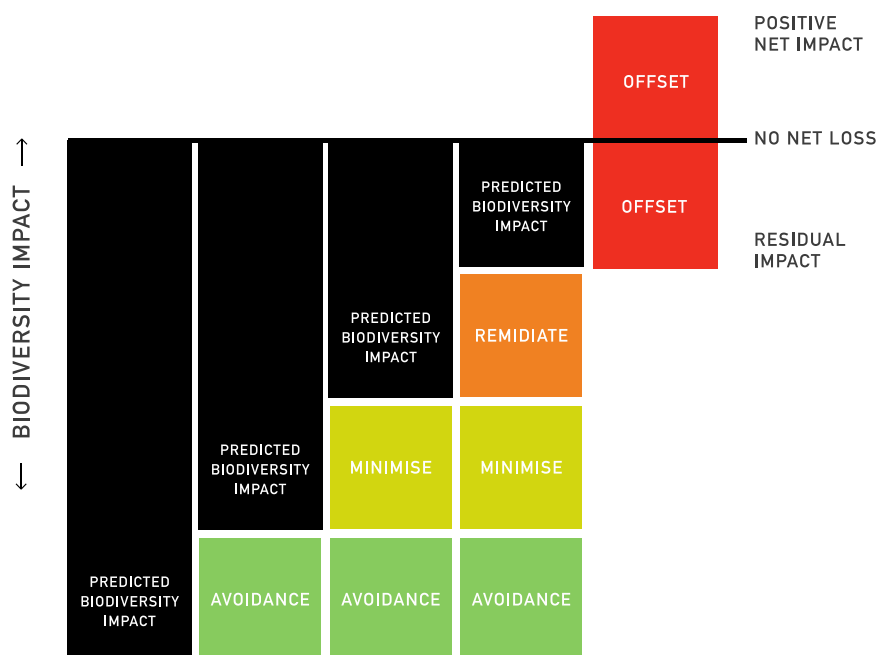


Figure 1.1. A depiction of the mitigation hierarchy framework for biodiversity mitigation. The size of the boxes is arbitrary and likely highly context-specific. Adaptation of this figure is after BBOP 2013 and Ten Kate et al. 2014.

Criticisms of biodiversity offsets centre around the ethics of biodiversity trading and commodification of nature (Apostolopoulou & Adams 2015; Ives & Bekessy 2015; Spash 2015), the choice of metrics to effectively measure biodiversity (Bull et al. 2014), the potential for sidestepping conservation obligations (“license to trash”) through perverse incentives (Gordon et al. 2015; Maron 2015), and considerations of social equity (Mandle et al. 2015). Currently, there is a notable lack of evidence regarding the actual outcomes of net outcome policies to address many of these debates (zu Ermgassen et al. 2019b). The lack of quantitative evidence on the achievement of net outcomes is, in part, due to the policies and biodiversity lag times (zu Ermgassen et al. 2019a), barriers with data transparency (Bull et al. 2018), and challenges evaluating the negotiation stages of the EIA process (most significantly surrounding the identifying of avoided impacts; Figure 1.1; Sinclair 2018).

Proponents for biodiversity offsets note that if correctly framed within the context of the mitigation hierarchy, offsetting is a tool with a specific purpose, for addressing residual impacts on biodiversity caused by economic development after avoidance, mitigation and remediation measures have occurred (IUCN 2016a). Data on global biodiversity offset

implementation is building (Bull & Strange 2018), and the understanding of the mechanisms that result in biodiversity offset failings continue to improve (Bull et al. 2013; Bull et al. 2014; Maron et al. 2016; Lindenmayer et al. 2017; Simmonds et al. 2019). Critically, without the approach, negative residual impacts to biodiversity from human developments would otherwise be unaddressed and left to accumulate – hailing the “death by one thousand cuts” debate (von Hase & ten Kate 2017).

Net outcome policies remain theoretically attractive because they provide a mechanism to help navigate trade-offs between development and conservation; however, in practice, EIA is a fine-scale tool. Thus, net outcome goals are predominantly applied on a project-by-project basis, which can underestimate the cumulative impacts of multiple current or projected development projects within an area, and also limit flexibility in applying the approach (Kiesecker et al. 2009; Maron et al. 2018a). Furthermore, the EIA process does not provide direct insight into welfare gains and losses or optimum levels of conservation (Braat & De Groot 2012; Squires et al. 2018).

To be fully effective, the mitigation hierarchy should be applied in a coherent manner, accounting for cross-sectoral and cumulative effects both in space and time (Kiesecker et al. 2009; Squires & Garcia 2018b). As the concept continues to be applied to nations government policy (IUCN 2016b; Maron et al. 2016; Shumway et al. 2018), business performance standards (IFC 2012) and novel sectors (Aiama et al. 2015; OECD 2016; Milner-Gulland et al. 2018; Squires et al. 2018), further investigation into a more strategic approach that nests project level application within multi-scale ecologically and politically relevant frameworks is warranted.

1.2 A case study of marine megafauna conservation in fisheries

Traditional fisheries management targets species in the geographically and temporally confined area where the fishery operates (Hanna 1999). Managing fish stocks, according to traditional fisheries management principles (FAO 1995), can lead to sustainably harvested target species on a single-species basis (Hilborn & Ovando 2014). However, there is now increasing recognition of the need for a broader systems perspective beyond the single-species approach to fisheries management – one that integrates biological, economic and social considerations – to cope with intensifying system pressures from human population increase and climate change (Crowder & Norse 2008; Halpern et al. 2013; Arbo & Thũy 2016).

Systems thinking, when focused on fisheries, is known as ecosystem-based fisheries management (EBFM; FAO 2003; Pikitch et al. 2004), and when integrated across multiple marine uses, ecosystem-based management (EBM; Christie et al. 2007). Over the last 15 years of research into understanding and operationalising broader systems thinking approaches in fisheries, some rapid advances in understanding and implementation were made (e.g., the integration of system-level ecosystem indicators into fisheries management models can outperform single species models in practice; Link 2017; Fulton et al. 2019). However, as the scope for management broadens across regions, species, and scales, the need for guidance in using the multitude of decision-making tools and processes available to fisheries management is increasingly necessary.

While EBFM is not designed to manage only data rich fisheries, it is data rich fisheries where the majority of EBFM research and practical operationalisation has occurred to date (Barnes & McFadden 2008; Fulton et al. 2011a; Fulton et al. 2014; Smith et al. 2017). Nevertheless, while our collective experience is reduced when applying EBFM to data-poor fisheries, methods and understanding in applying the approach when data are limited continues to develop (Smith et al. 2004; Smith et al. 2007). Critically, it is often not the lack of data that is the issue in implementing EBFM. Instead, a more important hurdle can be the need for fisheries managers and practitioners to use adaptive methods and consider the available information across a range of factors, shifting thought from single-system processes to whole-of-system outlooks (Patrick & Link 2015).

One of the major challenges in broadening fisheries management to an EBFM approach is that, as a management issue traverses an increasing number of stakeholders, sectors, and nations, the complexity increases (Dichmont & Fulton 2017). Natural resources such as fisheries are national or global public assets that are entrusted to only a few individuals, communities, or businesses through rights or rules (Gordon 1954; Schlager & Ostrom 1999). The objectives of resource users, whether these be profit maximising, or harvesting for subsistence, must then be traded off against the medium-term environmental impacts from fishing, and long-term intergenerational equity implications of exploiting species and ecosystems (Grafton et al. 2007; Squires et al. 2016). This temporal variability makes the task of clearly defining goals, targets and trade-offs inherently complex (Hilborn 2007). In data-poor fishing systems, the resulting high uncertainty can drive complexity higher still (Dutton & Squires 2008).

A particularly challenging conservation issue that encapsulates many of these complexities is managing the recovery of depleted populations of marine megafauna species (Hall et al. 2000; Gray & Kennelly 2018; Lewison et al. 2018). The term marine megafauna defines large-bodied ocean dwellers like sea turtles, seabirds, marine mammals and sharks. Marine megafauna species have experienced substantial declines in their population numbers in many ocean regions (Wallace et al. 2011; Davidson et al. 2012; Dulvy et al. 2014; Paleczny et al. 2015). Sea turtles, for example, have historically had significant population declines across the globe and nearly two-thirds of the 58 specified sea turtle regional management units (RMUs) remain subject to high threats; 11 RMU sea turtle populations are in a critical state threatened with extinction (Wallace et al. 2010a; Wallace et al. 2011). Elasmobranchs are now one of the world's most threatened species groups (Dulvy et al. 2014), and many individual marine megafauna species groups or populations have suffered severe depletion (Lewison et al. 2014). The dramatic population declines of many marine megafauna populations can at least, in part, be explained by the conservative life-history characteristics many of these species have, including longevity, late sexual maturity, low fecundity, and large migratory ocean ranges. These characteristics can increase a species' susceptibility to continual direct impacts, such as bycatch in fisheries (the portion of the capture discarded at sea dead or injured to a point where death results; Hall 1996), and indirect impacts to nesting sites or other essential habitat areas occupied during their life cycles (Dutton & Squires 2008). Marine megafauna species' life-history traits, and the multitude of threats they face throughout their life cycles, makes managing the recovery of their populations a complicated task. Conservation efforts spanning long periods, multiple nations, and requiring involvement from multiple stakeholders and sectors is required, particularly for transboundary species such as sea turtles.

As the management focus in fisheries has been broadening to EBFM, there has been parallel interest in the potential marine conservation applications of economic incentive and market-based policy instruments applied in terrestrial systems, such as biodiversity mitigation, taxes or fees, and ecolabelling (Hall 1996; Wilcox & Donlan 2007; Ferraro & Gjertsen 2009; Gjertsen et al. 2014; Gelcich & Donlan 2015; Innes et al. 2015; Milner-Gulland et al. 2018; Squires & Garcia 2018b; Squires et al. 2018). Such thinking provides an opportunity for new problem-solving perspectives to manage fisheries' impact on marine megafauna and can precipitate the integration or broad uptake of new mitigation options into fisheries management (Milner-Gulland et al. 2018; Squires et al. 2018). Taking a broader

perspective to possible management actions, in turn, offers the potential to enhance the conservation status of vulnerable bycatch species in innovative ways that address a range of externalities such as user cost, transnational, public good, and information failure, which currently result in unaccounted and unaddressed adverse impacts to marine megafauna (Milner-Gulland et al. 2018; Squires et al. 2018).

1.2.1 Small-scale fisheries

One particularly intractable issue when managing the population recovery of marine megafauna populations is tackling the pressures from coastal, small-scale fisheries, where marine megafauna can be either incidentally captured and discarded as bycatch, or, used as non-target catch (Dutton & Squires 2008). The permitted direct take of marine megafauna still occurs, for example, the legal capture of sea turtles in over 42 countries and territories (Casale & Cannavò 2003; Poonian et al. 2008; Quiñones et al. 2017; Humber et al. 2014). This portion of the capture can be considered target catch (Figure 1.2). Due to the protected status of many marine megafauna species in waters throughout the world, such practices are not as widespread as they once were (e.g., a substantial reduction in the historically high sea turtle consumption rates in Peru followed legal protection in 1995, yet an illegal trade for turtle meat still exists; Morales & Vargas 1996; Quiñones et al. 2017). Capture and mortality in small-scale fisheries remains a key pressure driving population declines for many marine megafauna species, including sea turtles (Peckham et al. 2007; Alfaro-Shigueto et al. 2018), seabirds (Moreno et al. 2006), cetaceans (López et al. 2003; Mangel et al. 2010), and elasmobranchs (Alfaro-Cordova et al. 2017).

Small-scale fisheries comprise the vast majority of global fishers, account for 30 per cent of the global landed value (revenues at the dock), and in developing countries can be the primary source of food production while employing millions of people living in coastal communities (Sumaila 2018; Taylor et al. 2019). These fisheries are particularly prevalent in lower-income nations, where monitoring and management regulations are frequently underdeveloped, unenforced, or nonexistent (Berkes et al. 2001; Chuenpagdee 2011). While the capture of marine megafauna is known to be a major conservation issue in many small-scale fisheries (Peckham et al. 2007; Alfaro-Shigueto et al. 2018), many suffer from data paucity meaning the full magnitude of marine megafauna mortality in the small-scale fisheries subsector remains unknown.

Small-scale fisheries encompass traditional, low-technology, low-capital fishing methods, and are often decentralised, with many owner-operators spread across remote- or large-areas of coastline (Chuenpagdee 2011; Smith & Basurto 2019). These fisheries can comprise a mix of vessel types, gears, and species, the latter which fishers may not necessarily target, but are nonetheless retained and either sold at markets or consumed by the fishers and their families (Khalil et al. 2017; Alfaro-Shigueto et al. 2018). The heterogeneity in the make-up of small-scale fisheries can make understanding the conservation issue and implementing appropriate interventions a challenge (Dietz et al. 2003).

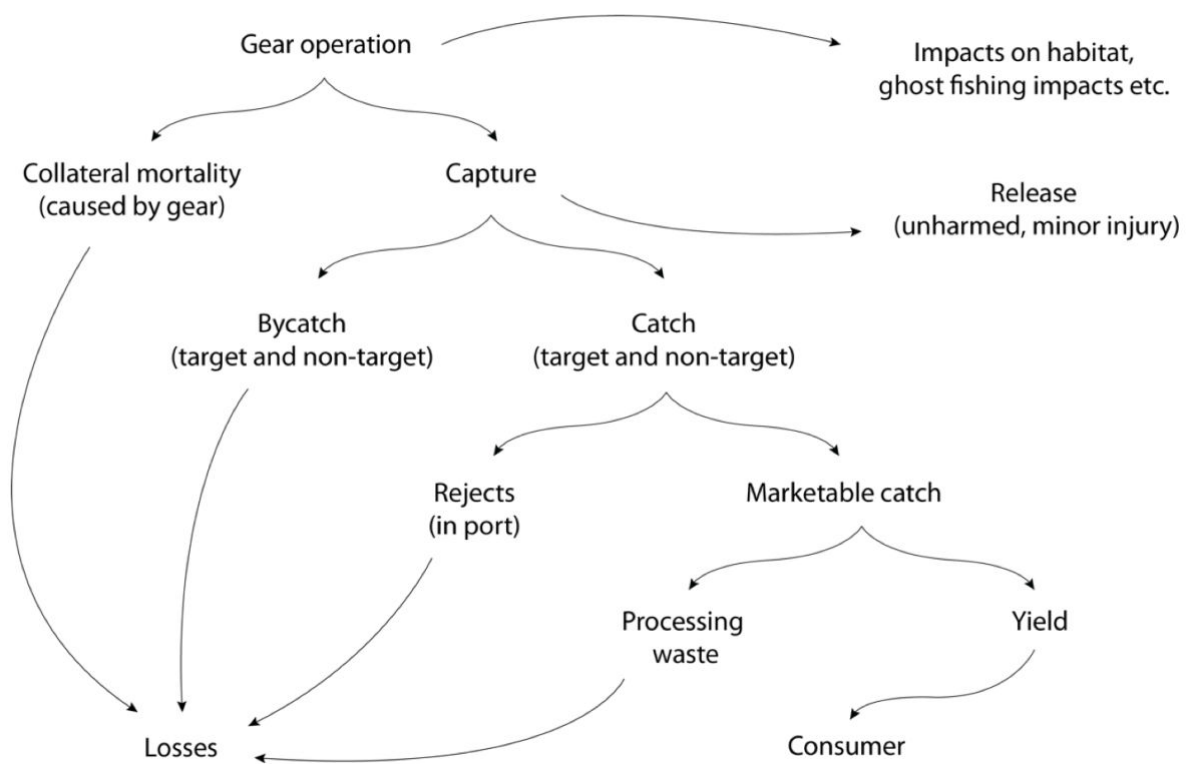


Figure 1.2. Ecological impacts of fishing operations illustrating divisions between capture, catch, and bycatch. Redrawn from Hall (1996).

1.2.2 Migration and jurisdiction

Another complex issue when managing the recovery of marine megafauna species is the large migratory ranges of many marine megafauna species, meaning that they move through the waters of multiple nations' exclusive economic zones (EEZs). Far ranging species create a

transboundary resource and jurisdictional issue because no central authority is in place to organise and enforce conservation actions (Dutton & Squires 2008). Unlike sharks, cetaceans and sirenians, the sea turtles, seabirds, and pinnipeds also face terrestrial-based anthropogenic mortality during the portion of their life cycles that take place on land. During terrestrial-based life cycle periods, eggs, hatchlings, and adults are susceptible to a variety of pressures, including invasive egg harvesting, invasive species predation, nesting habitat destruction, human predation, light pollution, and climate change-related impacts (Wallace et al. 2011; Kovacs et al. 2012; Brooke et al. 2018).

Conservation actions focused solely within individual nations (i.e., unilateral measures) are likely to fail the necessary level of conservation to achieve population recovery for many depleted populations of marine megafauna, which instead require cooperative multilateral conservation efforts and investment from many nations (Dutton & Squires 2008; Wallace et al. 2011). In the absence of a central authority, self-enforcing and voluntary agreements are often the only practical solution to address anthropogenic mortality at all life stages of marine megafauna species (Barrett 2003; Dutton & Squires 2008).

1.2.3 Multilateral treaties

The far-ranging nature of many marine megafauna species (e.g., sea turtles) means that they face transboundary impacts from fisheries in multiple nations' EEZs and on the high seas, which are not always fully regulated by international agreements (Dutton & Squires 2008). The 1982 United Nations Convention for the Law of the Sea (UNCLOS) prescribes nations' legal rights over the coastal strip of the ocean from their shores to 200 nautical miles at sea (United Nations 1982). UNCLOS also allows all States to exploit resources of the high seas. The exploitation of marine resources on the high seas is subject to adherence to the duty of conservation, which stipulates that “all States have the duty to take, or to cooperate with other States in taking, such measures for their respective nationals as may be necessary for the conservation of the living resources of the high seas”. Additional conservation measures for marine megafauna are specified in the 1995 United Nations Fish Stocks Agreement (United Nations 1995), which prescribes that States with “a real interest in the fisheries concerned” participate and cooperate in regional fisheries management organisations (RFMOs). RFMOs includes entities such as the Inter-American Convention for Tropical Tuna Commission and the Comisión Permanente del Pacífico Sur. Looking forward, the development of an internationally legally binding instrument for the conservation and sustainable use of marine

biodiversity in the high seas, legislated through UNCLOS, offers an avenue for further legal requirements to drive active participation in the conservation of transboundary marine species (United Nations 2015a). However, the degree to which participating nations implement management actions to mitigate the harmful impact from their fisheries on marine megafauna varies greatly, with many countries failing to implement sufficient management strategies, yet benefiting from the efforts of others, leading to free-riding (Dutton & Squires 2008). In economics, the failure to multilaterally cooperate is called a transnational externality, for which free-riding related issue can be further subdivided into market failure, overexploitation, biodiversity loss, and economic inefficiency (Barrett 2003; Squires & Garcia 2018b).

When a marine megafauna species enters into the EEZ of nations with little or no management measures in place to reduce incidental captures, mortality rates can be high, and these nations can act as a sink driving population decline (e.g., Alfaro-Shigueto et al. 2011; Alfaro-Shigueto et al. 2018). The recovery of marine megafauna populations may, therefore, require multilateral cooperation or coordination measures that help drive positive management actions in many jurisdictions using non-binding agreements, such as the Inter-American Convention for the Conservation and Protection of Sea Turtles (IAC). Lower-income nations may need support to implement these, as they may have no or limited financial resources to implement effective mitigation measures at sea or on land (Dutton & Squires 2008).

1.2.4 Measures to manage bycatch

Many management measures are available for mitigating harmful fishing impacts on marine megafauna species. Traditionally these measures are broadly categorised as output controls, such as catch limits (Gilman et al. 2012), or input controls, such as technical modifications to fishing gear, and restrictions on fishing locations and times using spatio-temporal area closures (Hall et al. 2000). Here I highlight if strategies are input and output while also categorising measures according to the steps of the mitigation hierarchy.

1.2.4.1 Avoidance

One input control is marine protected areas (MPAs) that stop or restrict fishing activity and that are called for by international agreements for biodiversity conservation (CBD [Convention on Biological Diversity] 2010; United Nations 2015b). The ecological benefits

of strategically placed and well-resourced MPAs are clear; they have been shown to increase species populations (Babcock et al. 2010), maintain cover and improve resilience of benthic organisms (Selig & Bruno 2010; Mellin et al. 2016), and generally result in higher biomass than unprotected areas (Lester et al. 2009; Edgar et al. 2014). When MPAs are implemented at a large- or dynamic-enough scale to take into account marine megafauna species' distributions, positive population recovery for these species can result (Hooker & Gerber 2004; Wilhelm et al. 2014).

In some cases, MPAs have also improved the economy of local communities by producing new profits based on ecosystem services such as tourism and fish production (Sala et al. 2016). In others, economic, social, political, and institutional contexts can result in trade-offs between desired ecological and social outcomes that yield positive outcomes for some conditions but adverse outcomes for others (Hargreaves-Allen et al. 2017). Critically, adequate investment in human and financial capacity is needed to ensure optimal conservation outcomes. For example, MPAs with sufficient staff and financial resources yield 2.9 times the ecological effect than MPAs with inadequate investment (Gill et al. 2017).

Prior experience with MPAs as a tool to support megafauna populations illustrates the challenges. For example, the USA National Marine Fisheries Service's establishment of an annual spatio-temporal closure between 15 August and 15 November in the operating ground of a large-scale commercial drift gillnet fishery. The fishery targets swordfish off the coast of Oregon and California. The spatio-temporal closure was put in place in 2001 because of concerns about bycatch of endangered leatherback turtles, and when active, closed around 90 per cent of the fishery's geographic extent (Janisse et al. 2010). Leatherback turtle captures in the area of the closure were significantly reduced following its implementation (Carretta et al. 2017; Eguchi et al. 2017). But difficulties arose because, despite the MPAs large size, the distribution of individual leatherback turtles is more extensive. Therefore turtles left the closure area, exposing them to capture in the area where vessels of the fleet were operating with intensified effort as a result of their displacement from the closure (Carretta et al. 2017). The loss in swordfish catch as a result of the closure also increased swordfish imports from overseas fisheries with fewer management restrictions in place to mitigate sea turtle bycatch (an outcome known as production, conservation, and trade leakages; Barrett 2003; Bruvold & Faehn 2006; Squires et al. 2010).

1.2.4.2 Minimisation

Output controls, such as restrictions on a total allowable catch (TAC) are primary management mechanisms in many fishery management frameworks around the world (Gordon 1954; Karagiannakos 1996). However, many small-scale fisheries located in lower-income countries have weak state-led management enforcing any effort limits on the catch (Allison & Ellis 2001). In such situations, fishers often oversee harvesting through locally-developed, or traditional self-governance approaches (Ostrom 2009; Gutiérrez et al. 2011). When thinking in terms of a biodiversity mitigation process, catch and effort restrictions primarily perform as a minimisation action to mitigate the negative fishing impact on marine megafauna as adverse fishing impacts are reduced but not completely avoided. However, thresholds of risk may be imposed, and if fishing impact is highly unlikely, an action such as night setting longline gear, may be considered an avoidance action for seabirds, rather than a minimisation measure (Booth et al. 2020).

Many management actions to reduce the capture of marine megafauna species in fisheries involve minimising the negative impact through input controls that involve a technical change at sea (Squires & Vestergaard 2013). Examples include implementing technologies such as light-emitting diodes (LEDs) on gillnets to deter sea turtles (Wang et al. 2013; Ortiz et al. 2016), acoustic alarms (that are also known as ‘pingers’) to deter small cetaceans (Carretta et al. 2008; Mangel et al. 2013), or bird scaring ‘tori’ lines to reduce seabird captures in longline fisheries (Løkkeborg 2003; Constable 2011). Along with technical innovation, social innovation supports measures to mitigate marine megafauna captures, including changes in fishing practices, such as the aforementioned night setting of longlines to reduce seabird captures (Clarke et al. 2014).

1.2.4.3 Remediation

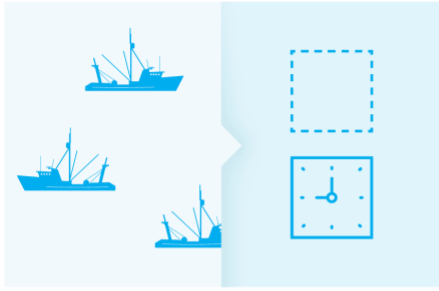
In the terrestrial literature, remediation actions in the context of EIA primarily refer to habitat restoration at the impact site, including attempts to bring impacted biodiversity as close as possible to its status before the impact in question, or, to another known and agreed pre-impact state (Maron et al. 2012). In fisheries, remedial actions primarily focus on stock restoration (i.e., reducing impact once a target stock falls below the Maximum Sustainable Yield; MSY). For marine megafauna, population recovery is a more applicable ‘restoration’ term. Conceptually, however, this can complicate the application of the mitigation hierarchy

framework as population recovery will often be specified as an overarching goal; thus, all actions across the hierarchy feed into its achievement. Restoration can also be considered at the vessel level. For example, improving post-capture survival rates of individual species by implementing best handling and release practices (e.g., for sea turtles; Epperly et al. 2004). Technological and social changes such as these can occur in response to regulations put in place by fisheries management (secondary effects). Changes can also occur due to increasing scarcity of a captured species, rising social valuations of the captured species of management concern, and consumer market effects (Squires & Vestergaard 2013).

1.2.4.4 Biodiversity offsets

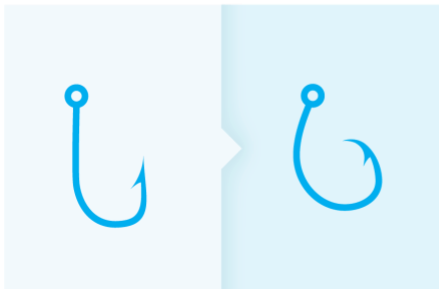
The discussion of biodiversity offsets in fisheries has occurred for over 15 years (Bellagio Conference on Sea Turtles 2004; Quigley & Harper 2006a; Quigley & Harper 2006b; Wilcox & Donlan 2007; Donlan & Wilcox 2008; Dutton & Squires 2008; Janisse et al. 2010; Dutton et al. 2011; Pascoe et al. 2011; Gjertsen et al. 2014; Gelcich & Donlan 2015; Milner-Gulland et al. 2018; Squires et al. 2018; Booth et al. 2020). In practice, few biodiversity offsets have been implemented in fisheries. Known examples include a partial compensation scheme (or ‘managed net loss’) implemented for Pacific turtle bycatch by the USA California drift gillnet fleet (Janisse et al. 2010). This sea turtle offset involves three USA tuna fishing companies and the International Seafood Sustainability Foundation (ISSF), who all assess the landings of companies processing longline-caught tuna to fund turtle nesting site protection, and turtle bycatch reduction in Mexican artisanal fisheries. In this case, the project-level management at net loss implies acceptance of loss in the fishery in the hope that the overall population impacts will be low enough to reduce recovery elsewhere. In Canada, fisheries managers rehabilitate salmon habitat using biodiversity offsets policy (Quigley & Harper 2006b; Quigley & Harper 2006a).

As in the terrestrial conservation literature, biodiversity offsetting in fisheries remains controversial. For example, the debate around the Wilcox & Donlan (2007) analysis of the potential for offsetting flesh-footed shearwaters (*Puffinus carneipes*) taken as bycatch in commercial longlines by undertaking invasive species eradication on Lord Howe Island where the shearwaters nest (Doak et al. 2007; Donlan & Wilcox 2007; Finkelstein et al. 2008; Priddel 2008; Igual et al. 2009; Wilcox & Donlan 2009; Žydelis et al. 2009). However, more recent assessments of the potential for biodiversity offsetting to fund invasive nonnative



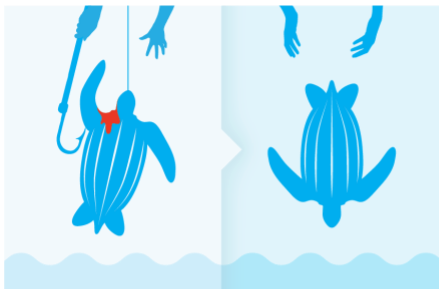
Step 1. **Avoid**

Avoidance of bycatch, partially or completely. Example: Area closures or time closures



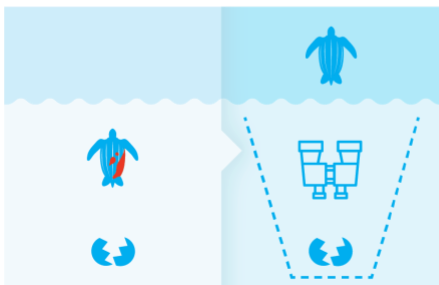
Step 2. **Minimise**

Technical / behavioural change (modifying gear, or how gear is used, to increase selectivity). Example: Gillnet fishing boats increase net visibility to turtles but not target species, using illumination.



Step 3. **Remediate**

Post-capture approaches that enable fishermen to put animals back alive. Example: Better handling and removal practices for leatherback turtles captured in gillnets.



Step 4. **Offset**

Example: Supplementation in leatherback turtle nesting areas through protection of eggs at beach nests and protection of newly hatched baby turtles making their way to the ocean

Figure 1.3. Examples of management measures for sea turtles taken as bycatch in a commercial trawl fleet. The figure depicts management measures for each step of the mitigation hierarchy.

species control and eradications on offshore islands highlight potential as a source of funding for biodiversity conservation (Norton & Warburton 2015; Holmes et al. 2016).

Another marine megafauna taxon proposed as a potential candidate for biodiversity offsets is sea turtles (Bellagio Conference on Sea Turtles 2004; Dutton & Squires 2008; Janisse et al. 2010; Gjertsen et al. 2014; Squires et al. 2018). Because sea turtles' nest on beaches, it has been proposed that terrestrial conservation action can be implemented as an offsetting action. Conservation actions could include protecting unprotected nesting sites from predators (including human hunters of turtle eggs and nesting adults, dogs, feral pigs, and various invasive species depending on the nesting site location), or climate change-related impacts such as rising sea level. For example, implementing artificial barriers to support nesting habitat to remain viable through nesting beach migrations, or relocating nesting sites that fall below an increasing high-water line to more stable beach areas or hatcheries (Dutton & Whitmore 1983). Despite the range of impacts sea turtles face throughout their life cycles, nesting site protection can be cost-effective (Gjertsen et al. 2014), and has been demonstrated to drive long-term increases in several sea turtle populations (Chaloupka 2003; Balazs & Chaloupka 2004; Dutton et al. 2005; Troëng & Rankin 2005). Used in conjunction with other management strategies within the formalised framework of the mitigation hierarchy (Figure 1.3), additional nesting site protection initiatives at currently unprotected nesting sites have the potential to improve sea turtle population recovery over and above what would be possible from at sea management strategies alone. Integrating biodiversity offsetting mechanisms into a holistic sea turtle recovery plan may help to provide clear mechanisms for funding and desired conservation actions for less cost (Milner-Gulland et al. 2018; Squires et al. 2018). Taking a broader approach to fisheries management offers a unique opportunity within fisheries management frameworks to integrate a proactive approach to conservation into a broader recovery strategy for these species, rather than focusing solely on reducing at sea mortality from fishery interactions (Dutton & Squires 2008).

For biodiversity gains to count towards an offset, there is a need to demonstrate “additionality” (McKenney & Kiesecker 2010). For example, considering a sea turtle biodiversity offset in the form of nesting site protection, only those emerged hatchlings and nesting adults that would have died from predators, high temperatures, pollution, and other site-specific threats had the offset not been implemented but were saved because of the additional protection the offset provides, could be counted towards the offset. Depending on the reference scenario set against which the net outcome goal is assessed (Figure 1.4), this

number would need to either match the level of residual loss from the fishery of management focus, measured against the current population growth rate of the sea turtle population affected by the fishery of focus (fixed reference scenario), or the projected population growth rate in the absence of sea turtle mortalities from the fishery in question, but continuation of other processes affecting vital rates like climate change or other unrelated conservation actions (dynamic reference scenario; Maron et al. 2018a). Critically, protection of an unprotected nesting site in question must not be planned or required in the absence of the offset under other conservation obligations (i.e., resulting in double counting; Maron 2015).

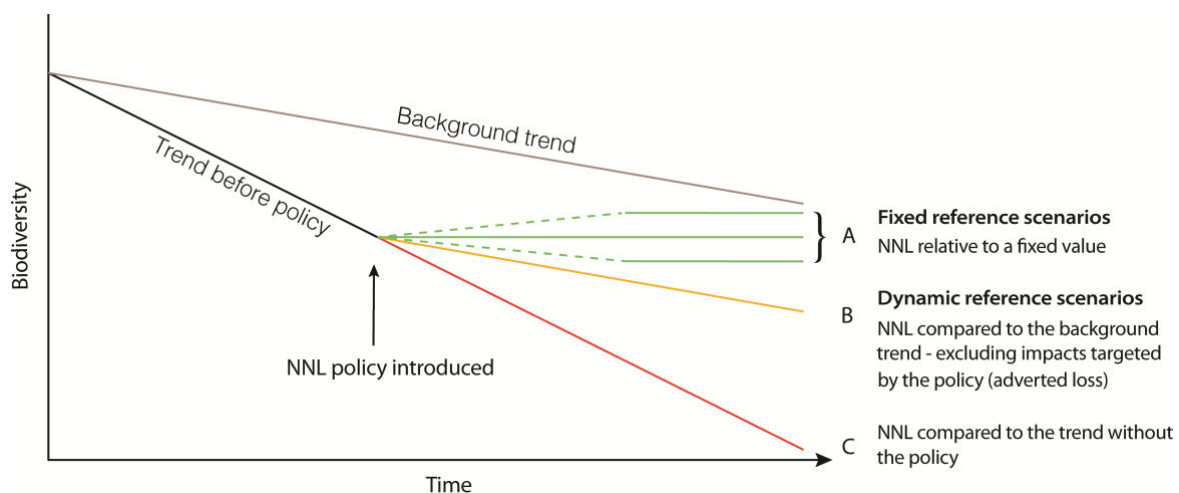


Figure 1.4. Examples of potential trends in biodiversity resulting from the implementation of no net loss (NNL) policies. The different types of reference scenarios shown include three fixed states (A) and two dynamic reference scenarios (B and C). The fixed state reference scenario can be considered a restoration offset. Note that B is parallel to the grey line that indicates the background trend – the expected change in biodiversity caused by various factors, including only impacts not targeted by the NNL policy (an adverted loss offset). The background trend is not necessarily one of decline. Assuming perfect implementation of the relevant NNL policy, the net outcome would match the reference scenario set for the policy. Redrawn from Maron et al. 2018a and narrowed from natural capital to biodiversity.

Proposals for biodiversity offsetting actions have not been restricted to terrestrial-based conservation measures focused on early life cycle stages for which large numbers of newly emerged hatchlings must be saved to equal the loss of an adult. A biodiversity offset can focus on mitigating adverse impacts to later life-stages of marine megafauna outside of the fishery. For example, the aforementioned voluntary tax in the USA California drift gillnet fleet is used to mitigate Pacific turtle bycatch in small-scale fisheries where captures are

known to be high; the trade-off is that these efforts are implemented in an effort to slow further area closures in the taxed fishery (Janisse et al. 2010). Like all conservation actions, such approaches require measured implementation and careful management to avoid perverse outcomes resulting in an exacerbation of biodiversity decline (Bull et al. 2013). However, foreseeable benefits include the potential to reduce the bycatch of adult marine megafauna species in sink areas whilst preventing inefficient and costly management measures restricting fishing effort where bycatch is of low-risk (Squires & Garcia 2018b). Cases, where such approaches are applicable, may or may not be widespread. Nevertheless, there is a need for further exploration as to how biodiversity offsets can be managed appropriately and used to offset residual impacts to sea turtles from fisheries following other management measures.

1.2.5 The importance of fishers and people

The importance of integrating a broad range of stakeholder perspectives, regardless of what other management measures are introduced, is vital for fisheries, and natural resource management more widely. Most conservation interventions require changes in human behaviour and the views of affected individuals should be understood and reflected in intervention design should management strategies be expected to succeed in the long term (Milner-Gulland 2012; St John et al. 2013; Cinner 2018). Despite this growing understanding as to the importance of understanding human behaviour, capturing stakeholder perspectives in decision making remains a major source of uncertainty for fisheries management (Fulton et al. 2011b).

Many processes and tools to better integrate social conditions into management decision making can be considered indirect incentive approaches in conservation, as they aim at improving conservation performance indirectly (e.g., development of local leadership and stewardship that reinforces sustainable fishing practices; Gutiérrez et al. 2011). Such approaches are particularly pertinent in coastal fishing communities located in countries with low social and economic indicators where top-down, centralised governance processes are often lacking (Berkes 2003; Karr et al. 2017). For example, participatory research has the potential to reduce fisher and community uncertainties around whether or not bycatch mitigation measures are reducing fish catches and having a beneficial impact on marine megafauna populations (Hall et al. 2007).

Implementing comprehensive population recovery initiatives for depleted marine megafauna populations can foster integrating partnerships between large-scale commercial fisheries, the broader environmental sector, and local resource harvesting communities. Collaborations can form by building upon community property and intrinsic motivation in the form of social norms, customary management, and co-management in local fishing areas and integrating those management approaches into a more comprehensive management framework (Ostrom 2009; Cinner et al. 2012a; McClanahan et al. 2015; Friedman et al. 2018).

Understanding and managing the uncertainty surrounding social elements of the system requires both inter-disciplinary approaches to research (i.e., integrating knowledge and methods from different disciplines, using a synthesis of approaches) and trans-disciplinary approaches to research (i.e., creating an intellectual framework beyond the disciplinary perspectives, that fully integrates stakeholders; Alexander et al. 2018a). An excellent example of this is when implementing newly established conservation initiatives, where the long-term outcomes can strongly depend on the establishment of supportive social norms in the early stages of the intervention's implementation (Walmsley & White 2003). In such cases, modelling information-sharing in a social network, for example, can help researchers, practitioners, and managers better understand heterogeneity in fisher behaviour, which, in turn, can lead to well-designed conservation interventions that target key individuals within a community who support knowledge transfer and behavioural shifts relating to the intervention in question (Alexander et al. 2018b; Barnes et al. 2019). Critically, throughout all collaborative management exercises, consideration must be given to respecting fishing as a livelihood, beginning a stakeholder dialogue early on in the management process and continuing this throughout the implementation of the intervention in questions, and understanding risks fishers take by working on conservation interventions that seek to reduce marine megafauna captures (Hall et al. 2007). Engagement with the fisher social networks is critical for researchers and policymakers to understand the local situation in which management actions are taking place and having an awareness of the broader social and political context in which the fishery in question sits (Hall et al. 2007; Campbell & Cornwell 2008).

1.3 Aims and objectives

The aims of the research presented in this thesis are twofold. The first aim is to develop and improve the theoretical basis that reconciles commercial production of renewable natural resources with biodiversity conservation through case studies focused on commercial fisheries and more widely across the renewable resource sectors. The second aim is to help develop broader systems thinking in fisheries management emphasizing anticipation and prevention of adverse human impacts across the life cycles of incidentally captured marine megafauna species.

In particular, this research focuses on managing the impact of small-scale fisheries in lower-income nations on marine megafauna, where the trade-offs between biological, economic, and social conditions pose a considerable challenge to management. I explore the integration of existing fisheries management tools and processes into an overarching framework that supports the assessment of management strategies towards biodiversity goals standardised across fisheries and scales. I primarily frame this investigation using the case study of turtle captures in a coastal fishing community in Peru. This case study is framed within the development and application of an overarching framework to account for all human impact on biodiversity.

The main research objectives are:

1. to explore the use of an approach for the mitigation of impacts on marine biodiversity (widely used in terrestrial EIAs) as an overarching framework for integrating the multiple elements of conservation goals and interventions towards a common goal;
2. to explore specific challenges and trade-offs in implementing a holistic management strategy, based on the mitigation hierarchy framework, for sea turtles in a data-poor, small-scale fishing system, particularly obtaining the necessary data to understand the bycatch issue and implement appropriate management measures;
3. to consider novel approaches to collecting and analysing information in data-poor fisheries scenarios to support a holistic recovery framework for sea turtles;
4. to investigate innovative approaches to understanding social dimensions of conservation problems by mapping social network structure to support conservation intervention expansion and uptake.

1.4 Thesis outline

This thesis has four main sections, as well as an Appendix containing additional relevant research outputs. These comprise i) this introduction, ii) a section exploring the applicability of an overarching framework for mitigating adverse impacts to biodiversity, first using a global case study for all human impact to biodiversity, and then focusing on the management of turtle captures in a small-scale commercial fishery system in Peru, iii) a section gathering baseline data from the Peru case study system – including estimating turtle captures across two gillnet fleets operating from the community of focus and mapping the social network of the case study fishery to improve understanding of fisher perspectives towards turtle bycatch, and iv) a general discussion and conclusion. The appendices comprise additional research undertaken throughout my DPhil, which both contributes to this thesis's primary research narrative or was influenced by it.

The chapter structure of this thesis is as follows:

Chapter 2 is a conceptual analysis, which outlines the potential utility of applying an overarching conservation framework based on the mitigation hierarchy to all anthropogenic impacts on biodiversity (including fisheries and other renewable resource sectors). This research makes several contributions to the conservation science research field. Firstly, it is the first study to conceptually apply a single overarching framework to all human impact on biodiversity. In doing so, it highlights the significant research gap in processes to align the multitude of conservation interventions across sectors and scales towards global biodiversity conservation targets. Secondly, it highlights the mitigation hierarchy's requirement for transparency in setting goals, baselines, metrics and actions. These characteristics could provoke positive change in global biodiversity loss management, by helping to prioritise consideration of conservation goals and drive the empirical evaluation of conservation investments through the explicit consideration of counterfactual trends and ecosystem dynamics across scales. This chapter has been published in *BioScience* and was awarded Editor's pick for April 2018:

Arlidge WNS, Bull JW, Addison PFE, Burgass MJ, Gianuca D, Gorham TM, Jacob C, Shumway N, Sinclair, SP, Watson JEM, Wilcox C, Milner-Gulland EJ (2018). A Global Mitigation Hierarchy for Nature Conservation. *BioScience* **68**: 336–347.

WNSA and EJM-G conceived and developed the research idea. WNSA generated the tables and initial figures and wrote the manuscript with support from EJM-G. All the co-authors provided extensive review and commentary.

Chapter 3 presents an application of the mitigation hierarchy framework for fisheries management and bycatch mitigation to a case study fishing system in San Jose, Lambayeque, Peru. This research provides an overview of the case study, assesses the conservation issue requiring management (turtle captures), and investigates how the conceptual mitigation hierarchy approach can be applied to evaluate potential management actions that can support a reduction in turtle captures in the fishery. This research makes three contributions to the conservation science research field. Firstly, it is the first study to apply the mitigation hierarchy framework for fisheries management and bycatch mitigation using data from a real-world fishery. Secondly, this research develops an understanding of how the established decision-making processes in fisheries of ecological risk assessment, and management strategy evaluation can integrate into a broader mitigation hierarchy framework. Thirdly, it contributes to research developing processes to mainstream biodiversity into fisheries management, with a focus on clearly aligning high-level goals for biodiversity to a local-level fishery in a lower-income country setting. This chapter has been published as:

Arlidge WNS, Squires D, Alfaro-Shigueto J, Booth H, Mangel J, Milner-Gulland EJ (2020). A mitigation hierarchy approach to managing sea turtle captures in small-scale fisheries. *Frontiers in Marine Science*, 7: 49.

WNSA, EJM-G and DS conceived this study. JAS and JCM supported the data cleaning of the observer database. WNSA wrote the R code to clean the San Jose observer data, explore the turtle capture rates, and assess fisher perspectives of potential management strategies; and undertook the literature review. WNSA implemented the qualitative ecological risk assessment, led the preliminary qualitative management strategy evaluation, and wrote the first draft. JAS, JCM, and HB contributed to the development of the proposed framework's application in the case study fishery. All authors provided comments on the manuscript.

Chapter 4 uses green turtle, and leatherback turtle capture rates, calculated from a fisheries observer programme operating in the case study system as a baseline to compare with collected bycatch data from expert judgements elicited using the IDEA (“Investigate,” “Discuss,” “Estimate” and “Aggregate”) structured elicitation protocol. This research makes

four contributions to the research field. First, it collects baseline turtle bycatch data in the San Jose gillnet fishery. Second, it provides a new context for using the IDEA protocol, which before this study, had never been applied for eliciting estimates of captures of endangered, threatened, and protected species in small-scale fishing systems. Third, few small-scale fishery systems have observer data for bycatch rates. Therefore, this study provides a valuable comparative analysis of observer and expert knowledge data sources. Fourth, this study demonstrates the use of structured elicitation methods when involving experts inexperienced in providing quantitative estimates (here, the gillnet skippers), as well as illustrating the validation of expert judgment when the ‘truth’ from observational data is also uncertain. I conclude that the IDEA protocol may prove helpful for rapid, exploratory evaluations of bycatch impact in data-poor small-scale fishery management scenarios. A version of this chapter has been accepted for publication:

Arlidge WNS, Squires D, Alfaro-Shigueto J, Ibañez-Erquiaga B, Mangel J, Milner-Gulland EJ. (2020). Evaluating Elicited Judgements of Turtle Captures in Small-scale Fisheries. *Conservation Science and Practice*, e181.

WNSA conceived this study. WNSA and EJM-G developed the methodology. WNSA and BI-E piloted the study. Data collection was carried out by WNSA and BI-E. JAS and JCM, supported the data cleaning of the observer database. Statistical analysis was completed by WNSA with support from DS and EJM-G. WNSA wrote the manuscript. All authors provided review and commentary.

Chapter 5 uses network null models to compare networks mapping information sharing between gillnet skippers in the San Jose case study fishery to better understand fisher perspectives towards turtle bycatch, and to inform the expansion of a turtle bycatch reduction strategy throughout the wider gillnet fisher community in future. Using social network analysis, I show that information-sharing links are generally consistent across contexts, but that networks of information sharing regarding turtle bycatch reduction measures are relatively disconnected. I use network permutation methods to show this is driven by a combination of a general lack of information sharing within this context, as well as the fact that information sharing is organised in a way that promotes disconnection. This research makes three contributions to the research field. First, it collects social data on turtle bycatch data in the San Jose gillnet fishing system. Second, it is the first application of null model network analysis to the fields of conservation science and natural resource management, and

one of the very few applications to human social networks. Third, it illustrates how network analysis techniques may offer in-depth insights into the fine-scale structure of human social systems of conservation interest, beyond what could be gained through the network methods which are usually employed, such as counting the number of associates an individual in the community has. This chapter will be submitted for publication to *PNAS* as:

Arlidge WNS, Firth JA, Alfaro-Shigueto J, Ibañez-Erquiaga B, Mangel JC, Squires D, Milner-Gulland EJ. Understanding the potential for information spread about fisheries bycatch reduction initiatives using cross-contextual information-sharing networks.

WNSA and EJM-G conceived the study. WNSA, EJM-G, BI-E, JAS, and JCM contributed to the survey design. Data gathered by WNSA and BI-E, with additional support from Natalie Bravo. JAF and WNSA carried out the analysis in R. WNSA, JAF, and EJM-G interpreted the data and planned the first draft. WNSA wrote the first draft. All authors contributed to revising the manuscript.

Chapter 6 presents a general discussion and conclusions. The chapter provides a summary and synthesis of key findings across the research presented in this thesis, highlighting contrasts and common themes between the central data chapters. Implications for future conservation and management are discussed and the constraints of the research considered. Suggestions for future research are provided throughout.

1.5 Additional research

I am a co-author on the following research. I contributed to this research during the course of my DPhil. This research adds to this thesis's research narrative, but was not the majority of my own work:

Milner-Gulland, E.J., Garcia, S.M., **Arlidge, W.N.S.**, Bull, J.W., Charles, T., Dagorn, L., Fordham, S., Hall, M., Schrader, J., Vestergaard, N., and Squires, D. (2018). Translating the terrestrial mitigation hierarchy to marine megafauna bycatch. *Fish and Fisheries* **19**: 547-561.

Bull, J.W., Milner-Gulland, E., Addison, P.F., **Arlidge, W.N.S.**, Baker, J., Brooks, T.M., Burgass, M.J., Hinsley, A., Maron, M., Robinson, J.G., Sekhran, N., Sinclair, S., Stuart, S., Zu Ermgassen, S.O.S.E., and Watson, J.E.M. (2020). Net positive outcomes for nature. *Nature ecology & evolution*, **4**: 4-7.

Davis, K.J., Alfaro-Shigueto, J., **Arlidge, W.N.S.**, Burton, M., Gelcich, S., Mills, M., Milner-Gulland, E.J., Mangel, J., Palma Duque, J., Romero, C. Disconnects in global discourses – the unintended consequences of marine mammal protection on small-scale fishers. bioRxiv 2020.01.01.892422.

Chapter 2

A Global Mitigation Hierarchy for Nature Conservation

Published as:

Arlidge, W.N.S., Bull, J.W., Addison, P.F.E., Burgass, M.J., Gianuca, D., Gorham, T.M., Jacob, C., Shumway, N., P., S.S., Watson, J.E.M., Wilcox, C., and Milner-Gulland, E.J. (2018). A Global Mitigation Hierarchy for Nature Conservation. *BioScience* 68, 336–347.

2.1 Introduction

Humans' growing demand for resources is resulting in the rapid erosion of natural habitats (Watson et al. 2016b). This is leading to an irreplaceable loss of biodiversity (Hoffmann et al. 2010) that can compromise the healthy functioning of ecosystems (Hooper et al. 2012). The primary causes of biodiversity loss include overexploitation of species, habitat modification, invasive alien species and disease, pollution, and climate change (Maxwell et al. 2016). Yet while we have an increasing understanding of the causes of biodiversity loss, the main drivers can be obscured, in part, because existing frameworks for conservation planning, implementation and evaluation do not consider conservation efforts to tackle drivers of biodiversity loss as a cohesive whole. The current patchwork of international goals and targets (e.g. the United Nations Convention on Biological Diversity [CBD] Aichi Targets and Sustainable Development Goals), national plans, and local interventions can result in the gaps and weaknesses of conservation efforts that are difficult to identify or articulate (Rands et al. 2010). For example, the global Protected Area (PA) network now covers 14.8% of all terrestrial surfaces and 5.1% of the global ocean (UNEP-WCMC and IUCN 2016), yet many of these PAs occur in residual areas, avoiding locations with high value for natural resource extraction (Devillers et al. 2014; Venter et al. 2017). The result is a significant shortfall in the

protection of nature across ecoregions, and important sites for biodiversity remaining unprotected (Butchart et al. 2015; Dinerstein et al. 2017).

Biodiversity loss, much like climate change, is an environmental crisis that requires a coordinated international effort if it is to be managed effectively. The 2015 Paris climate agreement specifies a clear goal to limit global warming by 2 °C above pre-industrial levels (UNFCCC 2015), and the recent publication of a roadmap for rapid decarbonisation offers guidance on actions required at the national level to effectively limit carbon emissions to meet the goal (Rockström et al. 2017). A call has recently been made for a similar roadmap for global biodiversity conservation, to guide the necessary steps to achieve goals and targets for stopping the biodiversity crisis (Watson & Venter 2017). This requires an integrated global framework, capable of implementation at national and project levels, which would enable the quantification and subsequent reduction of humanity's impact on biodiversity. To date, no-one has tried to conceptualise all human biodiversity impacts and conservation efforts within such a framework. The benefits of such an approach would be to unite all aspects of conservation under a standardised paradigm with a broad biodiversity conservation goal, supporting multiscale, evidence-based decision making. Exploring the potential benefits of such a framework is particularly timely, given the CBD biodiversity strategy will be renegotiated in 2020 (CBD [Convention on Biological Diversity] 2010).

Industrial sectors such as mining, energy, and manufacturing are increasingly using a framework known as the mitigation hierarchy to guide their activities towards limiting the negative impact on biodiversity (BBOP [Business and Biodiversity Offset Programme] 2012; IFC 2012). A goal either of No Net Loss (NNL), or Net Gain, of biodiversity is typically set (also referred to as net neutral and net positive goals, respectively), relative to a predetermined baseline (BBOP [Business and Biodiversity Offset Programme] 2012; Maron et al. 2018a). The process is implemented through national planning process and negotiations between government agencies, conservation actors and developers, with elements of the process often formalised within an Environmental and Social Impact Assessment (ESIA). The mitigation hierarchy is comprised of four broad action steps: 1) avoid, 2) minimise, 3) remediate, and 4) offset (Figure 2.1). The first step involves avoiding impacts to biodiversity, for example, screening potential risks prior to project design and selecting an alternate development site (Phalan et al. 2018). The second step of the hierarchy requires that before and during development, impacts are minimised, such as by using more environmentally friendly construction methods. The third step requires that biodiversity loss is then remediated

within the footprint of the development, which could entail actions like reseeded impacted land, or developing a breeding programme for impacted species, during and after project completion. The fourth and final step requires that any residual impacts not captured by the first three steps of the hierarchy are offset elsewhere, such as through wetland restoration, or removal of invasive species from ecologically important areas (Gardner et al. 2013). The four steps of the mitigation hierarchy represent broad categories of biodiversity impact reduction and compensation, meaning most conservation actions can be categorised within these steps (Table 2.1).

As it stands, the mitigation hierarchy offers transparency between stakeholders with the flexibility to address a variety of anthropogenic impacts on biodiversity, across different sectors and scales. Many regulatory and financial instruments are now in place which aim to balance biodiversity conservation with (sustainable) economic development, by requiring the application of the mitigation hierarchy. For example, 69 countries have NNL policies in place or under development (Maron et al. 2016). Yet, taken overall, these commitments operate in a system that has allowed the significant loss of biodiversity, even when development was legally compliant (BBOP [Business and Biodiversity Offset Programme] 2012; Watson et al. 2016b).

The mitigation hierarchy is not widely applied to the most prevalent impacts on biodiversity that result from the direct removal of biological materials in sectors such as agriculture, fisheries, forestry, and wildlife trade (Rainey et al. 2015; Maxwell et al. 2016). Various frameworks exist to manage the impacts that result from extracting biological resources and promote sustainable use (e.g., forest certification schemes (Lattimore et al. 2013); ecosystem-based fisheries management (Pitcher et al. 2009); and agri-environment schemes (Pretty 2008)). Yet these frameworks often fail to account for all the negative biodiversity impacts caused by extracting target resources. For example, in forestry, road building to access previously inaccessible trees opens up remote wilderness areas to the secondary pressures of hunting, human colonisation, invasive species and fire (Bennett 2004). Major certification schemes such as the Forest Stewardship Council have also been criticised for failing to explicitly account for incidental biodiversity impacts, such as bushmeat harvesting (FSC 2015). Applying a standardised framework such as the mitigation hierarchy to all human impact would allow for seemingly disparate impacts to biodiversity to be categorised and accounted for between sectors, scale and nations. For example, the direct and immediate biodiversity impact of clearing species rich forest for an oil palm plantation, the

longer term and potentially more diffuse indirect biodiversity impacts that result from new forestry infrastructure (e.g., illegal hunting and informal clearance for settlement), and transboundary effects of air pollution from clearance fires could be accounted for within the same framework, while apparently disparate mitigation efforts could be linked (Figure 2.1).

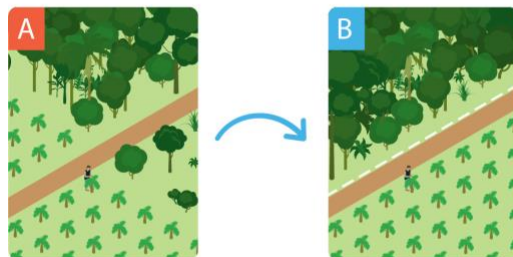
Table 2.1. Examples of biodiversity conservation tools and actions categorised into each of the four steps of the mitigation hierarchy.

Mitigation hierarchy: Examples of existing conservation tools and approaches	
Avoid	Protected areas†; Alliance for Zero Extinction Sites; Key Biodiversity Areas; no development in Vulnerable Marine Ecosystems (FAO vulnerable ecosystems) or critical habitat (International Finance Corporation PS6+); no damage to any listed threatened species or ecosystems (IUCN Red List of threatened species and ecosystems, and national conservation list species); no damage to intact habitat, UNESCO World Heritage Sites, Wilderness Areas
Minimise	Sustainable use; agri-environment schemes; shift from passive non-selective gear to actively targeted gear in fisheries; multi-use protected areas; payment for ecosystem services; demand reduction; certification and eco-labelling; economic incentives (market prices, taxes, subsidies and other signals); green infrastructure; corporate environmental strategies and operations; maintenance of ecosystem resilience.
Remediate	Rewilding †; restoration†; natural flooding of wetlands†; artificial habitat creation†; de-extinction.
Offset	Degraded ecosystem restoration away from impact site†; averted risk; reseedling/respawning†; captive breeding; invasive removal; species creation.

† Conservation tool or action that can shift between steps of the mitigation hierarchy depending on whether the biodiversity baseline is set at a present-day or historical point in time. Also depending on what national and regional legislation is in place to enforce action taken.

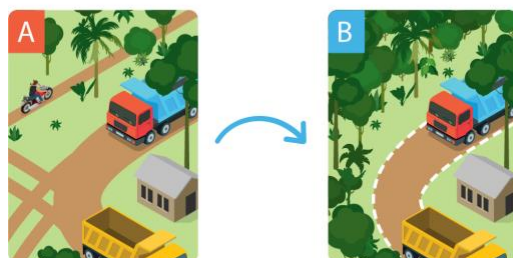


Left: Pre-plantation, or original state of the area prior to palm oil plantation.



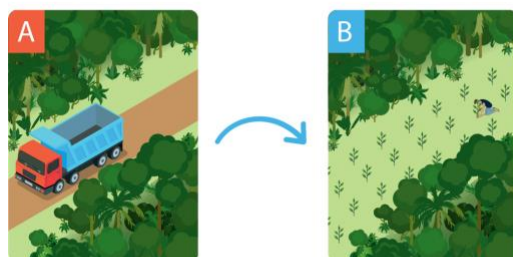
Step 1. Avoid

Avoid deforesting primary growth rainforest, or forest areas containing high levels of biodiversity or protected species. Example: Protected Area closure or new site selection following stakeholder consultation.



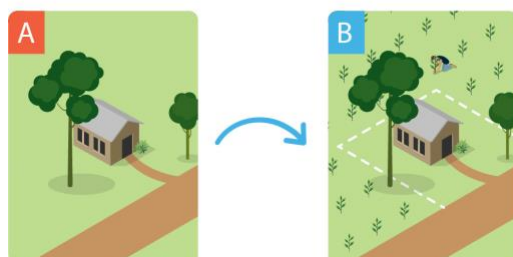
Step 2. Minimise

Minimise harm to biodiversity by adhering to best practice growing and extraction practices. Example: limiting the footprint of heavy machinery used to extract and transport palm oil to specific areas and ensure any runoff is contained to prevent polluting watercourses.



Step 3. Remediate

Remediate the biodiversity loss within the oil palm site. Example: replanting cleared areas of forest following road infrastructure development.



Step 4. Offset

Residual additional damage caused by the oil palm development through improvement of rainforest elsewhere. Example: Local areas with degraded rain forest is replanted near the development site.

Figure 2.1. An example of the mitigation hierarchy applied to the oil palm industry to achieve no net loss of biodiversity for the negative impact to biodiversity (deforesting rainforest) as a result of planting oil palm monocultures, in this case, African oil palm (*Elaeis guineensis*). Images marked with an 'A' represent the types of negative impacts from planting oil palm monocultures, and the corresponding images marked 'B' represent ways to address these impacts by undertaking the four steps of the mitigation hierarchy. Steps 1 to 3 occur at the site of negative impact on biodiversity, whereas step 4 occurs away from the impact site, addressing residual adverse impacts.

2.2 Critical elements of the mitigation hierarchy approach

Developers adhering to the mitigation hierarchy are first required to set a biodiversity goal (BBOP [Business and Biodiversity Offset Programme] 2012). This typically takes the form of NNL or Net Gain of biodiversity, though a goal such as improving trends in biodiversity could also be used (e.g., as in national species recovery plans). Next, quantitative targets and associated biodiversity metrics or indicators must be defined to measure the achievement of the goal (BBOP [Business and Biodiversity Offset Programme] 2012; Butchart et al. 2015). Undertaking this process means that assumptions surrounding what achieving the biodiversity goal would look like, and the calculations required to verify it, are made explicit.

The consideration of counterfactual scenarios (i.e., what would have happened in the absence of development and its associated mitigation measure(s)) is key to evaluating whether the biodiversity goal has been met (Bull et al. 2014; Maron et al. 2016; Table 2.2). The practice of empirically evaluating whether a specific intervention works better than alternate interventions or no action at all remains woefully lacking in conservation science (Ferraro & Pattanayak 2006), and a major benefit of the mitigation hierarchy is that it requires this critical thinking. This process requires the involvement of all stakeholders - regulators, industry, conservationists. The wider use of the mitigation hierarchy would, therefore, precipitate a shift towards the routine empirical evaluation of biodiversity conservation investments.

Arguably the most important step of the mitigation hierarchy is its first step, impact avoidance. This requires developers to predict and prevent negative impacts to biodiversity prior to any development actions taking place (BBOP [Business and Biodiversity Offset Programme] 2012). The conservation benefits of avoiding impacts are likely to outweigh, taking more uncertain remediation and offsetting measures once the damage has occurred (Watson et al. 2016b; Lindenmayer et al. 2017). Actions that drive adherence to the first step of the mitigation hierarchy include; following environmental regulations designed to protect biodiversity (e.g., through national planning process and negotiations between stakeholders), giving clear guidance on critical biodiversity areas (e.g., Key Biodiversity Areas), and making political decisions to set aside areas of high societal value (e.g., World Heritage Sites). Failure to comply with the avoidance step of the mitigation hierarchy may eventuate from a lack of political or regulatory enforcement, poor process, or lack of capacity and technical knowledge of regulators, developers and consultants (Phalan et al. 2018).

The "minimisation" step is central to current project-level conservation activities, including sustainable use, agri-environment schemes, alternative livelihoods and payments for ecosystem services. At the national level, many states have adapted ESIA legislation and guidance, which feeds down into the incorporation of biodiversity concerns into economic activities at the project level (Bull et al. 2017). Whereas rewilding, restoration projects and the natural flooding of wetlands align with 'remediation' measures for impacted biodiversity. Remediation equally applies to reestablish depleted resource stocks (Table 2.1).

Many of the key issues regarding quantifying and compensating biodiversity, which emerge when the mitigation hierarchy is applied, have parallels with the wider challenge of defining and measuring sustainability (e.g., Heal's 2012 review on managing natural capital and the interactions between humanity's economic activity and the environment). The most controversial element of the mitigation hierarchy is its last step, offsetting because it is here that these challenges come into sharp relief; they can be sidestepped to some extent in the first three steps of the hierarchy. Offsetting happens when significant residual impacts from a development remain after application of the first three levels of the mitigation hierarchy (BBOP, 2012). It is controversial because it requires the acceptance of a development that harms biodiversity, on the assumption that this harm can be accurately quantified and balanced by benefits elsewhere (Maron et al. 2016).

The theoretical and practical challenges of achieving NNL of biodiversity from development are increasingly well described and are widely reported (Table 2.2). For example, a nest box program in Australia intended to offset the clearing of hollow-bearing trees did not achieve the intended biodiversity outcomes for three threatened vertebrates reliant on the trees due to: 1) a failure to consider equivalency (the nest boxes failed to provide habitat for the target species), 2) incorrect use of multipliers (the 1:1 offset ratio did not account for the risk of offset failure), and 3) a lack of compliance and monitoring to evaluate the true effectiveness (Lindenmayer et al. 2017). As illustrated here, many of the issues with offsets result from poor operationalisation, monitoring and compliance, rather than inherent to the concept itself (Quétier et al. 2014).

Table 2.2. Approaches to addressing theoretical and practical challenges of applying the mitigation hierarchy, with particular focus on the offsetting step, based on practical experience to date (as articulated in e.g., Bull et al. 2013, BBOP 2012).

Challenge	Description	Current project-level best practice recommendations	Conceptual examples of global-level best practice
Additionality	Biodiversity benefits.	Only biodiversity benefits that are additional to a baseline scenario count as valid offsets.	Nations required to account for offset-funded biodiversity protection (alongside associated biodiversity losses that triggered offset) separately from biodiversity protection going towards existing global conservation commitments (e.g., CBD Aichi Target 11; Maron et al. 2015b).
Compliance and monitoring	Non-compliance with mitigation hierarchy; insufficient compensation resulting in lack of incentive; legislative changes during development.	Ensure relevant authorities follow up with monitoring to ensure compliance.	No net loss impact to biodiversity targets are made legally binding where possible (e.g. for all United Nations fisheries through UNCLOS, requiring stipulation of defined baselines, indicators, and best practice implementation); Global-level monitoring and evaluation programme created; Requirements for national-level reporting to an international body (e.g., United Nations CBD).
Biodiversity indicators	Unitary measures of biodiversity lost, gained or exchanged.	Use multiple or compound indicators; incorporate a measure of ecological function as well as biodiversity.	Use established mechanisms to develop and test indicators (e.g., the Biodiversity Indicators Partnership which evaluates the CBD Aichi targets & biodiversity SDGs): https://www.bipindicators.net/
Equivalency	Demonstrating equivalence between biodiversity losses and gains.	Encourage ‘in kind’ or like-for-like trades and prevent ‘out of kind’ trading unless ‘trading up’ from losses that have little or no conservation value; Ensure there are requirements for spatial constraints within which biodiversity offsets will and will not be considered.	An international governing body such as the United Nations stipulates that biodiversity offsets are restricted to ‘in kind’ trades implementable within a predetermined radius of the impact site, based on ecologically meaningful scales for the biodiversity concerned.
Least-cost	Guiding actions economically by costs so that efficiency dictates that each hierarchical step be undertaken to the	Ensure offset cost is set at a sufficient level to incentivise adherence to avoidance and minimisation steps higher up in the mitigation hierarchy.	Evidence that alternate scenarios representing actions higher up the mitigation hierarchy have been investigated and their ruling out is justified prior to any offsets commencing. Require this to be recorded in Environmental Impact Assessments and submitted by all signatory

	point where marginal costs are equalised.		nations to the international governing body. Free public access to reports is granted.
Longevity	The length an offset scheme should endure.	Offsets should last the length of the negative impacts at a minimum; offsets should be adaptively managed in the light of ongoing external change.	Nations required to adopt the stipulated time period for agreed global biodiversity goals, and in addition to enforcing regulation that ensures the longevity of biodiversity offsets. Failure to successfully manage offsets for their necessary lifetime would result in censure.
Multipliers	A factor that increases the amount of biodiversity gains required by an offset	Calculation of multiplier is based on various factors (e.g., the discount rate for future biodiversity gains, and uncertainty in definition and measurement of biodiversity).	Legal requirements are put in place to ensure that appropriate biodiversity offset calculators are used for all offset projects, ensuring a minimum biodiversity offset multiplier accounts for time discounting, additionality and permanence of project (e.g., Laitila et al. 2014)
Reversibility	Defining a development's reversibility.	Ensure all biodiversity losses are reversible otherwise categorise the affected biodiversity as a 'no go'.	Nations' goals for preventing species extinction and ecosystem collapse would be required to map on to international goals, with international reporting requirements concerning compliance and monitoring.
Substitutability	The degree to which the 'value' of a certain biodiversity type influences demand for one or more other biodiversity types.	The value of biodiversity types must be based on national legislation and societal value.	Clarify and justify when is one ecosystem, species or population is seen as equivalent to another and therefore tradable.
Thresholds	Areas or components of biodiversity which should not be compensated for because they are too important.	Define explicit thresholds for biodiversity losses and gains that cannot be offset.	Internationally recognised 'no go' zones for biodiversity offsets such as the Protected Area network, Key Biodiversity Areas, crisis ecoregions, and the Wildlife Conservation Society's Last of the Wild places; Consideration is also given to aspects of human development which should not be traded off due to their contribution to the future of humanity, for example adequate safe water for all.
Time lag	Deciding whether to allow a temporal gap between development and offset gains.	Incorporating a pre-offset step in the form of mitigation banking.	A pre-impact conservation gain requirement could be built in to international funding for economic development.

2.3 Expanding the mitigation hierarchy to encompass all human impact to biodiversity

The direct extraction of biological resources is the dominant driver of current species loss (Maxwell et al. 2016), yet the practical application of the mitigation hierarchy to the biological resource use sectors has received little attention (but see Aiama et al. 2015). In fisheries management, all four steps of the mitigation hierarchy are discussed (Wilcox & Donlan 2009; Gjertsen et al. 2014), but they have yet to be formalised into a conservation framework to manage fishing impacts. Using a mitigation hierarchy, NNL of biodiversity (or similar goal such as population recovery) could be extended to managing the incidental impacts to biodiversity caused by extracting target resources (e.g., fisheries bycatch management; Milner-Gulland et al. 2018; Table 2.3). A NNL goal could then be incorporated into international natural resource management agreements such as the UNCLOS conservation and sustainable use of marine biological diversity instrument (United Nations 2015a).

Particularly crucial to an extension of the mitigation hierarchy to global conservation is the consideration of the scale at which goals and targets are evaluated (Table 2.3). Although achieving NNL of biodiversity is often a goal for individual projects, some have suggested that net human impact on biodiversity should be evaluated at the landscape or national scales, considering the aggregate impact of individual developments and their associated mitigation programs (Kiesecker et al. 2009; Bull et al. 2014). Bull and Maron (2016) also consider the global conceptual application of the NNL principle to changes in species richness worldwide. A strategic approach to NNL could evaluate biodiversity gain and loss at ecologically and institutionally meaningful scales (ranging from local to global), enabling conservation efforts, of different types and at a range of scales, to be integrated and categorised within the hierarchy's four steps: avoid, minimise, remediate, offset. A multiscale approach to NNL, not just a project-level one, would mean that wider goals are not contradicted by piecemeal approaches to NNL at the project level (Maron et al. 2018a). Table 2.3 shows how the application of the mitigation hierarchy would change depending on the scale under consideration. By considering local, regional and national actions under the same framework, we could begin to piece together a global picture of action towards an overarching net goal, offering a coherent framing for conservation efforts.

2.4 Key factors for successful application of the global mitigation hierarchy

2.4.1 Goals, targets and indicators

At each scale of application (global, regional, local), there would be a need to set NNL goals (or similar goals that account for losses and gains) that focus on particular facets of biodiversity (Figure 2.2). These could include elements of biodiversity embodied in the Essential Biological Variables (Pereira et al. 2013; Gonçalves et al. 2015), which are recommended to guide the setting of biodiversity goals and indicators in policy-making (Pereira et al. 2013). Ideally, these goals would be set to reflect existing aspirations for sustainable development (e.g., the SDGs; United Nations 2015b), international conservation (e.g., CBD Aichi targets; United Nations 1992), and national legislation relating to environmental protection.

To successfully achieve biodiversity goals, there is a need to set targets that specify a quantitative amount of change required for success. SMART (specific, measurable, ambitious, realistic, time bound) targets are preferred; a hypothetical example of a SMART target would be: all United Nations countries' fishing fleets will achieve a NNL impact on biodiversity by 2050, set against the frame of reference of the FAO 1955 global fish stock assessments and benthic biodiversity assessments from the IUCN (Table 2.2). Currently, many targets suffer from ambiguity, complexity, and redundancy; lessons need to be learnt from failings with the CBD Aichi Targets, more than two-thirds of which were found to lack a quantifiable component (Butchart et al. 2016).

Next, relevant biodiversity indicators can be developed to measure the desired change in biodiversity, to achieve specific goals and targets at varying scales (e.g., those developed by the Biodiversity Indicators Partnership; Butchart et al. 2007; Table 2.2). Biodiversity indicators need to be context-dependent, with best practice suggesting they should be: 1) sensitive to and respond predictably to human impact, 2) feasible to monitor, 3) informative at different spatial and temporal resolutions, and 4) practical in terms of monitoring costs and data availability (Jones et al. 2011; Gonçalves et al. 2015). There are added levels of complexity surrounding indicator development that is not outlined here; for a more detailed explanation, see Jones et al. 2011.

The clear articulation of desirable biodiversity outcomes then drives relevant conservation actions through different levels of the mitigation hierarchy. For example, an ecosystem-focused target to drive action in the *avoid* part of the mitigation hierarchy might be: by 2020, twenty-five per cent of areas currently in a predominately natural state in each of Earth's 825 terrestrial ecoregions (Olson et al. 2001) and 232 marine ecoregions (Spalding et al. 2007) will have full no-take protected area status and non-declining biodiversity value relative to a 2017 baseline (Figure 2.2). A species-focused target to drive action in the *minimise* part of the hierarchy could specify: by 2020, all fish stocks are managed according to the Food and Agriculture Organisation (FAO) Code of Conduct for Responsible Fisheries (FAO 1995) and all forests according to the Resolution on Sustainable Forest Management (United Nations 2008).

In this way, a global mitigation hierarchy framework could help achieve a desired future state of biodiversity by setting multiple goals and targets at meaningful scales, measured through relevant biodiversity indicators. We present one example of setting goals and targets in Figure 2.2.

2.4.2 Frames of reference and counterfactuals

Assessing achievement of NNL requires specification of a frame of reference, containing a biodiversity baseline or counterfactual scenario (Bull et al. 2015; Maron et al. 2018a; Table 2.2). This could take the form of a static baseline (i.e., biodiversity levels at a fixed point in time), for example, the current state of biodiversity (i.e., 2017 levels) as expressed using the chosen indicator set. Alternatively, a historic level of biodiversity could be set as a static baseline, such as species status in the year 1990 (to be compatible with the baselines used in the United Nations Framework Convention on Climate Change). Or a counterfactual scenario could be chosen, such as the expected state of nature in the absence of any further development or conservation interventions (Figure 2.2).

A frame of reference is key to incorporating biological resource extraction into the framework. For example, incorporating a NNL of biodiversity goal into the management of a natural resource, such as fish stocks, does not require compensation for losses related to this harvest if the baseline is 'current biodiversity status' and the stock in question is sustainably harvested and non-declining. By contrast, taking a pre-exploitation baseline, or evaluating against a reduced human impact counterfactual, could require compensation for lost

Table 2.3. Applying the mitigation hierarchy to the examples of housing development and commercial fisheries bycatch, to demonstrate its applicability at multiple scales and for different sectors.

Mitigation hierarchy step	Harmful event: Housing development leading to loss of biodiversity and habitat			Harmful event: Pacific leatherback sea turtles bycaught in commercial fisheries		
	Local (one house built)	National (State housing plan implemented)	Global (human urbanisation footprint increasing)	Local (one turtle killed by one vessel)	National (local extinctions or population reduction in a nation's Exclusive Economic Zones)	Global (species sent to extinction)
Avoid	Restriction of building permissions to given areas only	Strategic plan identifies areas set aside for housing and areas for conservation	International protected area commitments	Enforcement of small-scale time / area closures	Nationally legislated caps on turtle takes for countries operating fisheries in areas frequented by turtles	Multi-national no-take fishing zones tracking leatherback turtle migration
Minimise	Drainage areas, fence to prevent overflow of extracted dirt	Regulatory requirements for house building	International lenders require all new housing to be ecologically friendly	Gear modification resulting in an increased likelihood of turtle survival	Fleet wide gear changes (e.g., implementing circle hooks, branch lines long enough to allow turtles breathing at the surface, effort restrictions)	Demand reduction through international education campaigns targeting consumers of Pacific sourced tuna and swordfish
Remediate	Restoration of land along digger tracks	Land area restoration plans at State scale	International fund for urban greening projects	Better turtle handling and gear removal practices resulting in higher survival rates for post-capture release	Increased marine protected area monitoring & enforcement resulting in fewer illegal fishing events allowing turtle populations to recover	Protection and reallocation of nests to increase hatching success at known Pacific leatherback turtle nesting sites throughout range
Offset	Protect an area of existing wetland or create a new wetland nearby	State supports the protection of similar natural areas in other parts of the country	International fund for the restoration of habitat types preferentially affected by urbanisation	Protection of nesting turtles and their eggs at local nesting beaches & restoration of degraded nesting sites	Protection of nesting turtles and their eggs at nesting beaches within another area of the country	Protection of Atlantic leatherback sea turtles in an effort to ensure they don't meet the same fate

biodiversity even if harvesting is sustainable. It is also important to note that harvesting can be sustainable under a target species-focused goal and still have adverse effects on non-target biodiversity which would need to be compensated under another part of the overall framework (e.g., to address the negative impact of leatherback turtle bycatch from long-line fishing; Table 2.3).

When setting a frame of reference, there is also an essential need to clearly specify which elements of biodiversity are or are not appropriate to address at lower stages of the mitigation hierarchy (such as through offsetting). Some elements may be deemed too valuable to incur any human impact, and therefore impact must be avoided (Table 2.2; Bull et al. 2013). There are many situations in which offsets are unacceptable, regardless of whether large multipliers are applied (e.g., more than 10 units of habitat supplied elsewhere for every one unit destroyed; Moilanen et al. 2009). Irreplaceability is one criterion for whether biodiversity damage should be allowed and then offset. This may relate to a critically endangered or endemic species, a keystone species, an iconic area of wilderness, or biodiversity characterised by long restoration times, such as deep-sea coral systems, hydrothermal vents and old-growth forest. For example, a recent study demonstrated that if delays between a development project and the compensation of the resultant biodiversity losses through restoration are ≥ 55 years, then an offset is unlikely to be successful at achieving a NNL effect on biodiversity (Gibbons et al. 2016). Improving international recommendations for "no go" areas to protect biodiversity (e.g., that all categories of Protected Areas and World Heritage sites be considered 'no go' areas for large scale development; IUCN 2016c), backed up by national legislation, could help address this issue for a global mitigation hierarchy, and provide a strong and agreed basis for the *avoid* step of the hierarchy (Phalan et al 2017).

The lack of counterfactuals remains a widespread problem in practice, both for the mitigation hierarchy as currently applied to development (e.g., Maron et al. 2015a), and in the wider conservation and environmental policy literature (e.g., Ferraro 2009). It is only just starting to be applied to measuring the impact of traditional conservation interventions (e.g., Hoffmann et al. 2015). Failure to properly consider counterfactual scenarios promotes the idea that loss in one



Figure 2.2. Key steps required to implement a global no net loss of biodiversity target through the mitigation hierarchy, with associated goals and targets. The left column shows the basic framework for setting a global no net loss target. The right column gives a specific example focusing on the International Union for Conservation of Nature's Red List. This example shows one particular set of approaches among many that would be needed to achieve global no net loss human impact on biodiversity.

place can be offset by "protection" in another, even if that protection involves no more than re-labelling already-secure places (Maron et al. 2015b). A global mitigation hierarchy with a clear set of goals and targets would enable the integration of the different commitments and legislative requirements already in place, facilitating explicit consideration of how commitments at different scales complement or conflict with each other. Transparent consideration of baselines, and of where each biodiversity conservation action sits within the levels of the hierarchy, would reduce the risk of indirect leakage of environmentally damaging activity to other areas following locally-avoided losses (Moilanen & Laitila 2016). It would also mitigate against perverse outcomes such as governments using industry money generated by offsets to achieve existing national biodiversity commitments (Maron et al. 2015b; see additionality in Table 2.2).

2.4.3 Ensuring equity and subsidiarity

An important consideration for a global biodiversity conservation framework is the equitable distribution of costs and benefits between stakeholders (Ives & Bekessy 2015; Bull et al. 2017). For example, the management of any global biodiversity conservation goal through the mitigation hierarchy could follow a similar framework to the United Nations' management of carbon emissions, with nation states setting their own national goals and targets that then sum to achieve overarching planetary goals. Managing the framework in such a way could allow for equity between nations, recognising that industrialised countries reached their present wealth through exploiting natural resources and reducing biodiversity. Mediated by the Intergovernmental Panel on Climate Change, mechanisms exist which allow for transfer of funds and capacity from richer to poorer countries to enable the latter to meet their obligations (i.e., the Central African Forest Initiative; Müller 2016), as well as a staged process for poorer countries to reduce emissions in line with their capacity to do so. A similar framework for differential development, such that the burden of reducing impacts on biodiversity was equitably distributed, could support the achievement of a global NNL of biodiversity goal. Such an adjustment could also consider the international market drivers of biodiversity loss, for example, China's demand for soy (mainly as cattle feed) driving biodiversity loss in Brazil's Cerrado, a biodiversity hotspot of conservation priority (Strassburg et al. 2017).

This raises the issue of the equivalency of biodiversity in space and time, between biodiversity types, and by type of conservation action (equivalency of offsets; Table 2.2). We are not advocating the creation of a global market allowing the trading of biodiversity offsets towards>NNL over large scales; instead, the mitigation hierarchy must be applied at biologically meaningful scales to avoid "out-of-kind" actions that allow one part of the planet to be damaged in return for enhancement of others (BBOP [Business and Biodiversity Offset Programme] 2012; Bull et al. 2013). Although organisations like the United Nations can endorse best practice at the international level, individual nations would need to implement the legal framework that would ultimately drive adherence. This is increasingly happening within the industrial development sector, with countries supported to draft appropriate legislation and build capacity for implementation (e.g., the COMBO Project ; combo-africa.org). In addition, as is the case for all conservation actions, effective monitoring, independent evaluation and sanctions are required over the long term to ensure compliance with agreed targets and actions at all levels.

2.4.4 Categorising conservation actions within the framework

The framing of global conservation efforts in terms of a mitigation hierarchy, for all human impacts on biodiversity, is novel. However, the interventions constituting the different components of such a hierarchy – at the international, national, landscape and project levels – are already in place. Presently we know that most of the terrestrial environment is exposed to some form of human impact (Watson et al. 2016a) and no area of the world's oceans remain free from human pressures (Halpern et al. 2015). The options for avoiding intact biodiversity (devoid of significant human impact) are already significantly constrained by the current human footprint. The benefits of complete retention of large intact areas of wilderness are self-evident to many conservationists (Watson et al. 2016b), as are the benefits of avoiding destruction of small but important areas of biodiversity value within modified settings (such as sites containing populations of very vulnerable species, for example, AZE sites; Ricketts et al. 2005). However, the opportunity costs of degrading many of these areas are not currently well articulated; adopting the mitigation hierarchy framework would catalyse consideration of these costs, because it requires the comparison of relative biodiversity gains achievable at each step of the hierarchy and the associated uncertainties.

For example, the incidental environmental impacts of deep-sea fishing gear contacting continental slopes and offshore seamounts are rarely accounted for in fisheries policy (Clark et al. 2016). Making a requirement of NNL for biodiversity targets legally binding (e.g. for all United Nations fisheries through UNCLOS) would drive stipulation of defined baselines, indicators, and best practice implementation concerning deep-sea fishing using National Biodiversity Strategies and Action Plans (NBSAPs), formalised through ESIA processes. This would drive stronger avoidance and minimisation actions for deep-sea fishing nations, due to the high level of uncertainty surrounding whether it is possible to generate biodiversity gains for benthic deep-sea organisms like corals using remediation and offset measures such as the construction of artificial reefs.

What kinds of conservation action fall within a given stage of the mitigation hierarchy depends crucially on the baseline, goal and target chosen (Table 2.1). Taking a 2017 static baseline, for example, avoidance would comprise efforts to ensure that existing but currently unprotected areas of biodiversity value are preserved at the current status rather than being developed (e.g., to meet an area-based target this could be done through new PAs), minimisation reduces the damage of future developments on existing biodiversity in the newly developed areas (e.g., taking a species-based target this could be done by minimising the extent of new roads in close proximity to PAs), remediation increases the biodiversity values associated with new human impact (e.g., for an ecosystem-based target this could be done through clean-up of new pollution in fished coastal areas), and offsetting improves biodiversity over the current status quo in ways or locations not associated with a particular new impact (e.g., for an ecosystem target by mangrove reseedling, or for a species-based target by eradication of invasive species). Any entities causing new or ongoing biodiversity damage at the local, national or international level (from road building to non-target fishing impacts to climate change) would need to demonstrate how they were investing in conservation in a way that would appropriately balance that damage to meet the goals and targets set out at the same spatio-temporal scale, and institutional level, as the damage.

Current protection status of terrestrial ecoregions is being mapped to prioritise conservation actions (Dinerstein et al. 2017). Similar mapping efforts have begun for forest restoration opportunities (Potapov et al. 2017). Maps such as these could provide the roadmap for

global biodiversity conservation called for by scientists (Watson & Venter 2017), and guide avoidance, mitigation and restoration activities, and highlight opportunities for offsetting, within the mitigation hierarchy. For example, the effectiveness of the remediation step is open to question, with evidence suggesting that artificial or restored ecosystems do not reach the levels of ecological functionality of natural systems (Moreno-Mateos et al. 2012). At the project level, costing each step at a level which reflects biodiversity gains and losses (with associated uncertainties) could incentivise developers to move up the hierarchy, because avoiding sensitive sites could be made significantly cheaper than developing them and then offsetting.

2.5 Conclusions

Scaling up and expanding the mitigation hierarchy concept will provide a systematic framework within which to think about what humanity wants for the planet's natural systems and how we could get there. It will help overcome the lack of cohesion in conservation efforts, which has facilitated the continuing loss of the planet's biodiversity (Rands et al. 2010). A global mitigation hierarchy framework could act as the foundation for a biodiversity conservation roadmap, that would allow the international community to get behind a strategic goal and understand what is needed to fulfil their commitments to biodiversity. This would result in more explicit consideration of humanity's capacity to conserve different components of biodiversity. It forces the consideration of key questions such as what baseline for biodiversity we are evaluating against, how much damage could be averted or minimised given where we are now, and what this implies for the requirement for more uncertain restoration and offsetting.

Nations are uniting on the issue of biodiversity loss and setting aspirational global goals and targets, and there are a number of effective systematic planning processes already in place at national levels. However, there is an obvious need for a more strategic and coherent global approach to deal with the loss of biodiversity, as current efforts are manifestly failing. The mitigation hierarchy is one potential framework, which would force the explicit consideration of the relationship between conservation and development, and how sustainable development can be achieved. Such a reframing could be a step towards a clearer strategy for keeping within our planetary boundaries, for the sake of both humanity and nature.

Chapter 3

A mitigation hierarchy approach for managing sea turtle captures in small-scale fisheries

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3.1 Introduction

Fisheries often seek to achieve ‘triple-bottom-line’ outcomes that entail trade-offs between economic returns, social welfare, and biodiversity conservation (Halpern et al. 2013; Costello et al. 2016b). Managing the recovery of depleted populations of marine megafauna species, which are defined as large-bodied, ocean dwellers like sea turtles, seabirds, marine mammals, and sharks, often sits in the middle of this nexus and persists as one of the major challenges in achieving ecologically and socioeconomically sustainable fisheries (Hall et al. 2000; Gray & Kennelly 2018; Lewison et al. 2018). The complex and dynamic nature of attempting to target catch while minimising the impact on non-target species means that fisheries management requires integrative processes to identify and mitigate the negative ecological impacts of fisheries while examining economic and social considerations on a fishery-by-fishery basis.

A variety of risk-based decision-making processes to assess the ecological impacts of fishing have been developed – also commonly known as ecological risk assessment (ERA; Lackey 1994; Hobday et al. 2011). Management Strategy Evaluation (MSE) is a complementary simulation-based process for assessing trade-offs in potential management strategy performance

(Smith 1993, 1994; Fulton et al. 2014). While these and other structured decision-making processes are vital for fisheries management, there remains a need to further integrate fishery-specific management into national and international goals for biodiversity conservation – for example, those specified by multilateral agreements like the Convention on Biological Diversity (CBD). Since the 1992 adoption of the CBD (United Nations 1992), the Food and Agriculture Organisation (FAO) and regional fisheries management organisations (RFMOs) have made substantial progress in mainstreaming biodiversity conservation into fisheries management processes through frameworks, policies, and practices aimed at promoting more sustainable fishing practices (Friedman et al. 2018). But it is necessary to further support integrated partnerships between fisheries and the wider environmental sector, particularly in lower-income countries, to ensure beneficial biodiversity conservation outcomes across fisheries at scale (Karr et al. 2017).

The mitigation hierarchy is a conceptual framework that can support integrating fisheries management with biodiversity conservation objectives (e.g., a scalable framework for linking actions to reduce sources of anthropogenic mortality over a species life cycle, migratory range, and habitat). In terrestrial and coastal ecosystems, the mitigation hierarchy is widely used as part of the decision-making process of Environmental Impact Assessment (EIA; CEQ [Council on Environmental Quality] 2000) to identify and manage the negative impacts of human economic activities on biodiversity – most commonly applied to infrastructure development projects (e.g., roads, mining sites, wind farms; Bennett et al. 2017; Shumway et al. 2018). If implemented effectively, the framework can help to guide actions towards mitigating the negative impact on biodiversity following a traditionally damaging or extractive activity (zu Ermgassen et al. 2019a). Following widespread application in terrestrial and coastal development projects (Maron et al. 2016; Shumway et al. 2018), the mitigation hierarchy was proposed as an overarching framework for mitigating marine megafauna bycatch in fisheries (Milner-Gulland et al. 2018; Squires et al. 2018), and more broadly, for all human impacts on biodiversity (Arlidge et al. 2018).

A key benefit of the mitigation hierarchy is that it begins by setting a desired end-goal that can support the summation of multiple positive and negative impacts into a net, scalable, outcome (Bull et al. 2019). This goal is conventionally a no net loss or a net gain of biodiversity (Rainey et al. 2015). In a fishery setting, goals such as population recovery when managing

protected species, or Maximum Sustainable Yield Biomass (*B_{msy}*) when managing stocks of target catch, are equally feasible (Wolf et al. 2015; Squires & Garcia 2018a). The chosen goal is then measured using a quantitative target and metric(s) with reference to a baseline of biodiversity. Following goal-setting, the framework follows a step-wise decision-making process to identify a suite of measures for mitigating the negative impacts of human activity on biodiversity to achieve the specified goal. The mitigation hierarchy progresses in four sequential stages. The first three - avoid, minimise, and remediate – take place at the impact site (i.e., at sea where fishing is taking place). Then if any residual negative impacts remain, biodiversity offsetting actions can take place through off-site compensatory measures (CEQ [Council on Environmental Quality] 2000; Bonneuil 2015). All actions may not be applicable in all management scenarios (e.g., in-kind offsetting actions are not feasible for deep-sea trawl impacts on seamounts; Niner et al. 2018). Rather, the broad steps of the mitigation hierarchy act as a guide, with enough flexibility to achieve the integration of diverse fisheries management approaches towards a unified biodiversity goal that translates across scales (Milner-Gulland et al. 2018; Squires et al. 2018). Yet despite its theoretical attractiveness, there remains a need to empirically evaluate how the mitigation hierarchy can support fisheries management and bycatch mitigation in practice.

Peru's small-scale fisheries total more than 16,000 fishing vessels, with an estimated 44,161 fishers and 12,398 ship owners (Guevara-Carrasco & Bertrand 2017). Of these vessels, approximately 4800 fish primarily with gillnets (Estrella & Swartzman 2010). In Peru, the capture of sea turtles in coastal gillnets is a major conservation issue in the nation's northern fishing ports and landing sites (Alfaro-Shigueto et al. 2011; Alfaro-Shigueto et al. 2018). Gillnet fishing also plays an important role in food security, local employment, and social identity throughout Peru's coastal communities (Christensen et al. 2014). I explore the applicability of the mitigation hierarchy as an overarching framework for managing the population recovery of depleted sea turtle populations, by integrating multiple sources of data, highlighting uncertainties, and supporting management decisions that consider biological, social, and economic conditions in the coastal gillnet fishery. Throughout this investigation, attention is paid to integrating the established decision-making processes of ERA and MSE with the mitigation hierarchy.

I draw on qualitative ERA (consequence $\times$ likelihood) theory to consider risks and associated impacts on multiple turtle species captured in our case study fishery (Fletcher 2014). I then consider the integration of a qualitative MSE assessment with the mitigation hierarchy to measure the performance of management options aiming to reduce turtle captures and consider how they trade-off against economic and social considerations (Smith et al. 2004; Dichmont & Brown 2010). The management objectives sought through both ERA, and MSE assessments, are typically fishery or management-region specific (Fulton et al. 2011b; Fletcher 2014). Objectives typically focus on achieving economic efficiency and ensuring that exploitation is consistent with the principles of ecologically sustainable development, and the exercise of the precautionary principle. The mitigation hierarchy's goals, by contrast, are often specifically chosen to be translatable between global, management-region, national, and fishery scales. Goals and targets at the fishery level may vary depending on the biodiversity component assessed, as well as data availability and capacity while combining to achieve the overarching goal at higher levels (Milner-Gulland et al. 2018). The mitigation hierarchy framing, therefore, constitutes a shift in approach to objective-setting at an individual fishery level towards the summation of positive and negative impacts into a net, scalable, outcome (Bull et al. 2019). Economic and social management benchmarks could also be set using a mitigation hierarchy framework. An economically-focused goal may seek to maximise net economic returns to the community within a fisheries management region, measured by summing fishery-related profits and losses against a predetermined baseline. A social goal may seek to ensure that the community within the fisheries management region is no worse off, or preferably better off, in terms of their wellbeing as a result of fishery management (Griffiths et al. 2018).

In this study, I explore the potential of the mitigation hierarchy for integrating fisheries management and biodiversity conservation processes, with a focus on achieving population recovery goals for captured sea turtles: a globally endangered taxon primarily threatened by negative fisheries impacts (Wallace et al. 2010b). The mitigation hierarchy builds on ERA and MSE in two ways: Firstly, it requires a clear definition of management benchmarks (a biodiversity goal and associated targets measured against a baseline of biodiversity) that are generalisable across fisheries and scales (Milner-Gulland et al. 2018). Secondly, its consideration of a broad suite of purely technical conservation actions and market-based mechanisms for environmental conservation (avoidance, minimisation, remediation, offsetting) – rather than the

common focus on at sea minimisation and remediation – encourages a more holistic recovery strategy for marine megafauna species. This can support an integration of measures to reduce anthropogenic mortality over a species life cycle, migratory range, and habitat into the management process (Dutton & Squires 2008; Squires et al. 2018). The mitigation hierarchy can also help to identify key uncertainties and knowledge gaps, as well as difficult trade-offs between biodiversity conservation goals, different management strategies, and other socioeconomic objectives of fishers and fisheries.

I implemented the mitigation hierarchy in an iterative process; collating existing data to characterise the fishery and the turtle capture problem, identified areas of uncertainty, and gathered primary data to address key knowledge gaps. I then integrated all existing and gathered data under the mitigation hierarchy framework to assess risk, using methods taken from ERA. We (the authors) explored how potential management measures can be assessed by drawing on qualitative MSE methods. Finally, I discuss the potential for, and limitations of, the proposed mitigation hierarchy framework, with a focus on the need to better integrate diverse fisheries management approaches, impacts and mitigation actions across scales.

3.2 Methods

3.2.1 Study site

San Jose, Lambayeque, Peru (6°46' S, 79°58' W) is a key site for coastal gillnet fishing (Guevara-Carrasco & Bertrand 2017). Longline, purse seine, trawl, squid jigging, handline fishers, and divers also operate from the community. Amongst the diverse range of fishing gears, gillnets are the most prevalent. Two distinct gillnet fleets operate from San Jose. First, the 'San Jose inshore gillnet fleet' (IG) comprises a class of open-welled vessels with a capacity range from 1-8 t. Second, the 'San Jose inshore-midwater gillnet fleet' (IMG) comprises a larger vessel class with small closed bridges ranging in capacity from 5-32 t. I refer to the 'San Jose gillnet fishery' when referencing both fleets together. Fishers operating in the San Jose gillnet fishery use both surface driftnet and fixed demersal nets configurations, with some fishers switching between the two on a single trip (Alfaro-Shigueto et al. 2010).

I focus on a single marine megafauna taxon for our assessment of bycatch impacts - sea turtles (superfamily Chelonioidea). Three turtle species are regularly captured in the San Jose gillnet fishery (Alfaro-Shigueto et al. 2007; Alfaro-Shigueto et al. 2018). The eastern Pacific populations of leatherback sea turtles *Dermochelys coriacea*, green turtles *Chelonia mydas*, and Olive ridley turtles *Lepidochelys olivacea*. The eastern Pacific population of leatherback turtles is of particular conservation concern as is presently at critically low numbers with a declining population trend (Spotila et al. 2000; Wallace et al. 2010a; Mazaris et al. 2017).

Peru has a history of sea turtle consumption (Aranda & Chandler 1989), and turtles are still consumed despite protection under Peruvian law since 1995 (Morales & Vargas 1996). Thus, I make a distinction between the capture of turtles in fishing gear, the targeted and non-target use of captured turtles, and bycatch (the capture and discard at sea, dead or injured to an extent where death is the result), following the definitions used in Hall (1996).

In Peru, the Peruvian Marine Research Institute IMARPE (Instituto del Mar del Peru) conducts government-managed marine research. The Peruvian Coastguard DICAPI (Dirección de Capitanías y Puertos) undertakes enforcement in most cases. Despite an IMARPE and DICAPI presence in San Jose, Peru's current regulatory structure does little to help mitigate the capture of marine megafauna species like sea turtles. San Jose's gillnet fishery operates as an open-access fishery – with no catch restriction (e.g., a total allowable catch; TAC) in place for target species (Bjørndal & Conrad 1987). In 2010, the Peruvian government implemented an effort restriction with a ban on new boats above 5m³ gross registered tonnage (GRT) entering the nation's small-scale fisheries (Supreme decree N° 018-2010-PE), but limited enforcement of this rule means that vessel builders still operate actively (Estrella 2007; Christensen et al. 2014).

The fishery is predominantly beyond the reach of RFMOs, and trade measures are limited by the coastal desert environment. In San Jose, many coastal gillnetters work alternate jobs, often throughout the winter when fishing effort and catches are low because of rough weather conditions preventing fishing. Few options exist for alternate revenue streams for fishers in San Jose (e.g., 'mototaxi' driver, construction worker, general store clerk). Alternate livelihoods such as these are often tied to the success of local fishing (i.e., more people will use local transport or spend money at local shops when their revenue is high from a good fishing period). With limited

regulatory efficacy, not-for-profit organisations play a key role in filling data gaps and implementing conservation interventions in this data-poor, open-access fishery.

3.2.2 Applying the Mitigation Hierarchy

I use the mitigation hierarchy as a conceptual model and framework for structuring data and generating management recommendations towards a standardised biodiversity conservation goal. Milner-Gulland et al. (2018) present two main steps to make the mitigation hierarchy relevant to fisheries management and mitigating marine megafauna bycatch. These are i) defining the problem (by characterising the fishery and bycatch issue, and setting the goal, target, metric and baseline), and ii) exploring potential management options by systematically stepping through the mitigation hierarchy using a conceptual framing. Booth et al. (2020) explore the potential for the application of the mitigation hierarchy to shark bycatch management. These authors subdivide the two steps in Milner-Gulland et al. (2018) into five. These include i) defining the problem, ii) exploring potential management measures using the mitigation hierarchy, iii) assessing the hypothetical effectiveness of management, iv) making an overall management recommendation or decision, and v) implementing, monitoring, and adapting implemented management measures. Here I use the steps proposed in Booth et al. (2020), in combination with data from a real-world fishery, to explore the advantages and disadvantages of the mitigation hierarchy framework. I further develop the framework by exploring its potential for integration with MSE to evaluate trade-offs between management scenarios.

3.2.3 Data collection and analysis

I adopted a mixed-methods iterative approach to data collection and analysis, drawing on primary and secondary data sources and multiple analytical methods to understand the fishery problem and explore potential management measures.

3.2.3.1 Understanding the problem: the fishery and species of concern

I collated all available information on the San Jose gillnet fishery and each of the turtle species of management concern from published and unpublished sources using a literature review and

available datasets. I then collected primary data through field-based surveys to fill several key knowledge gaps.

3.2.3.2 Secondary data

I sourced secondary data on turtle bycatch in the IMG fleet from a voluntary at sea human observer program managed by a local not-for-profit, *ProDelphinus*. This program has been operating with skippers and crew of IMG vessels along Peru's coastline since 2007. Observer surveys have been undertaken in the IMG fleet since the program's inception, but there are no site-specific turtle capture or bycatch per unit effort rates calculated. No observer data exists for the IG fleet, but I was able gain insight into the turtle species captured in this fleet from existing data collected through harbour-based surveys of fishers and local government representatives (e.g., Alfaro-Shigueto et al. 2007; Alfaro-Shigueto et al. 2018).

I also collated relevant information on leatherback, green, and olive ridley turtles with consideration for management in our case study fishing system. I identified potential turtle capture and bycatch reduction strategies based on a literature search, which was later refined using stakeholder consultation (see Primary data).

3.2.3.3 Primary data

To better understand the fishery impacts on sea turtles, I collected primary data to quantify the fishing seasons and geographic extents of the two gillnet fleets. To quantify local fishing seasons, I conducted key informant interviews with a local IMARPE scientist and the presidents of the two at sea fishing groups in San Jose (with support from my research assistants). To estimate the geographic extent of the gillnet fleets, I used a combination of key informant interviews and focus group discussions (FGDs). I held two FGDs, one for each gillnet fleet. The FGD estimating the IG fleet's geographic extent had 15 participants, comprising 13 skippers of inshore gillnetting vessels, an IMARPE scientist, and a not-for-profit employee (JAS). The FGD estimating the IMG fleet's geographic extent had five participants, comprising three gillnet skippers and two not-for-profit employees (JAS & JCM). I used simple random sampling by number generator to select gillnet skippers from lists of 150 actively fishing IG skippers, and 18 actively fishing IMG

skippers. I assigned skippers fishing within each fleet to the relevant FGD. For supplementary analysis, I present a summary of demographic data (see Supplementary Materials).

I asked respondents to estimate the maximum geographic range that fishing vessels from their fleet travelled from San Jose (north, south, west). Respondents' maximum geographic extent was then averaged across each group's participants and displayed using ArcMap (ESRI 2018). I gave the respondents the option to input additional information or adjust their estimates. No respondents adjusted their estimates in this final round. I collected all primary data during field surveys in San Jose from 1 July – 30 September 2017. This research has Research Ethics Approval (CUREC 1A; Ref No: R52516/RE001 and R52516/RE002).

3.2.3.4 Assessing fishery risks

To better quantify fishery risks I first analysed available at sea fisheries observer records from the IMG fleet from August 2007 to March 2019. I calculated turtle captures per trip for the IMG fleet and considered the portion of mortalities and captures returned to sea injured or unharmed. I used an analysis of variance and a post hoc Tukey test to compare capture rates between species groupings. All analyses were completed using core packages in R (R Core Team 2019).

To evaluate the risks for sea turtle populations captured in the San Jose gillnet fishery, I use the consequence–likelihood (probability) matrix methodology that originated from Australian and New Zealand Standard Risk Analysis (Standards Australia 2000, 2004) for fisheries management (Fletcher et al. 2003; Fletcher 2005). The methodology is widely implemented (e.g., Fletcher 2008; FAO 2012). Iterative updates to the ERA method have followed to ensure compliance with the revised international standards for risk management (ISO 2009) and to enable consideration of ecological, economic, social, and governance risks (Fletcher 2014).

I focused on direct risks posed to turtles captured in our case study fishery (addressing both the IG and IMG fleets) relative to each species distribution and estimated population sizes throughout their respective Pacific East regional management unit (RMU), as developed by Wallace et al. 2010a. RMUs delineate global turtle populations according to regional areas that are distinct from one another based on genetics, distribution, movement, and demography, and provide a practical management unit for assessment analogous to the IUCN - World

Conservation Union's Red List of Threatened Species subpopulation categorisations, but for all extant marine turtle species (Wallace et al. 2010a). RMUs allowed for an evaluation of the relative risk posed from the two San Jose gillnet fleets to each turtle species' population that is directly affected by fishing activity within our case study system. The analysis assessed how the biology and distribution of each species within the Pacific East RMUs affected susceptibility to the risk from each gillnet fleet, and whether the current management arrangements in place in our case study fishery (i.e., fishing regulations and compliance therewith) were working effectively or not. Consideration was also given to the wider fishing impacts on each species throughout their respective Pacific East RMU distributions (see Supplementary Materials). When implementing an ERA in full, a complete evaluation of all risks posed to all target catches, non-target catches, habitat, and social and governance structures across the focus fishery is necessary (Fletcher 2014).

Critically, risk analysis evaluates the level of risk that a given impact (e.g., incidental capture in gillnets) poses to achieving the goals and targets set over a specified assessment period with the current management measures in place (Fletcher 2014). I evaluated the risk posed from the IMG and IG fleets against achieving the high-level biodiversity goal of population recovery of leatherback, green, and olive ridley turtle populations (Pacific East RMUs) in the shortest time possible (in line with international biodiversity targets). The mitigation hierarchy framework specified that goals must be operationalised through quantitative targets, for which metrics and baselines can be defined (Milner-Gulland et al. 2018). The San Jose gillnet fishery does not have management benchmarks in place to meet high-level goals of turtle population recovery. Thus, I propose a fishery-specific target of reducing turtle captures from 2020 levels by 15% every year for five years while maintaining total catch weight. As more data becomes available and population models develop, a net change in population growth rate target measured against an agreed baseline is recommended (Milner-Gulland et al. 2018).

I ranked the risk from each gillnet fleet in terms of a consequence (C) level (specifying a level of impact) the fishing fleet in question is likely to have for each turtle species assessed, using a four-point scale from minor [1] to extreme [4], and the likelihood (L) that a specific consequence level will occur, also using a four-point scale from remote [1] to likely [4] (Table 3.1). Sources of risk (i.e., the two San Jose gillnet fleets) were then assigned a score for each

Table 3.1. Consequence (level of impact) and likelihood (a subjective probability) descriptors used to evaluate identified risks (following Fletcher 2014).

Level	Descriptor
Consequence for protected species	
Major (C4)	Further declines generated and major ongoing public concerns
Severe (C3)	Recovery may be affected and/or some clear public concern
Moderate (C2)	Catch or impact at the maximum level that is accepted by public
Minor (C1)	Few individuals directly impacted in most years, no general level of public concern
Likelihood of a specific consequence occurring to protected species	
Likely (L4)	A particular consequence level is expected to occur in the time frame (indicative probability of 40–100%)
Possible (L3)	Evidence to suggest this consequence level may occur in some circumstances within the time frame (indicative probability of 10–39%)
Unlikely (L2)	The consequence is not expected to occur in the time frame but some evidence that it could occur under special circumstances (indicative probability of 3–9%)
Remote (L1)	The consequence not heard of in these circumstances, but still plausible within the time frame (indicative probability 1–2%)

Table 3.2. Consequence (C) × likelihood (L) risk matrix (following Fletcher 2014). The descriptions of each of the consequence and likelihood levels are presented in Table 1. The numbers in the cells indicate the risk score values and the colours/shades represent the levels of risk as described in Table 1. The level of impact is determined by summing C × L. Impact levels include: minor (1-2), moderate (3-4), major (6-8), and extreme (9-16).

		Likelihood level			
		Remote	Unlikely	Possible	Likely
Consequence level		1	2	3	4
Minor	1	1	2	3	4
Moderate	2	2	4	6	8
Major	3	3	6	9	12
Extreme	4	4	8	12	16

turtle species, calculated by multiplying the consequence and likelihood values (e.g., consequence level of impact x on turtle species $y \times$ the likelihood of consequence x occurring to turtle species y). The risk posed from each gillnet fleet for each turtle species were then assigned one of four levels of impact ranging from minor to extreme (Table 3.2). If more than one combination of consequence and likelihood was plausible, I chose the combination that generated the highest risk score (i.e., consistent with taking a precautionary approach; Fletcher 2014).

3.2.4 Exploring management options

Based on the quantified risks, I then used the conceptual framework for bycatch mitigation presented in Milner-Gulland et al. (2018) to consider how additional management strategies could be implemented to reduce the risk of fishing-related mortality for leatherback, green, and olive ridley turtles (equation 1):

$$\Delta\lambda_T = f(E_B \times \text{BPUE}) - O_T \quad (1)$$

In Milner-Gulland et al. (2018) the equation relates to a particular bycatch species, in which the unit ($\Delta\lambda_T$) is the rate of change in population size as a result of bycatch and its mitigation. $f(E_B \times \text{BPUE})$ is the effect on the population growth rate of the bycatch-relevant component of fishing effort, broken down into the bycatch-relevant effort, E_B , and the bycatch taken per unit of that effort, BPUE, where $f()$ is the effect of this effort on a given species of sea turtle's population dynamics. A reduction in E_B is equivalent to a fishery avoiding bycatch of turtle population x , partially or completely. A reduction in BPUE is the result of the on-site measures encompassed in the “minimise” and “remediate” steps of the mitigation hierarchy. O_T is the net effect on the population growth rate of policies aiming to improve the overall viability of turtle population x , representing the “offsetting” of any remaining residual damage caused, using measures away from where the fishing impact occurs (e.g., nesting site protection). In this data-poor case study, the relationship between BPUE and each turtle's population growth rate (i.e., $f()$) is unknown. As such, I do not attempt to solve equation 1 for a population growth target. Rather, I use the equation as a conceptual model for evaluating how management strategies can help reduce

different components of turtle bycatch risk, and for illustrating where a potential management strategy sits within the wider mitigation hierarchy. The flexibility of the model allows for components of the equation to be further deconstructed in to separate factors. For example, BPUE can represent the sum of individual turtle species x that are dead on arrival to the vessel, individuals captured and dying on the vessel, and individuals dying after live release, as follows:

$$BPUE = B_{DOA} + P_{DV} \times B_{OB} + (1 - P_{DV}) \times B_{OB} \times P_{DR} \quad (2)$$

where B_{DOA} is the bycatch per unit effort that arrives at the boat dead, B_{OB} is the bycatch per unit effort that arrives at the vessel alive, P_{DV} is the proportion dying on the vessel, and P_{DR} is the proportion dying after release. This decomposition can help with identifying different points for management interventions within the fishing process (Milner-Gulland et al. 2018).

To understand the feasibility of different management measures and support the selection of multi-strategy scenarios for the MSE assessment, I interviewed a subset of gillnet skippers operating in San Jose about their personal preferences for potential management options using questionnaires that gathered basic demographic information and incorporated a quantitative five-point Likert-scale assessing strong disagreement to strong agreement with each strategy proposed. The data were analysed using core packages in R version 3.6.1 (R Core Team 2019).

3.2.5 Assessing the hypothetical effectiveness of management options

To assess trade-offs in potential management strategy performance, I draw on the conceptual integration of the mitigation hierarchy with MSE (Bull & Milner-Gulland 2020) and demonstrate the implementation of the two processes in a data-poor management scenario. MSE generates simulations within an operational model such as the Atlantis model framework, adapted from the work of Fulton (2001). However, it is possible to assess management strategy scenarios against performance indicators qualitatively (e.g., area fished, catch, BPUE) derived through a process of expert judgment and stakeholder consultation (e.g., Smith et al. 2004; Dichmont & Brown 2010).

Qualitative MSE assessments can be undertaken in data-poor management scenarios as a preliminary assessment with the intent to undertake a quantitative evaluation of management scenarios during the next iteration of the management project (Dichmont & Fulton 2017). The evaluation phase implemented in the current study involved a project team (the authors) made up of several subject matter experts (with over 125 years of collective experience in conservation science and fisheries management research, and over 25 years of collective experience working in the case study fishery). The analysis was undertaken through an iterative web-based evaluation process, with participants drawing on their expert opinion and the collated and collected data presented in the current study (see Supplementary Materials for further presentation of data used during the assessment).

Based on indicators applied in MSE analyses (Smith et al. 2004), I compiled a list of performance indicators (Table 3.3) to evaluate management strategy scenarios against the high-level biodiversity goal (i.e., population recovery of the Pacific East RMU population for each turtle species), and the proposed fishery-specific target (i.e., reducing turtle captures from 2020 levels by 15 % every year for five years while maintaining total catch weight). It is assumed that managers would maintain capture rates at or below the five year target level going forward or update the target at the end of this assessment period as data becomes available.

The project team evaluated three management scenarios that were subjectively selected based on our fieldwork results, the compiled data, and the output of the ERA. Once I had specified the management scenarios and performance indicators (Table 3.3), I evaluated the consequence of each scenario by predicting how each performance indicator would change over a 10 year assessment period given the project team's knowledge and assumptions about the system dynamics of the San Jose gillnet fishery system (Smith et al. 2004). I chose a more extended assessment period for the qualitative MSE (10 years) over the ERA (5 years) to reflect a realistic timeframe for implementing potential management strategies. I present predicted trends in performance indicators. It should be highlighted that the current qualitative MSE assessment remains preliminary. When implementing a full MSE (whether this be qualitative or quantitative), potential management scenarios should undergo a broad stakeholder consultation and engagement process during which time, stakeholders representing different sector's interests can input ideas and submit other management strategy combinations for evaluation (as

undertaken in a qualitative MSE process for Australia's South-east Shark and Scale Fishery; Smith et al. 2004).

Table 3.3. Proposed performance indicators for assessing management scenarios against set goals, targets, and baselines for bycaught turtle species in the San Jose gillnet fishery.

Indicator
Technical/Biological
Threatened, Endangered, Protected species
BPUE
Leatherback turtle
Green turtle
Olive ridley turtle
Ecological Sustainable Development (ESD)
Impact from the San Jose gillnet fishery on biodiversity composition
Fishing effort
Geographic extent
Set number $\times$ set time
Distance travelled
Discards
Habitat & sessile communities
Socioeconomic
CPUE
Management costs
Stable management
Gear conflict
Revenue per tonne of fish landed
Revenue per day fished
Cost per day fished
Return on investment
Food security
Employment security
Local fish processing
Local transport, boat building and maintenance
Access to other services
Improvement in conservation values
Desire to participate in bycatch reduction initiatives in future
Social networks (leadership)
Formation of local institutions
Public perceptions of conservation
Trust and confidence in authorities

3.3 Results

3.3.1 The fishery and turtle bycatch rates

The San Jose gillnet fishery comprises two distinct gillnet fleets that fluctuate in vessel number and effort between the fishing seasons of summer and winter. The main uncertainties identified related to the geographic extent of the two gillnet fleets, fishing seasons length, and seasonal and annual fluctuations in fleet size (Table 3.4).

Respondents in the key informant interviews identified two distinct fishing seasons, with fishing effort varying between winter and summer conditions. In the northern regions of Peru, summer is usually December–February (3 months), but the government fisheries scientist noted that summer-like fishing conditions span December–May, with this longer seasonal division supported by the presidents of the two local at sea fishing groups, and by capture reports from the Lambayeque region (Guevara-Carrasco & Bertrand 2017). FGDs estimated the maximum geographic extent for two fleets comprising the San Jose gillnet fishery across the defined seasonal breaks (Table 3.4). I then overlaid observed turtle captures and fishing effort data from the IMG fleet to corroborate respondents’ estimates of this fleet’s geographic extent (Figure 3.1).

Two shore-based surveys recorded turtle captures in San Jose (Alfaro-Shigueto et al. 2007; Alfaro-Shigueto et al. 2018; Table 5). From July 2000 to November 2003, a combination of shore-based- and at sea observer surveys recorded nine leatherback turtle captures in San Jose (across gillnet and longline gear types; Alfaro-Shigueto et al. 2007). Turtle capture and bycatch rates were available for the towns of Constante, Salaverry, and Ilo (Alfaro-Shigueto et al. 2011). In Salaverry and Constante, most turtle captures in gillnets were green turtles (Alfaro-Shigueto et al. 2011; Figure 3.1). Turtle bycatch reports from Salaverry found leatherback turtles captured close to the coast, indicating a potential coastal foraging ‘hot spot’; if captured, consumption rates were high (Alfaro-Shigueto et al. 2007; Alfaro-Shigueto et al. 2018).

There were 461 fishing trips observed from San Jose, of which observers recorded the capture of 379 turtles in gillnets. Observer coverage for the IMG fleet is low at ~1–4% fleet coverage spanning 11 years and 7 months. Species proportions were 86.8% green (n=329), 9.2% olive ridley (n=35), 1.8% leatherback (n=7), and 2.1% unidentified hardshell turtle species

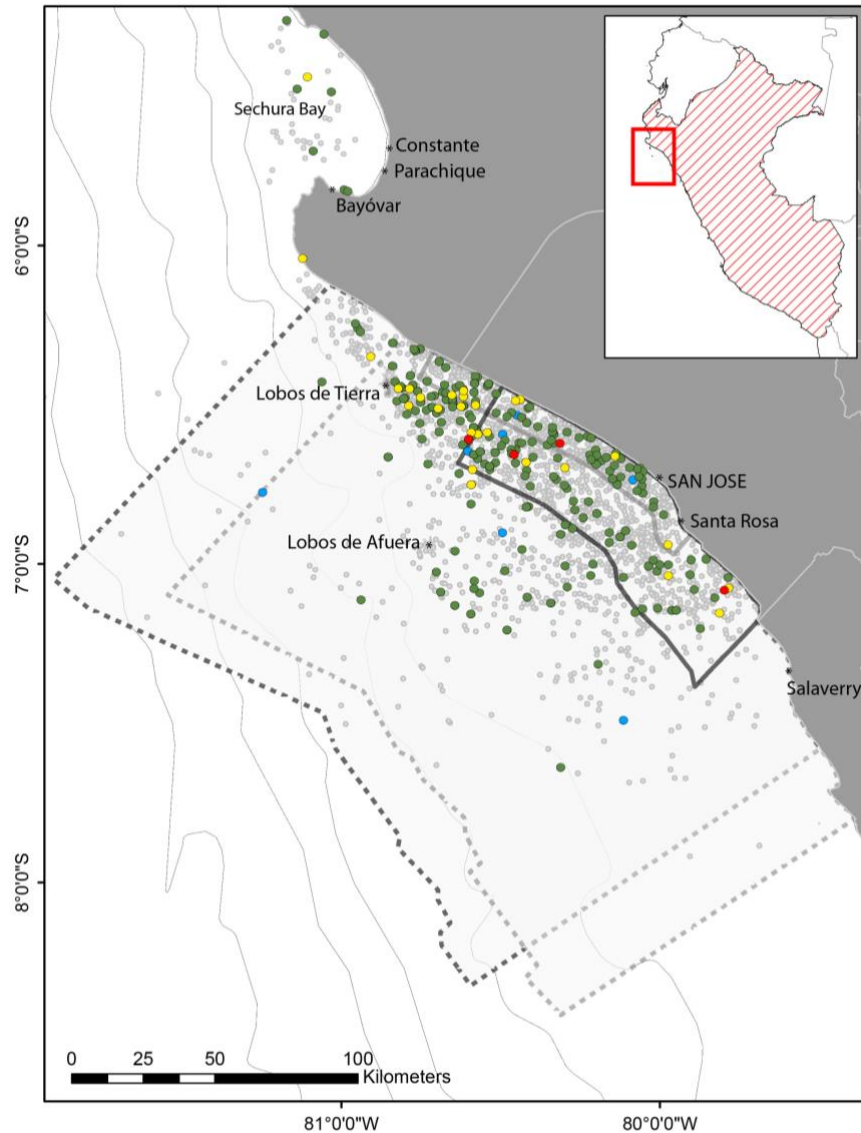


Figure 3.1. Estimated summer (December-March) and winter (June-November) geographic extent for the San Jose inshore-midwater gillnet (IMG) fleet, and the San Jose inshore gillnet (IG) fleet (dark grey dashed line = IMG winter, light grey dashed line = IMG summer, dark grey line = IG winter, light grey line = IG summer). The fleets' geographic extents are overlaid by a distribution of sea turtle bycatch by species (green circles = green turtles, blue circles = leatherback turtles, yellow circles = olive ridley turtles, red circles = unidentified hardshell turtle species) relative to observed fishing effort for the San Jose IMG fleet from August 2007 to March 2019 (small grey circles). No turtle bycatch data from the inshore gillnet fleet has been recorded. Fishing areas were elicited from San Jose gillnet skippers during focus group discussions. Distances are the maximum (group mean) distance skippers estimated any skipper fishes from San Jose (north, south, west). Captures and fishing effort north of Bayóvar show trips that either left or landed from San Jose but began or concluded at the Bayóvar port.

Table 3.4. Characteristics of the San Jose gillnet fishery, Lambayeque, Peru (6°46' S, 79°58' W). Here I define the bycatch problem by first collating lines of evidence on fishery type, fleet size and spatial extent, target catch, fishing seasons and relevant mark markets, and then evaluating known uncertainties. Text in **bold** highlights collected data filling identified knowledge gaps.

Fishery element	Lines of evidence	Uncertainties/filled data gaps
Vessel type	Peruvian law defines SSF vessels as displacing a maximum of 32.6m ³ Gross Registered Tonnage (GRT), up to 15m length, and operated predominantly manually (Legislative decree N° 012-2001-PE). San Jose vessels using gillnets can be divided into two distinct fleets: i) the 'inshore gillnet fleet' (IG) comprises vessels of 1-8 GRT, locally known as 'chalana', and ii) the 'inshore-midwater gillnet fleet' (IMG) comprising vessels of 5-32 GRT, locally known as 'lancha' (Guevara-Carrasco and Bertrand, 2017).	Rate of gear switching to gillnets from vessels that primary fish with another gear type.
Fleet size	IG fleet is increasing in size. The IMG fleet is thought to be decreasing in size as many fishers' switch from gillnets to squid jigging. Estimates of gillnet activity in San Jose recorded 47 gillnet vessels fishing in November 1995–April 1996 (Escudero 1997) and 95 gillnet vessels fishing in January–April 2004 (Alfaro-Shigueto et al. 2010). Skippers typically operate with 1-4 crew (Alfaro-Shigueto et al., 2010).	In the winter of 2017, the IG fleet was estimated at 150 actively fishing vessels, and the IMG fleet 18 actively fishing vessels. Not always known when vessels are active and inactive.
Fishery geographic extent	Two distinct fleets with different fishing footprints. Limited GPS coordinates for observed trips from the IMG fleet (5-32 GRT). Landing site/port surveys (Alfaro-Shigueto et al. 2007; Alfaro-Shigueto et al. 2011; Alfaro-Shigueto et al. 2018) and bycatch location reported from the HF two-way radio outreach program (Alfaro-Shigueto et al. 2012).	Focus group discussion mean estimates for the maximum geographic extent for the San Jose IG fleet was 1200 km² in summer and 3700 km² in winter, and the IMG fleet 27000 km² in summer and 31500 km² in winter.
Target catch	Surface drift net: target sharks, rays mahi mahi, bonito, swordfish <i>Xiphias gladius</i> , flathead grey mullet <i>Mugil cephalus</i> , Peruvian silverside <i>Odontesthes regia</i> ; Bottom set net: target sharks, rays flounder, lobster (Alfaro-Shigueto et al. 2010; Guevara-Carrasco & Bertrand 2017).	Target catch behaviour in relation to turtle bycatch reduction technologies (e.g., gillnet illumination). Impact that shifting target species would have on turtle bycatch.
Fishing seasons	Two main seasons in San Jose. Summer usually spans 3-months December-February and winter March-November. Lambayeque catch reports indicate summer-like fishing conditions span a longer period (Carrasco & Bertrand 2017).	San Jose winter fishing season is June-November and the summer fishing season as December-May.
Market type	The nearest fish market is in Santa Rosa located 21 km from San Jose (Figure 1). Catch is sold locally and domestically. Refrigeration trucks present daily.	Lack of oversight as to where all the catch taken using San Jose gillnets e.g., local in San Jose, Chiclayo (largest nearby city), wider Lambayeque region, other regions, international markets.

Table 3.5. Observed sea turtle captures per trip in the San Jose inshore-midwater gillnet fleet from August 2007–May 2019. CI = confidence interval. Mortalities and capture releases with injury are provided in text (see Supplementary Materials for the table format).

Turtle species	n	Per Trip (n=461)			
		Mean	SD	Min 95% CI	Max 95% CI
Green	329	0.71	1.98	0.53	0.89
Leatherback	7	0.02	0.12	0.01	0.03
Olive ridley	35	0.08	0.46	0.04	0.12
Unidentified	8	0.02	0.21	0.00	0.04
Total turtle captures	379	0.82	2.10	0.63	1.01

(n=8; Figure 3.1). Turtles released alive without visible injury made up 62% of the 379 captures. Live releases with injuries made up 28%. Mortalities 8% of captures (see Supplementary Materials). Capture per unit effort across trips (n=461) was significantly different between species (one-way analysis of variance; $F(2,1380) = 49.73$, $p < 0.001$). Green and olive ridley turtle capture rates per trip differed significantly ($p < 0.05$; Tukey post hoc tests), but leatherback and olive ridley turtle capture rates were not significantly different at the trip level (Table 3.5).

3.3.2 Risk assessment

3.3.2.1 Inshore-midwater gillnet fleet

I ranked the leatherback and green turtle RMU (Pacific East) populations as subject to an extreme risk from the San Jose IMG fleet over the next five years, and the olive ridley turtle RMU (Pacific East) as subject to a major risk given the current management measures in place (Table 3.6). No catch restrictions or effort limits exist, but five of the estimated 18-28 vessels comprising the IMG fleet (Table 3.4) were using light-emitting diodes (LEDs) on their nets – a form of at sea minimisation. This equates to illuminated nets on 27% of IMG vessels in winter and 18% of vessels in summer. In Sechura Bay, located approximately 150 km north of San Jose (Figure 3.1), controlled gear trials were implemented testing the turtle mitigating potential of LEDs on gillnets. The study found LEDs reduced green turtle bycatch by 64.7% with no reduction in target catch (Ortiz et al. 2016). While no fishery- or region-specific data on the effect that LEDs have on leatherback and olive ridley captures exist, anatomical, physiological and

behavioural studies show leatherback turtles also have a sensitivity to ultraviolet (UV) wavelengths (Wang et al. 2013; Wyneken et al. 2013). Over the assessment period, I assumed a 64.7% reduction in captures for each of the turtle species captured, across 27% of the IMG fleet in winter and 18% in summer. Workshops training fishers on safe handling and release of captured turtles are conducted in San Jose by the not-for-profit *ProDelphinus*. I estimated a small increase in post-capture survival rates of turtles based on known sea turtle survival rates following capture in gillnets (Epperly et al. 2004; Snoddy & Williard 2010).

The IMG fleet concentrates fishing effort nearshore between Lobos de Tierra in the north, Salaverry in the South, and west to Lobos de Afuera (Figure 3.1) – this shows fishing effort occurs in areas where each turtle species occurs. The fleet covers less than 5% of each turtle species' Pacific East RMU distribution. Using the lines of evidence (Table 3.4 and Table 3.7), all four levels of consequence (i.e., some level of turtle bycatch) were plausible for each turtle species, but with different levels of likelihoods (Table 3.6).

The olive ridley turtle is the most abundant sea turtle in the world (Wyneken et al. 2013). The Pacific East RMU population numbers approximately 1,500,000 individuals (Eguchi et al. 2007; Wallace et al. 2010a). The population has an increasing trend in the short-term but a predicted decreasing trend in the long-term (Wallace et al. 2010a). The olive ridley turtle species exhibit both solitary and "arribada" nesting; the latter is a behaviour unique to the *Lepidochelys* genus where large groups of females nest synchronously at a nesting site (Richard & Hughes 1972; Wyneken et al. 2013). The observer data did not record any olive ridley captures further offshore than Lobos de Tierra indicating the potential for a more inshore distribution within the San Jose gillnet fishery's geographic extent (Figure 3.1). Vessels number 18-28 (Table 3.4) and fishing trips average 7.5 days (see Supplementary Materials). Drawing on the lines of evidence of fishing effort, and a capture per trip rate of 0.08 (of which mortality rates were 21%, and capture release with injury rates were 25%), the annual mortality rates of olive ridley turtles in the IMG fleet are likely in the tens rather than the hundreds. This pattern highlights a 'moderate' consequence (C2) signifying the bycatch impact from the IMG fleet is at a maximum level of acceptability, is 'likely' (L4) to occur during the assessed period (Table 3.6). The evidence does not suggest that a 'severe' (C3) consequence level of impact 'may occur' (L3) or is 'expected' (L4).

Table 3.6. Results of the consequence × likelihood qualitative ecological risk assessment. Likelihoods (as indicated by ×'s) for each of the consequence levels for the bycatch (mortality following incidental capture) of leatherback turtle *Dermochelys coriacea*, green turtle *Chelonia mydas*, and olive ridley turtle *Lepidochelys olivacea* in the San Jose inshore gillnet fleet, and the San Jose inshore-midwater gillnet fleet. The final risk level is based on the highest risk score from multiplying consequence and likelihood scores. Turtle stock size was assessed at the East Pacific RMU scale. Fleet sizes were defined by the geographic maximum extent calculated. Consequence levels and associated likelihoods are based on the lines of evidence for biological factors, potential overlap/susceptibility, simple catch and effort, current management restrictions, effective effort levels, social use, and cultural values (see Table 5). Consequence levels: 1 = minor, 2 = moderate, 3 = major, 4 = extreme. Final risk levels: MI = minor, MO = moderate, MA = major, EX = extreme.

Source of risk	Turtle species	Consequence level	Remote	Unlikely	Possible	Likely	Risk score	Final risk level
			L1	L2	L3	L4		
Inshore-midwater gillnet fleet	Leatherback	C1	×				1	EX
		C2	×	×	×		6	
		C3	×	×	×	×	12	
		C4	×	×	×	×	16	
	Green	C1	×				1	EX
		C2	×	×	×		6	
		C3	×	×	×		9	
		C4	×	×			8	
	Olive ridley	C1	×	×	×	×	4	MA
		C2	×	×	×	×	8	
		C3	×	×			6	
		C4	×				4	
Inshore gillnet fleet	Leatherback	C1	×	×	×		3	EX
		C2	×	×	×		6	
		C3	×	×	×	×	12	
		C4	×	×	×	×	16	
	Green	C1	×	×			1	EX
		C2	×	×	×		6	
		C3	×	×	×		9	
		C4	×	×	×		12	
	Olive ridley	C1	×	×	×	×	4	MA
		C2	×	×	×	×	8	
		C3	×	×			6	
		C4	×				4	

Table 3.7. Summary table of the information used to complete the risk assessment for turtle species captured in the San Jose gillnet fishery. Three species of sea turtle are known to be captured in our case-study fishery, the leatherback turtle *Dermochelys coriacea*, green turtle *Chelonia mydas*, and olive ridley turtle *Lepidochelys olivacea*. Text in **bold** highlights collected data filling identified knowledge gaps.

Species of concern	Lines of evidence		
	Biological factors	Susceptibility to the fishery	Socioeconomic outcomes
Leatherback turtle <i>Dermochelys coriacea</i>	<i>Size:</i> Up to 215 cm <i>Weight:</i> Up to 900 kg The average lifespan in the wild: 45 years Sexual maturity: ~16 years <i>Fecundity:</i> One female may lay up to nine clutches in a breeding season. Average clutch size is approximately 110 eggs, with up to 85% of these in a viable state (Eckert 2012). <i>Habitat:</i> Primarily pelagic (open ocean) dwelling. Females require sloped sandy beaches for laying clutches of eggs. <i>Nesting sites:</i> The East Pacific population nests along the Pacific coast of the Americas from Mexico to Ecuador. No established nesting sites for leatherback turtles are present in Peru. The closest nesting area to San Jose is located in Ecuador (Eckert 2012). <i>East Pacific RMU geographic extent:</i> From the tip of Baja California Mexico south to Chile, out to 135W (Wallace et al. 2010a). <i>East Pacific RMU population size:</i> ca. >200 (Wallace et al., 2010a; Wallace et al., 2013). Preliminary data shows a small percentage of leatherback turtles present in the waters of the Pacific East Regional Management Unit (RMU) are from the Pacific West RMU (P. Dutton, pers. comm.). <i>Population trend (East Pacific RMU short and long-term/Global):</i> decreasing short- and long-term (Wallace et al. 2010a); global population decreasing.	<i>Catchability in the fishery:</i> BPUE per trip in San Jose is 0.02 ± 0.21 (mean $\pm$ SD). Seven recorded captures (all released alive) in San Jose inshore-midwater fleet from observer data between 2007-2017 (Table 3.5). <i>Distributional overlap:</i> 100% of total area within boundaries of the fishery (Figure 3.1). High leatherback captures in coastal gillnet locations near Salaverry port, south of San Jose (Alfaro-Shigueto et al. 2007; Alfaro-Shigueto et al. 2018). <i>Management restrictions:</i> poor – few restrictions are in place to support a reduction in leatherback turtle bycatch. Five inshore-midwater gillnet vessels are using LED lights and remote electronic monitoring systems as part of a trial community cooperative with a local not-for-profit (Ortiz et al. 2016; Bartholomew et al. 2018). <i>Overlap in the effective fishing effort:</i> Most of the fishing effort is concentrated in the first 25km of ocean from the shore (Figure 3.1). Few sets have been recorded further offshore than Lobos de Afuera. Management effectiveness and compliance: Unknown	<i>Social use:</i> Retention for human consumption is known to occur. Of the 133 leatherback turtles captures recorded in Peru's SSF 1985–2003, 58.6% were retained for consumption (Alfaro-Shigueto et al. 2007). <i>Value:</i> Unknown <i>Target market if sold:</i> If eaten, turtles are usually consumed on board the vessel or at home after a fishing trip. Black markets provide a platform for the sale of the illegal product (Quiñones et al. 2017). <i>Cultural values:</i> Turtle meat was historically eaten in Peru (Aranda & Chandler 1989; Morales & Vargas 1996).

Green turtle <i>Chelonia mydas</i>	<i>Size:</i> Up to 150 cm <i>Weight:</i> Up to 315 kg The average lifespan in the wild: 80 + years Sexual maturity: ~25 years <i>Fecundity:</i> Nesting occurs nocturnally at 2, 3, or 4-year intervals. Max nine clutches within a nesting season (average 3.3). <i>Habitat:</i> Shallow waters (except when migrating) inside reefs, bays, and inlets (Seminoff et al. 2015). <i>Nesting sites:</i> Nesting occurs in more than 80 countries. The southernmost nesting sites for the species have been reported in Los Pinos, Tumbes, northern Peru, approximately 466km from San Jose (Forsberg et al. 2012). <i>East Pacific RMU geographic extent:</i> Los Angeles south, sweeping down the coast of Chile and the Eastern Tropical Pacific out to 145 West (Wallace et al. 2013). <i>East Pacific RMU population size:</i> 3750 (Wallace et al. 2010a). <i>Population trend (East Pacific RMU short and long-term/Global):</i> Increasing short-term (Wallace et al. 2010a; Seminoff et al. 2015), decreasing long-term; global population decreasing.	<i>Catchability in the fishery:</i> BPUE per trip in San Jose is 0.71 ± 1.98 (mean $\pm$ SD). 329 captures in San Jose inshore-midwater fleet from observer data between 2007-2017 (Table 3.5). <i>Distributional overlap:</i> 100% of the total area within the boundaries of the fishery (Figure 3.1). Reports of high capture rates in gillnets in northern fishing locations during key information interviews in San Jose (Alfaro-Shigueto et al. 2018). <i>Management restrictions:</i> poor – few restrictions are in place to support a reduction in green turtle bycatch. See leatherback section for further details. <i>Overlap in the effective fishing effort:</i> Majority of fishing effort is concentrated in the first 25km of ocean from the shore (Figure 3.1). Few sets have been recorded further offshore than Lobos de Afuera. Management effectiveness and compliance: Unknown	<i>Social use:</i> Human consumption, but likely not for their eggs unless northern most nest sites in Tumbes, Peru are impacted. <i>Value:</i> Unknown <i>Target market if sold:</i> See leatherback target market if sold. <i>Cultural values:</i> See leatherback cultural values.
Olive ridley turtle <i>Lepidochelys olivacea</i>	<i>Size:</i> 60-75 cm <i>Weight:</i> Up to 45 kg The average lifespan in the wild: 50 years Sexual maturity: 10-18 years <i>Fecundity:</i> Commonly nest in successive years, 1-3 times per season, with ~ 100-110 eggs per clutch <i>Habitat:</i> Worldwide in tropical and warm oceanic and neritic waters. <i>Nesting sites:</i> Nesting occurs in nearly 60 countries worldwide. The southernmost nesting sites for the species have been reported in El Nuro, Piura, Peru (Kelez et al. 2009), approximately 375km from San Jose. <i>East Pacific RMU geographic extent:</i> Baja California Sur Mexico to southern Peru, the eastern Pacific and northwest of Hawaii (Wallace et al. 2010a). <i>East Pacific RMU population size:</i> 5000 (Wallace et al. 2010a). <i>Population trend (East Pacific RMU short and long-term/Global):</i> Stable short-term - Population in East Pacific RMU may have increased since the 1990s (Eguchi et al. 2007; Wallace et al. 2010a), long-term decreasing; global population decreasing.	<i>Catchability in the fishery:</i> BPUE per trip in San Jose is 0.08 ± 0.46 (mean $\pm$ SD). 35 captures in San Jose inshore-midwater fleet from observer data between 2007-2017 (Table 3.5). <i>Distributional overlap:</i> ~75% of the total area within the boundaries of the fishery (Figure 3.1). Reports of high capture rates in gillnets in northern fishing locations in key information interviews in San Jose (Alfaro-Shigueto et al. 2018). <i>Management restrictions:</i> poor – few restrictions are in place to support a reduction in the incidental take of green turtle. See leatherback section for further details. <i>Overlap in the effective fishing effort:</i> Most of fishing effort is concentrated in the first 25km of the ocean from the shore (Figure 3.1). No recorded olive ridley captures were recorded further offshore than Lobos de Tierra (Figure 3.1). Management effectiveness and compliance: Unknown	<i>Social use:</i> See green turtle social use. <i>Value:</i> Unknown <i>Target market if sold:</i> See leatherback target market if sold. <i>Cultural values:</i> See leatherback cultural values.

The green turtle RMU (Pacific East) population has been estimated at 3750 individuals (Wallace et al. 2010a). The population trend is projected upward in the short-term, but downward in the long-term (Wallace et al. 2010a). I found green turtle presence was likely throughout the fleet's geographic extent (Figure 3.1). Green turtle capture rates are high at 0.71 per trip (Table 3.5). The observed mortality rate of green turtle bycatch was 7% and captures released with injury at 30% (see Supplementary Materials). This data shows bycatch mortality rates of green turtles may have been occurring in the tens of turtles, to low hundreds of turtles per annum in the IMG fleet (Alfaro-Shigueto et al. 2010; Alfaro-Shigueto et al. 2018). These patterns imply that negative fishing impact from the IMG fleet will occur to more than a few individuals in most years over the assessed period (C1). The likelihood of this consequence occurring was ranked as 'remote' (L1). The capture and inferred bycatch rates could be consistent with capture or impact occurring at the maximum acceptable level (C2), or that recovery 'may be affected' or 'further declines are generated' (C3 or C4). The estimated short-term rising trend in the Pacific East RMU population of green turtles (Wallace et al. 2010a) in combination with existing IMG fleet management measures to mitigate turtle bycatch imply that further declines to the RMU population from this fleet (C4) were not 'likely' (L4) or 'possible' (L3) but 'unlikely' (L2) over the assessment period. I assigned this consequence of impact an indicative probability of 3–9%. Both the consequence levels of 'stock recovery impact' (C3) and the 'maximum level of acceptable bycatch occurring' (C2), were 'possible' (L3) as further data were not available to reduce uncertainty.

Leatherback turtle capture rates were the lowest of the turtle species assessed at 0.02 per trip (Table 3.5), but this BPUE could still equate to >10 leatherback turtle captures per annum. The observed mortality rate of leatherback turtle bycatches was 14% (1/7). The remaining six captures were released alive without injury (see Supplementary Materials). Leatherback turtle presence was considered 'possible' through the IMG geographic extent (Figure 3.1). The leatherback turtle's Pacific East RMU population (ca. >200) has an estimated decreasing mean growth rate of -0.156 (Mazaris et al. 2017). These data suggests that even a low amount of fishing-related mortality from the IMG fleet (i.e., only a few individuals per year) could 'likely' (L4) result in further population declines (C4) and increase the chances of extinction of the Pacific East RMU population of leatherback turtles (Spotila et al. 2000).

3.3.2.2 Inshore gillnet fleet

I ranked the recovery of the leatherback, and green turtle, East Pacific RMU populations, as subject to ‘extreme’ risk from the San Jose inshore gillnet fleet, and the olive ridley turtle East Pacific RMU as subject to ‘major’ risk given the current management measures in place (Table 3.6). The IG fleet covers an area of 1200 km² in summer and 3700 km² in winter (Table 3.4). The geographic extent of the IG fleet is considerably smaller than the IMG fleet (Figure 3.1), but IG vessel numbers are higher than the IMG fleet. During my 2017 winter field season, I along with my research assistants, recorded one-hundred and fifty inshore gillnet vessels fishing in San Jose – this represents a tripling in fleet size since 1996 (Escudero 1997). Unlike the IMG fleet, no fishery observer data for turtle captures exist for the IG fleet. This increased uncertainty when estimating the likelihood of consequences (Fletcher 2014).

A significant overlap between the IMG and IG fleets exists (Figure 3.1). Captures of leatherback, green, and olive ridley turtles in the IMG fleet have been recorded within the geographic bounds of the IG fleet (Figure 3.1). This data shows that turtle capture in the IG fleet is also probable. However, with only this data, the captures of sea turtles in the IG fleet remain unknown. Further insight can be gained from shore-based surveys investigating sea turtle bycatch in coastal fisheries across Ecuador, Peru, and Chile between August 2010 and March 2011 (Alfaro-Shigueto et al. 2018). San Jose, Lambayeque, Peru, was a survey site. In San Jose, 44 respondents, across both the IMG and IG fleets, acknowledged turtle bycatch in their gillnets. Of these 44 respondents, 43.2% reported green turtle bycatch, 25% leatherback turtle bycatch, and 20.5% olive ridley turtle bycatch (Alfaro-Shigueto et al. 2018). This pattern of data suggests that all the levels of consequence are ‘possible’ for green, leatherback, and olive ridley turtle species (Table 3.1). While the geographic extent of the fishing fleet is small compared to each species wider East Pacific RMU distribution, the high vessel number, inshore distribution of the turtle species overlapping with the IG fleet’s geographic extent (Figure 3.1), and high uncertainty resulted in ‘possible’ (L3) and ‘likely’ (L4) likelihoods for most of the consequence rankings (Table 3.6). A final risk level of extreme or major is unacceptable unless further management actions are undertaken (Fletcher 2014). This assessment highlights the need for further management actions in the San Jose gillnet fishery if the proposed target of reducing turtle captures by 15% every year for five years while maintaining total catch weight is to be achieved.

Table 3.8. Potential management measures for mitigating sea turtle captures/mortalities in San Jose's small-scale gillnet fishery. Here I limit potential management measures to thirteen. Additional management strategies could be evaluated in successive evaluation rounds. An effort was made to ensure representation of management strategies to address the negative anthropogenic impact that occurs throughout the life cycle of each of the sea turtle populations of management concern.

Mitigation hierarchy step	Management measure	Examples of use in existing fisheries management/policy, or, examples of use in a similar fisher. Effects on sea turtles are highlighted	Key references
Avoid - ensure spatio-temporal overlap does not occur; <i>E_B</i>	Gear trade-in initiatives swapping all gillnets to lobster pots or trolling gear (a form of handline fishing).	In 2007, a gillnet gear trade in initiative was trialled with gillnet fishers in Trinidad, where 3000 entanglements of leatherback turtles were reported in the year 2000 (Eckert & Eckert 2005; Lee Lum 2006). Fishers were provided training in the use of trolling gear (a form of handline fishing). At the conclusion of the 2007 field tests, fishers were presented with the results of the experiments and asked about their willingness to try new these new methods. Average daily trolling daily income was calculated at \$406 (Trinidadian dollars) with no sea turtle bycatch, relative to \$334 (Trinidadian dollars) per day with traditional nets. 90% of fishers said they would be willing to switch to trolling (Eckert et al. 2008).	Eckert et al. 2008; Eckert & Eckert 2005; Lee Lum 2006
Minimise - limit probability of capture in times/ places of overlap; <i>B_{DOA}</i>	No-take MPA extending the fishing restriction in place around the islands of Lobo de Tierra and Lobo de Afuera from 5 to 15 nautical miles offshore the islands (a potential turtle hotspot), all year.	The Peruvian government implemented national marine reserves around 30 offshore islands including two, Lobo de Tierra and Lobo de Afuera, located 100 km and 85km km from San Jose, respectively. National marine reserves in Peru only have an equivalent protection status to IUCN category VI Protected Areas, offering limited protection. A prohibition of bottom trawling exists that extends for 5 nautical miles from the islands' shoreline.	UNEP-WCMC & IUCN 2016; IUCN 2010
	A temporal gillnet ban (August –November) with gear switching to lobster potting or trolling during the gillnet ban period every year.	The Pacific Leatherback Conservation Area is a 250,000-square mile marine protected area off the California coast that is enforced during 3 months of the start of August to end of October when leatherbacks are present, shutting off all fishing including the California large-mesh drift gillnet fishery. Consideration of the spillover effects that resulted from the Pacific Leatherback Conservation Area is necessary when considering a time-area closure – notably biodiversity loss due to displaced fishing activity, displaced production activity, and trade leakages from an increase in imports to replace the displaced domestic production (Squires et al. 2016).	50 C.F.R. § 660.713; Curtis et al. 2015; Squires et al. 2016

	A dynamic gillnet ban shifting in space and time in relation to turtle movement (enacted with existing and available information).	In an effort to provide information to fill existing data gaps and support bottom-up monitoring of compliance, an information-sharing scheme was started by not-for-profit <i>ProDelphinus</i> in the form of a high-frequency two-way radio outreach program to raise awareness of fishers at sea of bycatch, and to provide them with any requested information using real-time spatial management. Now partnered with the not-for-profit's <i>Asociacion</i> and <i>Pacifico Laud</i> the initiative covers twenty-five ports and extends over 3500 km from Manta, Ecuador to San Antonio, Chile.	Alfaro-Shigueto et al. 2012; Hazen et al. 2018; Squires et al. 2018
	An offshore distance restriction with gillnetting only allowed to occur between 0-2 nautical miles offshore.	No-take marine reserves are established, important conservation and management tools that have proven to have positive responses in far more cases than no differences or negative responses.	Halpern 2003; Lester et al. 2009
	Soak time (effort) restriction of 6 hours per set for the IMG fleet only.	Average soak time for IMG vessels is 14.6 ± 3.9 hours per set. This strategy equates to a rough a halving of the IMG fleet's fishing effort.	Gilman et al. 2010
	Buoyless nets which entail removing the buoys from the float line of the net.	In 136 controlled sets of conventional (control) and buoyless nets (buoys removed from float line), buoyless nets reduced mean turtle bycatch rates by 68% while maintaining target catch rates and composition.	Peckham et al. 2016
	Fixed demersal nets only, surface driftnet ban.	Most gillnets in San Jose are surface drift nets, which take more turtle bycatch than fixed demersal nets. Reductions in bycatch of surface and near-surface swimming turtles would be expected.	-
	Light-emitting diodes (LEDs) on gillnets.	(i) <i>ProDelphinus</i> are running a trial bycatch reduction community co-management scheme in San Jose where participating skipper and crew use LEDs on their nets in an effort to reduce turtle bycatch. (ii) In Sechura Bay, northern Peru, 114 pairs of control and illuminated nets were deployed. The predicted mean catch per-unit-effort of green turtles was reduced by 63.9% in illuminated nets. (iii) Turtle capture rate was reduced by 39.7% in LED illuminated nets while having negligible impacts on target catch and catch value.	Wang et al. 2013; Ortiz et al. 2016
Remediate - limit capture-related mortality	An annual workshop on safe handling and release procedures, which includes the resuscitation of sea turtles (estimates represent	(i) Post-capture, sea turtles that appear lifeless are not necessarily dead. They may be comatose. While turtles returned to the water before they recover from a coma will drown. A turtle may recover on board a boat once its lungs have drained of water.	FAO 2009; Suuronen & Gilman 2019

once caught; BOB, PDV, PDR	mortality reduction rather than encounter reduction).	This could take up to 24 hours. By following best practice handling and resuscitation guidelines unnecessary turtle deaths can be prevented. (ii) <i>ProDelphinus</i> run workshops training fishers on safe handling and release of bycatch turtles in San Jose and other SSF communities along Peru's coastline.	
	Mandatory remote electronic monitoring on vessels to reduce possibility of turtle retention post-capture.	Remote electronic monitoring has been trialed on five San Jose boats with a total of 228 fishing sets monitored. Of these, 169 sets also had on board fisheries observers. The cameras were shown to be an effective tool for identifying elasmobranch catch >90% detection rates, though variable for sea turtles (with 50% positively identified). As well as improving data, remote electronic monitoring has potential to reduce the high rate of illegal consumption of leatherback turtles (Alfaro-Shigueto et al. 2007).	Bartholomew et al. 2018; Suuronen & Gilman 2019 Dutton et al. 2005; Janisse et al. 2010; Milner-Gulland et al. 2018
Offset - compensate for harm caused by residual bycatch mortality; Or)	Bycatch (Pigouvian) tax ¹ that funds turtle nesting site protection e.g., unprotected smaller nesting sites in Peru, Ecuador, Costa Rica, or Mexico (depending on species).	(i) Positive trends have been reported in leatherback turtle populations over decades as a result of nesting site protection and egg relocation (Dutton et al. 2005). (ii) The California drift gillnet industry, in 2004, financed Pacific sea turtle nesting site conservation efforts in Baja California through a voluntary bycatch tax for compensatory mitigation of sea turtle bycatch. The funds were in part driven in an effort to slow further extensive time-area closures (Janisse et al. 2010).	Dutton et al. 2005; Janisse et al. 2010; Milner-Gulland et al. 2018; Squires et al. 2018
	In-kind payment for ecosystem services program where fishers contribute their time, resource, and knowledge for conservation efforts in the San Jose region of Peru (e.g., reporting turtle sightings and captures to the local government science officer, contributing hours to protected green and olive ridley nesting sites in El Nuro, Piura, Peru, or monitoring marine reserves for illegal fishing)	The Kiunga Marine National Reserve Conservation and Development Project is a partnership between the Kenya Wildlife Service (KWS) and World Wildlife Fund (WWF) pays local women to report turtle nests and sightings of nesting turtles to KWS or WWF employees. In exchange, they are paid upon report verification and a payment conditional on hatching success is also made. Nest translocation is high (~70%) because they are located below the high-tide mark or at other risks of depredation (Flintan 2002; Ferraro & Gjertsen 2009).	Flintan 2002; Ferraro & Gjertsen 2009

¹ A bycatch (Pigouvian) tax can be a double dividend tax, acting as both as an offset and minimisation strategy. The tax minimises bycatch by internalising the external costs of bycatch (for both consumers and producers as part of the tax is passed up the supply chain, depending upon the price elasticities of demand and supply). The first dividend is the welfare increase (including conservation) from minimisation through the bycatch tax and the second dividend, and an additional source of welfare increase (including conservation), comes from the offset (Squires et al. 2018).

– this includes adding monitoring efforts to estimate baseline BPUE rates for each turtle species captured in the IG fleet. These BPUE estimates could then be compared to potential management strategies to reduce turtle BPUE in the future. For supplementary analysis, I present summary tables of evidence for the turtle species assessed (see Supplementary Materials).

Based on information obtained from a literature search, I defined a list of potential management measures and categorised them according to the steps of the mitigation hierarchy. This list was refined to 13 potential management strategies during key informant interviews and FGDs (Table 3.8). Management strategies included an avoidance strategy (to reduce E_B of equation 1), eight minimisation strategies (four spatial or temporal area closures and four technology or fishing behaviour changes), two remediation strategies (to reduce BPUE of equation 1), and two biodiversity offsetting strategies (to increase O_T of equation 1).

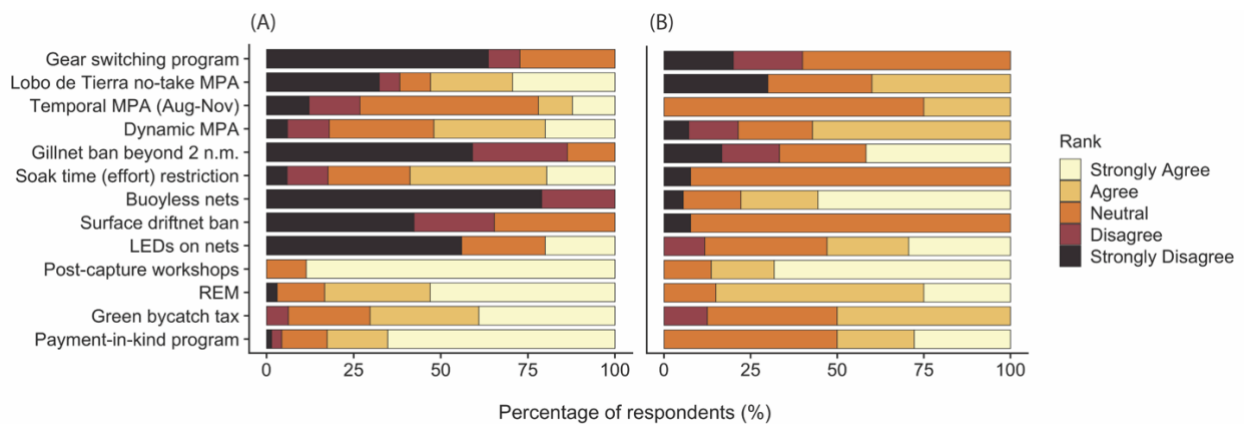


Figure 3.2. Stakeholder preference for the evaluated turtle bycatch reduction strategies. Two focus group discussions (FGDs) were run. FGD (A) concentrated on the inshore gillnet (IG) fleet and comprised 15 respondents, including IG skippers (n=13), a regional government scientist (n=1), and a not-for-profit employee (n=1). FGD (B) focused on the inshore-midwater gillnet (IMG) fleet and comprised 5 respondents, including IMG skippers (n=3) and local not-for-profit employees (n=2). REM = remote electronic monitoring. For full strategy descriptions, refer to Table 3.8.

The respondents in the inshore gillnet fleet’s FGD (comprising 13 of 150 possible IG skippers, a local government scientist, and a local not-for-profit employee) disagreed with more of the potential preventative measures proposed that fell into the avoidance and minimisation steps of the mitigation hierarchy and agreed with more of the compensatory

actions that fell into the remediation and offsetting steps (Figure 3.2). The same trend is present for the IMG fleet's FGD (comprising 3 of 18 possible IMG skippers, and two local not-for-profit employees) but in the this FGD responses were more mixed. Fishers were in strong disagreement with the proposed avoidance strategy of phasing out gillnets in favour of alternative fishing gear such as trolling (a form of handline fishing). The inshore group strongly disagreed with at sea capture reduction technologies such as LEDs on nets or shifting to buoyless nets. Responses to these measures were more distributed for the IMG group. Both groups were in strong agreement with participating in training workshops teaching better handling and release practices for captured turtles and most respondents agreed with the use of remote electronic monitoring on board their vessels (Figure 3.2). Biodiversity offsetting strategies received mixed responses in both FGDs (Figure 3.2).

3.4 The hypothetical effectiveness of management options

In the final section of this study, I illustrate the integration of MSE and the mitigation hierarchy for use in a data-poor management scenario. I explore the performance of three 'management scenarios' that each combines multiple management strategies in a preliminary and qualitative assessment to reduce the risk from the San Jose gillnet fishery posed to the recovery of leatherback, green, and olive ridley turtle populations defined by Pacific East RMUs.

3.4.1 Scenario synopses

Scenario 1 considers the status quo management over ten years between 2020 and 2030. The scenario maintains existing management strategies in the San Jose gillnet fishery and projects an expected level of expansion over ten years. Management strategies include expanding the use of LEDs on nets (minimisation) and remote electronic monitoring (remediation) from the five IMG vessels where these technologies are currently applied to all the vessels in the IMG fleet. Safe handling and release workshops held in San Jose continue (remediation). This scenario does not implement any management measures in the IG fleet (Table A1.2.7).

Scenario 2 takes a protectionist approach to sea turtles. The scenario implements a gear switching program that phases out gillnet for trolling (a form of handline fishing). A quarter of the San Jose fishery is proposed to undergo the gear switch every two and a half years (avoidance). The existing management actions in place in San Jose continue as expected

in scenario 1 (e.g., LEDs on nets would be implemented on IMG vessels that continued to fish during the gillnet phase out-period).

Scenario 3 takes a more incentive-based approach, implementing multiple strategies spanning at sea minimisation, post-capture remediation, and biodiversity offsetting actions. The scenario includes an effort restriction for all vessels operating in the IMG fleet. This limits the gillnet soak time to six hours per day as opposed to the current 14.6 ± 3.9 hours (Alfaro-Shigueto et al. 2010). This equates to a halving of fishing effort across the IMG fleet (minimisation). The scenario also includes a dynamic spatio-temporal marine protected area (MPA) for leatherback turtles (minimisation). This will make use of a local two-way high-frequency (HF) radio program that allows fishers to receive and report real-time information on turtle sightings and captures (Alfaro-Shigueto et al. 2012). The status quo management strategies in scenario 1 are also enacted, but in this scenario, they integrate the IG fleet as well as the IMG fleet (e.g., LEDs on all nets, remote electronic monitoring systems on all vessels and continued implementation of safe handling and release workshops across the San Jose gillnet fishery). To support further population recovery for the turtle populations impacted by our case study fishery, this scenario implements a bycatch (Pigouvian) tax as a biodiversity offset (Table 3.8; Squires et al. 2018). The tax applies to leatherback, green, and olive ridley turtles captured in an eastern Pacific pelagic longline fishery (e.g., Donoso & Dutton 2010). The means to negotiate this tax in practice goes beyond the scope of the hypothetical scenario assessed here, but volunteer bycatch taxes have been implemented by large-scale commercial fishing fleets before (e.g., a turtle bycatch tax through the California Drift Gillnet Fishery funding nesting site protection implemented by the Mexican non-profit organisation *Asumatoma A.C*; Janisse et al. 2010). In the present scenario, funds from the tax support the monitoring of leatherback secondary nesting sites in Costa Rica, where illegal egg harvesting can still occur (e.g., Ostional; Santidrián-Tomillo et al. 2017). Olive ridley turtles also nest in Ostional, Costa Rica, offering the potential for conservation actions at a single site to support the population recovery of two of the three turtle populations incidentally captured by the San Jose gillnet fishery.

3.4.2 Evaluation of scenarios

Scenario 1 ('the status quo') presents the existing management of the San Jose gillnet fishery between 2020 and 2030 (see Supplementary Materials). In this scenario, the turtle bycatch issue is expected to worsen because of a lack of management measures restricting fishing

effort (Figure 3.3). With no effective effort restriction in place (such as a TAC to reduce target fish catch per unit effort; CPUE), the incidental take of sea turtles is expected to increase as the IG fleet grows in vessel number and the San Jose gillnet fishery as a whole expands in geographic extent and fishing effort (Guevara-Carrasco & Bertrand 2017; Castillo et al. 2018). Despite increasing fishing effort (e.g., distance travelled) and fleet number (Table 3.4), we (the authors) projected the target fish CPUE to trend downward in line with historical catch trends for the Lambayeque region of Peru (Guevara-Carrasco & Bertrand 2017). The expansion of existing turtle bycatch mitigation measures trialled in the fishery (LEDs on nets reduce B_{DOA} of equation 2, and remote electronic monitoring and better handling practices reduce P_{DV}) are expected to reduce turtle BPUE rates for individual vessels, and remote electronic monitoring is expected to improve data paucity of turtle capture, bycatch, and consumption rates. Discard rate across a fishery is strongly influenced by shifts in individual human behaviour, so the uncertainty in our projected trend is high (e.g., Smith et al. 2004). We drew on data that indicates LEDs on nets have little impact on the volume of target catch (Ortiz et al. 2016). This was supported by my field observations where I noted that San Jose fishers retain all but the smallest fish species for use and sale at markets – which is supported by regional catch reports (Guevara-Carrasco & Bertrand 2017). This data highlights that current trends in discards are likely to persist under scenario 1. As fishing effort across a larger geographic extent is expected, the impact on habitat and sessile communities is predicted to have a slight upward trend (Figure 3.3).

We predicted that the overall management cost of this scenario would follow an increasing trend because of the expansion of LEDs on nets and remote electronic monitoring across the IMG fleet (Figure 3). Costs supporting our estimate came from price estimates reported from controlled gear trials of LEDs on nets and remote electronic monitoring in the local fishing system (Ortiz et al. 2016; Bartholomew et al. 2018). The IG fleet remains for all intents and purposes, an open-access fishery (cf. Supreme decree N° 018-2010-PE). Despite the ban on new vessel builds, we expect the IG fleet to expand in line with historical trends over the assessment period (Table 3.4). We predicted that an expanding IG fleet would decrease the stability of management across the San Jose gillnet fishery as a whole. The cost per day fished is expected to increase as distance travelled increases, forcing a higher consumption of fuel per vessel. Declines in food, employment security, and fish processing follow declining CPUE estimates (Guevara-Carrasco & Bertrand 2017). We predicted that an increasing IG fleet would drive positive trends in local transport, boat building, and maintenance, but uncertainty

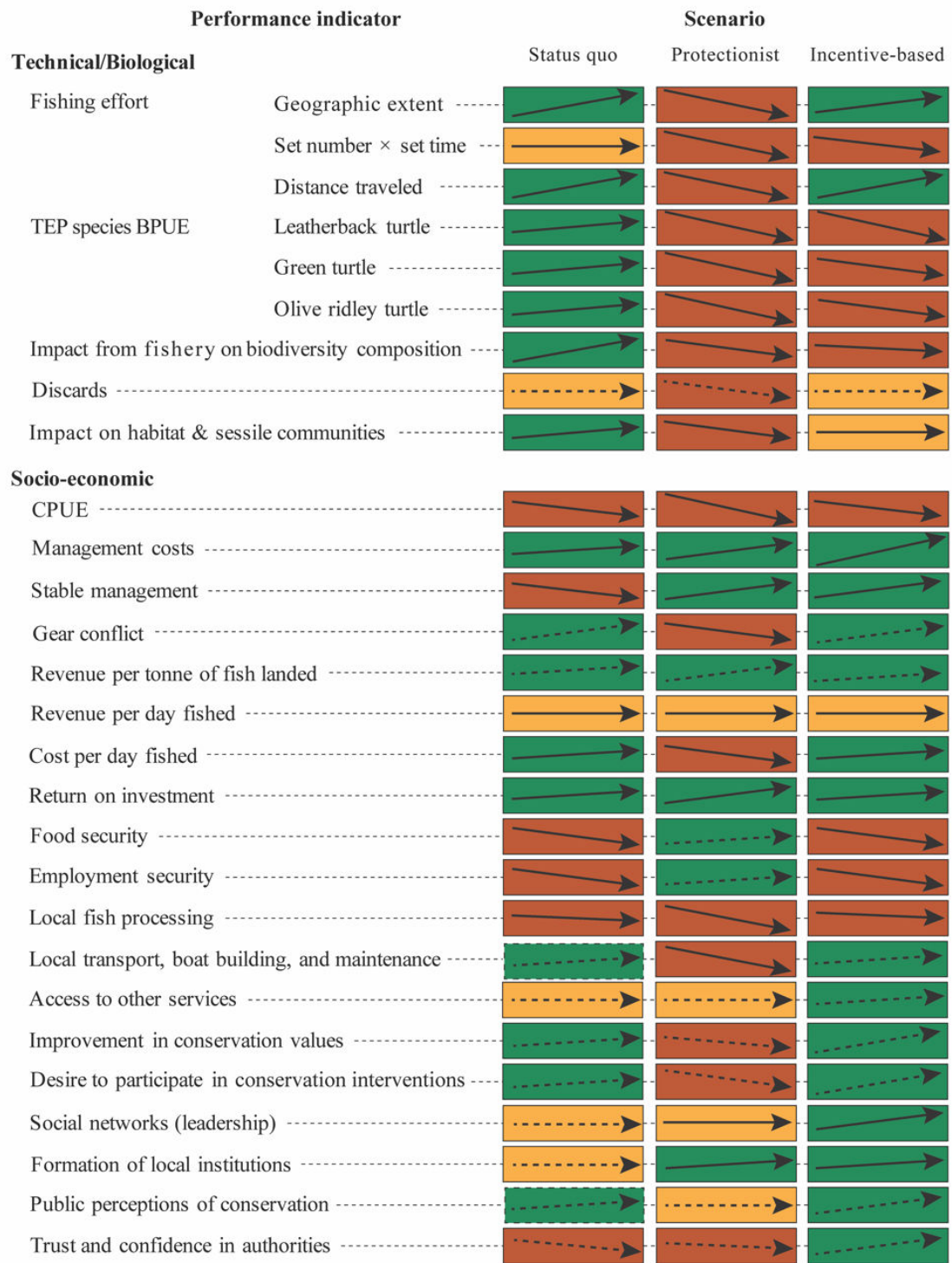


Figure 3.3. Trends over 10 years in indicators for each of the three management scenarios in the San Jose gillnet fishery (based on informed opinion). Green boxes indicate predicted positive trends, yellow boxes indeterminate trends, and red boxes declining trends. Dotted lines represent high levels of uncertainty in the presented trend direction. I present a management summary table with a full list of the management measures contained in each scenario (see Supplementary Materials).

remains high due to the potential to increase enforcement of the ban on new vessels. Access to other services is not well known but predicted to remain stable with high uncertainty. We predicted that the expanding conservation interventions (e.g., LEDs on nets and participatory workshops) would lead to a small improvement in local conservation values. Our survey data shows that IG skippers disagreed with LEDs on nets, but that remote electronic monitoring and training workshops had a stronger agreement (Figure 3.2). We predicted perceptions to improve over time as the existing interventions in the IMG fleet expand to the other fishers in the fleet, but this trend remains highly uncertain and warrants further investigation. Local leadership, the formation of new social organisations (e.g., a new fishing collective), and trust and confidence in authorities are not well known. They are predicted as indeterminate or in decline as the current scenario does little to investigate or manage the improvement of these social conditions.

Scenario 2 ('the protectionist') reduces the negative fishing impact on sea turtles (reduced E_B of equation 1) over the management period (Figure 3.3). We predicted that the area fished would increase as the gillnet fishery continues to expand in effort due to a predicted decrease in target catch following historic catch trends (Guevara-Carrasco & Bertrand 2017). We anticipate management costs to increase as the initiative expands. We considered the possibility of an increase in the gross value of the fish product as handline caught fish can offer a more sustainable consumer choice (e.g., Eckert & Eckert 2005; Eckert et al. 2008), but the uncertainty surrounding consumer interest and willingness to pay for a more sustainably sourced fish product in our case study fishing system remains high. We predicted that conflict between gillnet, trawl, and purse seine fisheries operating in the area would decline as operating gillnets decrease. Transportation, boat building and maintenance resulted in downwards trends as local fish processing and indirect income decline due to trolling bringing in a lower abundance of fish products over gillnets to process (Eckert & Eckert 2005; Eckert et al. 2008). We expect a steady decline in turtle BPUE (reducing B_{DOA} of equation 2) as trolling (handline fishing) takes no, or very little turtle bycatch. Public perception of conservation is highly uncertain. We predicted long-term economic improvement, but the decline in secondary fishery services anticipated to occur in the community over the short- to medium-term may negatively affect this predicted upward trend.

Scenario 3 ('the incentive-based') attempts a more balanced approach to mitigating the negative fishing impact from the San Jose gillnet fishery on sea turtles (Figure 3.3). We predicted fishing effort to continue to increase, but not as rapidly as in scenario 1, because of

the effort restriction halving the allowable set time in the IMG fleet (see Supplementary Materials). We predicted that the effort restriction would lead to an initial decline in CPUE, which would rapidly level out over the remaining management period. We projected declines in turtle BPUE (through a reduction in B_{DOA} of equation 2). We estimated a steeper decline for leatherback turtles over the green and olive ridley turtles because of the dynamic leatherback turtle MPA in this scenario. Our data shows support from most gillnet fishers' in San Jose for the best handling and release practice workshops, so we estimated high compliance and an associated small but measurable increase in post-capture turtle survival rates contributing to the declining turtle BPUE (Figure 3.2; reducing P_{DV} , and potentially P_{DR} in equation 2). It was also assumed that remote electronic monitoring if expanded, would result in broad uptake on gillnet vessels in San Jose (Figure 3.2; reducing P_{DV}). The green bycatch tax can act as a double dividend. The first dividend comes from the tax incentivising fishers in the large-scale pelagic fishery to change their fishing behaviour in favour of mitigating turtle bycatch. The second dividend comes from the funds that the tax produces supporting nesting site protection for leatherback turtles (secondary nesting site) and olive ridley turtles (major nesting site) in Ostional, Costa Rica (Squires et al. 2018). Predicting any meaningful shift in population trends from funding the monitoring of a single nesting site in Costa Rica is difficult over the ten year assessment period (O_T of equation 1). Additional conservation action protecting secondary nesting sites for the East Pacific population of leatherback turtles form an integral part of planning any holistic conservation and population recovery plan for this species (Santidrián-Tomillo et al. 2017). Over more extended periods (e.g., 20+ years) nesting site protection has driven long-term population recovery in several sea turtle populations (e.g., Chaloupka 2003; Balazs & Chaloupka 2004; Dutton et al. 2005; Troëng & Rankin 2005). We predicted that the likelihood of public perception and fishers' desire to be part of management strategies in the future would improve, but this trend is uncertain despite scenario 3 integrating multiple strategies that fishers supported (Figure 3.2). Our predicted trends were strongly influenced by the expected decline in food and employment security expected in San Jose. Scenario 3 has the most diverse suite of bycatch reduction strategies; thus, high management costs for this scenario were estimated (Figure 3.3).

3.5 Discussion

I applied the mitigation hierarchy for fisheries management and marine megafauna bycatch reduction (Milner-Gulland et al. 2018; Squires et al. 2018) to the San Jose gillnet fishery where sea turtle captures are a known conservation issue (Alfaro-Shigueto et al. 2018). Working through the proposed steps of the mitigation hierarchy framework, I characterised our case-study fishery and the species of management concern. The mitigation hierarchy framework helped prioritise research quantifying the fishery's geographic extent across fishing seasons. I identified gaps in fishery-specific turtle capture and bycatch rates, prompting us to calculate capture per trip rates for the turtle species regularly impacted by the IMG fleet. I then assessed the risk from the case study fishery (both IG and IMG fleets) on the turtle species of management concern based on a proposed qualitative turtle bycatch reduction target to contribute to a broader population recovery goal (Fletcher 2014). Drawing on the existing information collated and newly filled knowledge gaps, I compiled a list of thirteen possible management options to reduce sources of anthropogenic-impact posed to the turtle populations of management concern. We (the authors) then used fisher perceptions and a qualitative MSE framework to carry out a preliminary exploration of possible management scenarios by considering estimated trends for a range of biological, social, and economic indicators.

The wide migratory range of the turtle species assessed means that they spend much of their lives in waters or on beaches under other nations' jurisdictions. The highly migratory nature of the sea turtle taxon necessitates a wider international effort to manage transboundary externalities for these species at ecologically relevant levels (Dutton & Squires 2008). While I focused the current study on direct fishing impacts from a single small-scale fishery, the use of the mitigation hierarchy as an overarching framework encouraged consideration of a range of potential management strategies. Management strategies considered ranged from precautionary avoidance and minimisation measures at sea to supporting compensatory actions that seek to mitigate negative impacts from both large-scale pelagic fisheries, and those that occur at terrestrial-based nesting sites (Table 3.8). The framework helped drive simultaneous consideration of biodiversity losses and gains. This, in turn, allowed us to demonstrate the integration of a diverse set of management processes and tools to achieve a specific, qualitative target. This integration demonstrates how actions are undertaken across a wide variety of fisheries, and associated management structures might sum together to

evaluate progress towards high-level population recovery goals for depleted populations of marine megafauna species.

I supplemented the ERA using the turtle capture rates calculated for the IMG fleet, and existing research investigating turtle captures in Peru's coastal gillnet fisheries (e.g., Alfaro-Shigueto et al. 2007; Alfaro-Shigueto et al. 2018). Our analysis shows that the fishing impact from two gillnet fleets, which launch from a single port, could generate further declines of the Pacific East RMU populations of green and leatherback turtles (Table 3.6). San Jose is one of the major gillnetting ports in Peru but comprises only one of 106 landings sites or ports along the country's coastline (Castillo et al. 2018). While this assessment remains qualitative, it highlights the immediate need for additional management action to address the risk of local extinction for the Pacific East RMU leatherback turtle population (Spotila et al. 2000; Mazaris et al. 2017).

The integration of the qualitative MSE process with the mitigation hierarchy framework provided a preliminary evaluation of potential management scenarios incorporating a mix of turtle bycatch reduction strategies in a data-poor fishery. The assessment of how a diverse range of biological, technical, and socioeconomic indicators might change through time allows for trade-offs between management goals to be transparently assessed. The trends estimated in the predictive performance indicators demonstrated that further management action is necessary to mitigate the negative impact on sea turtle populations from the San Jose gillnet fishery. The results also demonstrated that none of the three bycatch reduction scenarios presents a straightforward management picture. We predicted a wide variety of biological, economic, and social shifts across the three management scenarios evaluated. Our results provide some insight into how a range of management measures aimed at reducing turtle captures and mortalities could impact fishers, the wider San Jose community, and indirectly on biodiversity. However, based on our available data, the uncertainty in many of the predicted trends was high, particularly concerning the social indicators (Figure 3). Our results highlight the need for further integrating natural and social science in marine ecosystem-based management research (Alexander et al. 2018a).

In several instances, it was easy to predict indicator trends under one of the three management scenarios evaluated (e.g., expecting green turtle BPUE in gillnets to decrease across the San Jose gillnet fishery as vessels switched from gillnets to handline trolling –

scenario 2). In most cases, predicting the trends was difficult and uncertain, based on the data available. We required an iterative process where the project team (the authors) assessed conflicting inputs to come up with the best guess of the likely trends (Smith et al. 2004). The assessment combined trends across the two gillnet fleets (i.e., IMG and IG), with weightings or emphasis applied to each fleet based mainly on the project team's knowledge of the fishery and the collated and collected data. We found that emphasis on any particular input (i.e., the efficacy of the proposed dynamic spatio-temporal MPA for leatherback turtles) often had a sizeable influence on the trajectory of the trend in the indicator.

The varying experiences and personal biases each member of the project team brought to the assessment meant that several different trends in an indicator could result depending on an individual's interpretation. We undertook an iterative evaluation process aimed to address any difference in opinion. These web-based discussions allowed team members to highlight differences in interpretation. Comprehensive face-to-face workshops guided using structured question protocols and feedback would have improved the project team's ability to address different interpretations (Valverde 2001; Burgman et al. 2011). The project team comprised representatives from academia, government, and a not-for-profit organisation. We acknowledge additional bias in the overall experience of the group towards a conservation science and fisheries science background. Recognition that these biases may influence the qualitative assessment is vital and points to the importance of seeking a diverse range of stakeholder inputs across multiple sectors (e.g., industry, local community members, local government, not-for-profit organisations; Smith et al. 2004). Our experience of undertaking the assessment highlights the necessity for a quantitative evaluation of management scenarios – this could be a mid-term goal for supporting effective mitigation of turtle captures and mortalities in the San Jose gillnet fishery (e.g., Smith et al. 2004; Fulton et al. 2011a).

Numerous management options could integrate under the umbrella of the proposed mitigation hierarchy framework. While I made every attempt to include consideration of management strategies that addressed the negative anthropogenic impact that occurs throughout the life cycle of each of the sea turtle populations of management concern, I did not evaluate many fishery management strategies. For example, implementation of a TAC on target fish species is a primary management mechanism in many fishery management frameworks (Gordon 1954; Karagiannakos 1996; Marchal et al. 2016). I decided not to include a TAC in any of the management scenarios because setting TACs for multiple, individual target species within a mixed-stock fishery must be carefully evaluated (Squires et

al. 1998). Such an evaluation went beyond the scope of this study. Instead, I chose to include a simple effort restriction as part of scenario 3, in the form of halving the soak time within the IMG fleet. However, it should be noted that the evaluation of proposed TACs for multiple species in a qualitative MSE process is achievable (Smith et al. 2004; Dichmont & Brown 2010).

In collating and collecting information about the San Jose gillnet fishery and bycatch species group of management concern, case-specific issues arose. For example, 33% of San Jose gillnet fishers, whom self-reported turtle captures also noted that they consume turtles (Alfaro-Shigueto et al. 2018). Supporting these findings is a report of 133 leatherback turtles caught between 2000 to 2003 off the coast of Salaverry (Figure 3.1). Of these captured leatherbacks, 41.4% were released alive, and 58.6% were retained for human consumption (Alfaro-Shigueto et al. 2007). These data highlight the need for an intervention focused toward shifting social norms and cultural values away from the consumption of turtle meat and towards alternate food sources. This could potentially be integrated as an offsetting measure (e.g., campaigns to engender pride in conserving turtles funded by a bycatch tax). Such an approach could be supported by compliance and monitoring in the form of the proposed expansion of remote electronic monitoring devices (Figure 3; Bartholomew et al. 2018).

I classified only one avoidance management strategy (a gear trade-in initiative swapping all gillnets for lobster pots or trolling gear), out of the thirteen management strategies evaluated. When we developed the theory for applying the mitigation hierarchy to fisheries management and bycatch mitigation, any spatial, temporal, and spatio-temporal area closures were classified within the avoidance step of the mitigation hierarchy (Milner-Gulland et al. 2018). Equation 1 of the mitigation hierarchy stipulates that avoidance measures ensure no spatio-temporal overlap occurs between the impacting risk and the species unit of management concern, thereby reducing E_B (Milner-Gulland et al. 2018). Thus, true avoidance measures require that the impacting fishing activity in question does not overlap with the bycatch of management concern (or has a very low likelihood of occurring; Booth et al. 2020). Because of this, spatio-temporal area closures only act as an avoidance measure if they are large enough or dynamic enough to ensure that fishing impact on the assessed unit of the species of management concern does not occur. For highly migratory marine megafauna such as sea turtles, this means that small spatio-temporal closures may displace the fishing impact to areas where turtles may still be located, thus creating a marginal benefit rather than

ensuring the fishing impact is avoided (Halpern et al. 2004; Agardy et al. 2011). Thus, consideration must be given to the size of the proposed spatio-temporal closure in regards to the size of the assessment unit for the species of management concern. Only following this consideration should management measure be classified in the mitigation hierarchy accordingly.

I identified and filled several knowledge gaps in the current analysis, but other knowledge gaps present more substantive uncertainties and a more comprehensive data gathering process. For example, during the qualitative MSE, we (the authors) had limited understanding of how the proposed management strategies would perform in the case-study system (except for net illumination and remote electronic monitoring; Ortiz et al. 2016; Bartholomew et al. 2018). Several trends estimated in the qualitative MSE were more uncertain as a result (Figure 3). In data-poor fisheries management situations such as the current study, it is often necessary to draw on elicited knowledge from fishers and local practitioners to support evaluations. Structured elicitation methods such as the IDEA protocol offer robust frameworks to reduce cognitive biases and more accurately quantify uncertainty (Hanea et al. 2016b). Elicited data can then be used with fishery-specific costs of management strategy implementation, alongside consideration of the social implications of implementation.

Finally, a fully quantitative application of the mitigation hierarchy (equation 1) would also require an understanding of the relationship between population growth rates and bycatch rates. This was not achievable in our case study and will be challenging for many fisheries and species, particularly those in data-poor situations. As such, targets based on population growth may need to be the ‘gold standard’, with more realistic measurable targets, such as those based on total catch or BPUE, used in the interim.

3.6 Conclusions

I presented a case study application of the mitigation hierarchy to evaluate management options to mitigate sea turtle captures and reduce bycatch in a small-scale gillnet fishery in northern Peru. The overarching conceptual framework provided by the mitigation hierarchy helped integrate a range of fisheries management processes towards a fishery-specific quantitative target that feeds into a more comprehensive goal for biodiversity (Milner-Gulland et al. 2018; Squires et al. 2018). In data-poor fisheries like our case study, such goals remain

aspirational, yet this framing clarifies how local-scale management action can translate to higher-level goals for biodiversity. The proposed framework supported explicit consideration of uncertainties and highlighted future areas of research before implementing a more comprehensive assessment of management strategies in the future. The mitigation hierarchy's step-wise precautionary approach towards biodiversity encouraged a more holistic appraisal of management actions to address the negative fishing impact to sea turtles from the San Jose gillnet fishery. The framing of management options within the context of the hierarchy helped with consideration of preventative and compensatory measures throughout the life cycle of each turtle species of management focus. Integrating the mitigation hierarchy framework with MSE offers potential, as both qualitative and quantitative assessments can be undertaken, catering to a full suite of potential fisheries. It also demonstrates how the mitigation hierarchy can add value to existing methods and procedures established within existing fisheries management processes. In identifying and filling key knowledge gaps and considering the socioeconomic implications of a diverse suite of management strategies, the mitigation hierarchy shows the potential for supporting effective fishery-specific solutions that translate to aspirational national and international biodiversity goals.

Chapter 4

Evaluating Elicited Judgements of Turtle Captures for Data-poor Fisheries Management

A shorter version of this chapter has been accepted for publication pending minor corrections as:

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4.1 Introduction

The incidental capture and subsequent mortality (*bycatch*) of vulnerable marine species, such as turtles, is a known conservation issue in many small-scale fisheries around the world (Peckham et al. 2007; Alfaro-Shigueto et al. 2018). The extent to which species of conservation concern are taken as bycatch, however, remains a major knowledge gap – data paucity having been identified as one of the key challenges to address in the management of the small-scale fisheries sub-sector (FAO 2018). Small-scale fisheries encompass traditional, low-technology, low-capital fishing methods. A single small-scale fishery can comprise a diverse array of vessels (often of small but varying sizes), participants, locations, resource and gears (Khalil et al. 2017). This heterogeneity can make gathering comprehensive empirical data on bycatch rates a challenge due to the complexity that these social-ecological systems represent (Dietz et al. 2003).

Obtaining reliable data on the incidental capture and mortality of vulnerable species is, nonetheless, necessary to achieve ecologically and socioeconomically sustainable fisheries (Suuronen & Gilman 2019). At sea, human observer programmes can produce accurate data on the incidental rates of capture and bycatch in fisheries. However, independent validation of the data are essential, and effective implementation of observer programmes in small-scale

fisheries can be complicated and expensive (Bartholomew et al. 2018; Suuronen & Gilman 2019). Note that here I define capture as everything that is caught and retained in fishing gear, and bycatch as capture that is discarded at sea, dead or injured to an extent where death is the result; following the definitions in Hall (1996). Further complexity arises because observer coverage is rarely available for an entire fleet; bycatch estimates are often inferred from a subset of fishing trips, typically using models to help control for sampling biases (Benoît & Allard 2009). Electronic monitoring programmes are increasingly trialed and implemented in both large- (Ames et al. 2007; Needle et al. 2014) and small-scale fisheries (Bartholomew et al. 2018) – trials in small-scale fisheries appear promising, with accuracy similar to at sea, human observers at a lower cost. Despite clear potential in the use of electronic monitoring in fisheries, multiple studies evaluating the technology note that improvements are still needed to detect certain species, and recording of catch released below the water level or in areas outside the camera view remains a major limitation (Bartholomew et al. 2018; Gilman et al. 2019; Suuronen & Gilman 2019). Post-trip interviews of skippers and crews also quantify bycatch in small-scale fisheries (Goetz et al. 2013; Alfaro-Shigueto et al. 2018). These data often take the form of questionnaires and provide the cheapest and most rapid source of information – at times, supporting near real-time bycatch management measures (Drew 2005). Quickly obtaining a broad understanding of bycatch impact can be particularly useful in small-scale fisheries, in which data are often poor (or absent). While potential exists for post-trip questionnaires to support rapid evaluations of incidental captures to inform bycatch impact in data-poor fishery scenarios, questionnaire-based interviews are often considered to be less reliable than data collected using observer programmes as they are subject to individual respondents biases and heuristics (Suuronen & Gilman 2019).

In conservation, expert knowledge (substantive information on a particular topic that is not widely known by others; Martin et al. 2012) can be used to inform the decision-making process. Expert knowledge is used due to the need to make timely management decisions about complex and dynamic environments, particularly in data-poor scenarios, unique circumstances, or when predictions under uncertainty are required (Cook et al. 2010; Burgman et al. 2011). When drawing on expert knowledge, however, it is essential to account for the contextual biases and heuristics that individuals bring with them, as these can affect the validity of the information they give and the subsequent management actions that result (O'Hagan et al. 2006; Kynn 2008). The need to design and implement effective conservation strategies that are rigorous, robust, repeatable, and include an estimate of uncertainty has

resulted in structured, evidence-based elicitation protocols such as the widely used Delphi method, which provides feedback from experts over successive questionnaire rounds (Helmer-Hirschberg et al. 1966; Cooke 1991).

While elicited data are not a substitute for empirical data, structured protocol techniques prove informative in various fishery management settings when empirical data are not readily available. For example, a risk assessment for New Zealand’s critically endangered Māui dolphin (*Cephalorhynchus hectori maui*) used a structured elicitation approach to estimate population abundance (Currey et al. 2012). Similar processes have been used to evaluate and rank threats to sea turtle populations in several fishing systems (Klein et al. 2017; Williams et al. 2017; Riskas et al. 2018). When fisheries lack the data and resources to implement more comprehensive observer procedures, significant potential exists to apply structured elicitation protocols to expert opinion to reduce personal biases and heuristics, and quantify associated uncertainty.

Peru’s small-scale fisheries significantly impact marine biodiversity through bycatch (Alfaro-Shigueto et al. 2010). The gillnet, the most commonly utilised gear (Castillo et al. 2018), has been identified as a major sink for several species of sea turtle of conservation concern (Alfaro-Shigueto et al. 2007; Alfaro-Shigueto et al. 2011; Alfaro-Shigueto et al. 2018). Peru’s current regulatory structure does little to help mitigate bycatch of protected species like sea turtles in the country’s small-scale fisheries. With limited government efficacy, not-for-profit organisations play a role in filling data gaps and implementing conservation interventions to minimise bycatch. For example, not-for-profit organisations in Peru implement and maintain volunteer observer programmes with small-scale fishers, and undertake post-trip interviews of skippers and crews (Alfaro-Shigueto et al. 2011; Alfaro-Shigueto et al. 2018). Conservation efforts such as these highlight the need for further management actions to help reduce vulnerable species bycatch in small-scale fishing systems. Here I compare at sea human observer data from a small-scale fishery to incidental capture estimates of green turtles obtained using a structured elicitation protocol, in an effort to improve rapid, exploratory evaluations of marine megafauna captures in data-poor small-scale fishery management scenarios.

In this study, I use the IDEA protocol (“Investigate,” “Discuss,” “Estimate” and “Aggregate”), which follows a modified Delphi method, incorporating many suggested adaptations to structured elicitation protocols that have been used in previous conservation

research. Specifically, the protocol uses a four-step elicitation process to reduce overconfidence (Speirs-Bridge et al. 2010), encourages consultation with a diverse group of experts (Burgman et al. 2011), affords experts the opportunity to examine one another's estimates and to reconcile the meanings of questions through discussion, and uses performance-based mathematical aggregation of judgements (Hanea et al. 2016a). To date, the protocol has produced robust estimates in several studies (e.g., Hanea et al. 2016a; Hanea et al. 2016b; Van Gelder et al. 2016; Hemming et al. 2018c), and shows promise as a useful tool for rapidly assessing bycatch impact in small-scale fisheries.

The aims of this research were to i) investigate if the IDEA protocol could support a rapid assessment of incidental captures of protected species (green turtles *Chelonia mydas*) occurring in a coastal gillnet fishery where sea turtle bycatch is a known conservation issue, ii) quantify the associated uncertainty, and iii) evaluate participant performance by comparing these estimates to incidental capture rates calculated from fisheries observer data obtained from the same fishery.

4.2 Methods

4.2.1 Study system

San Jose, Lambayeque, Peru (6°46' S, 79°58' W) is a coastal fishing community with a high density of gillnet vessels (Alfaro-Shigueto et al. 2010). The bycatch of several turtle species is known and problematic, including captures of the East Pacific population of green turtles (*C. mydas*) and the critically endangered East Pacific population of leatherback turtles (*Dermochelys coriacea*; Alfaro-Shigueto et al. 2011; Alfaro-Shigueto et al. 2018). A voluntary at sea, human observer programme has been running with San Jose's gillnet skippers since 2007; however, coverage has not been comprehensive (Alfaro-Shigueto et al. 2007; Alfaro-Shigueto et al. 2010). Structured questionnaires have also been used to further knowledge of turtle bycatch in the area (Alfaro-Shigueto et al. 2011; Alfaro-Shigueto et al. 2018). Several fishers in the San Jose community have been exposed to conservation interventions. For example, five gillnet skippers and their associate crew work with a not-for-profit organisation on at sea, bycatch mitigation technology trials (Ortiz et al. 2016; Bartholomew et al. 2018). A broader group of fishers in San Jose partake in workshops to learn better handling procedures for turtle releases post-capture.

The inshore-midwater gillnet fleet comprises vessels with small closed bridges that range in capacity from 5 to 32 gross registered tonnage (GRT), locally known as ‘lancha’ (Guevara-Carrasco and Bertrand, 2017). Vessel numbers fluctuate both seasonally and annually, as fishers migrate from inland areas seeking fishing work during favourable weather conditions. Over the past decade the San Jose inshore-midwater gillnet fleet has been reducing in size as fishers shift their vessels from handling gillnets to jigging gear to catch giant Humboldt squid *Dosidicus gigas*. Fleet size of the San Jose inshore-midwater gillnet fishery was approximately 60 vessels in 2008, with numbers decreasing to between 28 and 18 in the summer and winter of 2017, respectively (Alfaro-Shigueto et al. 2010; Supporting Information). A winter survey in San Jose in 2017 (July-September) estimated that 15 inshore-midwater vessels actively fished, primarily with gillnets, while 3 additional vessels used gillnets but primarily fished with another gear type. Another small-scale gillnet fleet comprised of small, open-welled vessels known as ‘chalana’, with a capacity range of 1-8 GT, also operates from San Jose in the inshore fishing area (Arlidge et al. 2020). Both skippers from the inshore-midwater gillnet fleet (respondents in the current study) and skippers of the other inshore gillnet fleet were part of a wider elicitation survey investigating the efficacy of bycatch reduction strategies in the San Jose fishing system. Only the inshore-midwater fleet is the focus of this comparative study because observer data were not available for the inshore gillnet fleet.

I separately assessed two seasonal categorisations due to the differences in fishing effort between winter and summer conditions in the Lambayeque coastal fisheries. Summer usually spans December-February (3 months), but the information provided by a government fisheries scientist in San Jose during a key informant interview noted that summer-like conditions last from December-May, with this longer seasonal division supported by capture reports from the Lambayeque region (Guevara-Carrasco & Bertrand 2017). Here I classify the San Jose winter fishing season as June-November and the summer fishing season as December-May.

4.2.2 Estimates of turtle encounters

To elicit judgements of incidental captures of green turtles in gillnets set by San Jose inshore-midwater vessels, participants were asked to consider a counterfactual scenario in which a total gear switch occurred, from gillnets to a fishing gear that results in very little chance of

turtle bycatch (such as lobster potting or trolling – a form of handline fishing). Estimates were provided as a monthly reduction in green turtle encounters with gillnets for the entire San Jose inshore-midwater fleet. Estimates for leatherback turtles were also elicited, but small numbers make these less reliable than the green turtle estimates (Supporting Information). Participants were asked to assume 100 per cent compliance with the counterfactual scenario. Judgements were given for summer and winter fishing periods. These data were collected as part of a wider elicitation study that elicited expert judgements on the efficacy of a range of turtle bycatch reduction strategies that will be used to inform a multi-bycatch mitigation strategy model (Milner-Gulland et al. 2018).

4.2.3 Expert elicitation procedure

I use the IDEA protocol with a combination of face-to-face group meetings and individual interviews over two elicitation rounds.

4.2.3.1 Participant selection

I used simple random sampling by number generator to select gillnet skippers from a census list (n=168) of skippers that were actively fishing during a wider survey period of 1 July – 30 September 2017. The expert group (n=5) comprised three local gillnet skippers of inshore-midwater vessels (representing 20% of the actively fishing inshore-midwater gillnet skippers in San Jose), and two not-for-profit conservation organisation employees (JAS & JCM). Both of the not-for-profit employees have carried out regular research and conservation action in the study site area and more widely along the western South American coastline. They have expertise in turtle ecology and the implementation of bycatch reduction strategies in small-scale fisheries.

4.2.3.2 Elicitation format

Data were elicited through individual face-to-face interviews over two elicitation rounds. Because lancha fishers spend little time on land between fishing trips of 1–13 days (averaging 7 days; Alfaro-Shigueto et al. 2010), this resulted in no time in which the inshore-midwater gillnet skippers were all on land, following an initial scoping meeting. Hence the decision was made to interview them separately.

4.2.3.3 Stage 1: Introductory meeting

The first stage of the elicitation procedure was undertaken in a face-to-face group meeting. I met with the invited participants and discussed the context of the elicitation procedure with them, including providing an overview of the IDEA protocol, the method, study rationale, and the rules of participation. I ensured that free, prior, informed consent to participate was given, in accordance with our ethics permission (CUREC 1A; Ref No: R52516/RE001 and R52516/RE002).

4.2.3.4 Stage 2: Investigate (Round 1)

Question format followed a four-point estimation method that has been shown to reduce overconfidence when eliciting individual judgements (Speirs-Bridge et al. 2010). This involves giving a (i) lower bound, (ii) upper bound, (iii) best guess, and (iv) a level of confidence that the real value lies between these limits. Participants were asked to give estimates of the expected reduction in green turtle captures in gillnets within the winter and summer fishing seasons, for the total gillnet ban bycatch reduction scenario. Estimates were given as monthly gillnet encounters, unless another time period was specified by the participants (e.g., turtle gillnet encounters per season). In cases when turtle captures per season were given, estimates were divided by the total number of months in the season.

4.2.3.5 Stage 3: Analysis and feedback

In the four-step question format, participants implicitly specify credible intervals for their estimates. For example, if in response to the question about how confident they are about their estimate, a participant says that they expect the true value to fall between their stipulated lower and upper limits in 7 of 10 cases; that implies a 70 per cent credible interval. Before providing the first round of feedback, I standardised the participants' estimated intervals to 90 per cent credible intervals to allow them to see the uncertainties across their estimates on a consistent scale. Linear extrapolation was used to standardise participants' elicited lower (l) and upper (u) uncertainty bounds to 90 per cent credible bounds (Hemming et al. 2018a). The standardised lower (l_{si}) and upper (u_{si}) bounds were calculated as:

$$l_{si} = B - \left((B - L) \times \left(\frac{S}{C} \right) \right) \quad (1)$$

$$u_{si} = B + \left((U - B) \times \left(\frac{S}{C} \right) \right) \quad (2)$$

where l_{si} = standardised lower estimate, u_{si} = standardised upper estimate, B = best guess, L = lowest estimate, U = upper estimate, S = level of credible intervals to be standardised to, and C = level of confidence given by participant. Any adjusted intervals that fell outside of reasonable bounds (i.e., negative values) were truncated at their extremes (i.e., to zero).

Following standardisation, estimates were combined using quantile aggregation, in which the arithmetic mean of participants' estimates is calculated for the lower, best, and upper estimates for each question (Hemming et al. 2018a). Graphs for each question were generated to display the estimates of each participant (labelled with codenames that each respondent was individually aware of) and the group aggregate mean. This output was presented to the participants for use in the discussion and re-estimation phase that followed (Supporting Information).

4.2.3.6 Stages 4 & 5: Discussion and re-estimation (Round 2)

The discussion and re-estimation phase took place through individual face-to-face interviews, led by the facilitator (BIE) with support from the coordinator and analyst (WNSA; Hemming et al. 2018a). We provided hard copies of each question's graphical output to the participants; this included justification comments from the other participants (when given) and any questions from the analyst (Supporting Information). No participants declined to partake in the second elicitation round.

4.2.3.7 Stage 6: Final aggregation and review

Following the second elicitation round, the revised data were analysed and aggregated. I

presented first and second round estimates, along with the arithmetic mean for the group's best, lower, and upper estimates to each participant in plot and table form for a final review. Participants were allowed to make fine-scale adjustments to their own estimates if desired; no participants did this.

4.2.4 Statistical analysis

4.2.4.1 Fisheries observer data

To obtain information on the turtle capture per unit effort, i.e., capture rates, for San Jose gillnet vessels against which to compare elicited estimates, I analysed longitudinal panel data. These data were recorded by fisheries observers operating in the inshore-midwater gillnet fleet from San Jose as part of a wider at sea volunteer observer program run by our local not-for-profit collaborators (JAS, JCM). Capture per unit effort was calculated per trip (n=461) averaged across seasonal (summer and winter) and annual time periods (n=10). Observed trips were across 32 different inshore-midwater gillnet vessels with varying vessel and net sizes. Historical vessel numbers for the inshore-midwater gillnet fleet were obtained from shore-based surveys (Escudero 1997; Alfaro-Shigueto et al. 2010); for years with no known vessel size, an interpolated approximation was used (Supplementary Information). Mean green turtle capture per unit effort per season were then converted to mean capture per unit effort/per month within each season by averaging across each season's months (Supplementary Information). Descriptive statistics are presented as mean, standard deviation (SD), and minimum and maximum 90 per cent confidence intervals (CI).

Using the observer dataset (n=461), I extrapolated green turtle capture rates from the proportion of the inshore-midwater fleet covered by observers to the wider gillnet fleet. I categorised vessel GRT into size classes, and then weighted these size classes using binomial logit Generalised Linear Mixed Models (GLMMs) using maximum likelihood estimation and AIC model selection criteria. GLMMs were constructed in R version 3.6.1 (R Core Team 2019) using the `nlme` package (Pinheiro et al. 2012). Explanatory variables were selected a priori and included GRT, season, year, gillnet soak time (the time the net spends in the water), net length (km), and crew number as fixed effects. Vessel identification was included in the model as a random effect. I tested for fixed versus random effects using the Hausman test in the `plm` package in R, failing to reject the null hypothesis of random effects (against fixed

effects; Croissant & Millo 2008; Supplementary Information). To avoid collinearity among variables in the model, I used Spearman's rho (rs) correlation coefficients to calculate the correlation between pairs of variables (Akoglu 2018). Any highly correlated variables ($r > 0.8$) would not be used together in the models. None of the variables selected a priori were correlated enough to warrant removal from the model (Table 4.1). After regressing sea turtle capture rates upon the independent variables, I tested for serial correlation and present serial correlation consistent standard errors. I then used the model's coefficients to weight the overall probability of capture of each turtle species by weight class (GRT) across the inshore-midwater gillnet fleet. I also modelled leatherback turtle capture rates; however, the low capture rate recorded ($n=7$) resulted in little predictive power in the model (Supplementary Information).

Table 4.1. Spearman's rho (rs) rank correlation test results for variable inclusion in binomial Generalised Linear Mixed Model (GLMM). GRT = gross registered tonnage, p = p-value. Correlation coefficients are highlighted in bold, p-values in plain text. Soak time is the length of time the net is in the water during each set (setting and hauling of the net). Fishers will often lay multiple sets during a fishing trip. To support interpretation, the correlation coefficient of GRT and season was 0.058, with a p-value of 0.005. The low level of the p-value indicated that 99.995% of the time the correlation is weak at an r of 0.058.

	GRT	Season	Year	Soak	Net length (km)	Crew number
GRT	1 ($p < 0.01$)	0.058 ($p = 0.005$)	-0.180 ($p < 0.001$)	-0.022 ($p = -0.296$)	-0.132 ($p < 0.01$)	-0.221 ($p = 0.286$)
Season		1 ($p < 0.01$)	0.01 ($p = 0.596$)	-0.061 ($p = 0.002$)	-0.081 ($p < 0.001$)	-0.027 ($p = 0.173$)
Year			1 ($p < 0.01$)	-0.038 ($p = 0.054$)	0.228 ($p < 0.01$)	-0.341 ($p < 0.01$)
Soak time				1 ($p < 0.01$)	0.002 ($p = 0.914$)	0.074 ($p < 0.001$)
Net length (km)					1 ($p < 0.01$)	0.062 ($p = 0.001$)
Crew number						1 ($p < 0.01$)

4.2.4.2 Bootstrap comparison of means

The small sample size in our elicitation group precludes directly comparing the dataset to the capture rates calculated from the observer dataset using a large-sample test such as an independent 2-sample t-test. Instead, I used a bootstrap method to simulate the expected

distribution of monthly turtle bycatch rates calculated per season from the elicitation dataset and the observer dataset and compare the two (Efron & Tibshirani 1993). The bootstrap methodology consists of generating a null data set that has the same number of subjects as in the original data set by randomly selecting subjects from the control group with replacement and using the whole series of repeated measurements from each randomly selected control subject (Nadziejko et al. 2004).

I tested the null hypothesis that, within each fishing season, the mean monthly number of green turtles captured in the San Jose inshore-midwater gillnet fleet calculated from the elicitation exercise is the same as the capture rate calculated from the fisheries observer data. This required comparing monthly elicited estimates of green turtle bycatch rates within summer (n=5) and winter (n=5) fishing seasons, with monthly capture rates within summer and winter across each fishing year of observer data (n=10 summer, n=10 winter).

The monthly green turtle capture rates per season from the elicitation data represents the expected value based on multiple years of data, which may or may not be the same length (i.e., one expert may be drawing on 20 years of fishing experience when considering their monthly green turtle bycatch rates, whereas another may be drawing on 5 years fishing experience). I estimated green turtle captures from the observer data for each year data was available (n=10 over a 13-year period) and then averaged across this period. Consideration of the potential source of temporal bias between the two datasets is highlighted for results interpretation. All analysis was carried out using core packages in R version 3.6.1 (R Core Team 2019).

4.2.4.3 Performance-based metrics for elicitation estimates

Participants were not asked to define whether their best estimates represent a mean, mode, or median, nor were they asked to specify the quantiles of distribution (i.e., how the residual uncertainty their interval judgements were distributed outside of their bounds; Hemming et al. 2018c). Under more standard elicitation circumstances, mean, median, or mode data may be requested from respondents. In the current study, however, it was not deemed socially appropriate to ask gillnet skippers to specify these measures. I therefore chose metrics that are not based on continuous probability distributions. Instead, participants' performance was evaluated using three performance-based metrics: (1) accuracy of point (best) estimates, (2)

calibration of interval judgements, and (3) informativeness of interval judgements (after McBride et al. 2012; Hemming et al. 2018c; Figure 4.1).

4.2.4.3.1 Accuracy

Accuracy of point estimates ('Accuracy') is classified as the distance of the respondent's best estimate from the turtle capture rates calculated from the fisheries observer data (typically referred to as the realised truth; Einhorn et al. 1977; Larrick & Soll 2006). Accuracy of point estimates ('Accuracy') was measured by calculating the average log-ratio error (ALRE) for participants' judgements. To calculate ALRE, I first standardised each response by the range of responses for that question, known as range-coding (McBride et al. 2012; Hemming et al. 2018c). Range-coding is used to minimise the effect that one or a few divergent responses have on the accuracy measure (Burgman et al. 2011). The calculation involves standardising the best estimates $b_e^{n,r}$ from each participant e , for each question n , in each round r by the range of responses for each question:

$$bc_e^{n,r} = \frac{(b_e^{n,r} - b_{min}^n)}{b_{max}^n - b_{min}^n} \quad (4.1)$$

where, $bc_e^{n,r}$ is the range-coded response for participant e , in round r , b_{max}^n is the highest best estimate response taken from the pool of best estimate responses from all participants for question n , across both elicitation rounds, and b_{min}^n is the minimum best estimate response (Hemming et al. 2018c). The realised truth (x^n) for each question is also range-coded using equation 3.

ALRE is then calculated using the range-coded values generated:

$$ALRE_i = \frac{1}{N^r} \sum_{n=1}^N \left(\log_{10} \left(\frac{x^{n+1}}{bc_e^{n,r} + 1} \right) \right) \quad (4.2)$$

where, N_r represents the number of quantities assessed in round r , $bc_e^{n,r}$ is the range-coded prediction, and x^n is the range-coded observed value ('realised truth') for question n (Hemming et al. 2018c). To avoid taking the log of zero when the realisation is standardised, ones were added to both the range-coded observed value and the range-coded prediction. The $\log_{10}$ ratio provides a measure that emphasizes order of magnitude errors rather than linear errors. In other words, a judgement that is five times the observed value x would be weighted the same as a value that is one-fifth the value of the observed value x . Smaller ALRE scores indicate more accurate responses (Hemming et al. 2018c). The log ratio scores have a maximum range of $\log_{10}(2)(0.31)$ (Hemming et al. 2018c). The maximum log range occurs when the true answer coincides with either the group maximum or group minimum, and a best possible score of zero (McBride et al. 2012).

4.2.4.3.2 Calibration

Calibration of interval judgement ('Calibration') measures the proportion of questions answered by a respondent for which their intervals capture the realised truth, with a score of 0.9 representing perfect calibration. The perfect calibration threshold is set at 0.9 because participants were asked to provide 90% credible intervals; therefore a participant would be considered perfectly calibrated if they capture the truth for 9 out of 10 questions answered. Following the protocol outlined in Hemming et al. (2018b), I used the standardised upper and lower values of participants' intervals and the standardised level of confidence associated with those intervals. Using participants' standardised intervals to score calibration is possible as the participants receive feedback on their standardisations between Round 1 and Round 2. Participants were informed they should adjust their estimates if they are not in accordance with their true beliefs. The actual number of realisations captured was calculated using:

$$C_e^r = \frac{t^r}{N^r} \times 100 \quad (4.3)$$

where, C_e^r is the score for calibration for participant e in Round r , t is the number of standardised intervals provided by the participant which contained the realised truth, and N_r is the total number of questions the participant answered in round r . Because it is possible for

participants to obtain a high calibration by providing wide (uninformative) intervals, this calibration measure is considered alongside a measure of informativeness (described below).

4.2.4.3.3 Informativeness

Informativeness of interval judgement ('Informativeness') measures the width (i.e., the precision) of the participant's intervals relative to the group range provided by participants for a question. First, I calculated the width of standardised intervals (e.g. 90%) supplied by participants for each question in each round:

$$w_e^{n,r} = u_e^{n,r} - l_e^{n,r} \quad (4.4)$$

where, $w_e^{n,r}$ is the width of the standardised interval of participant e for question n , in round r , while $u_e^{n,r}$ is the upper standardised estimate provided by participant e for question n , in Round r , and $l_e^{n,r}$ is the lower standardised estimate provided by participant e for question n .

Then for each question, a background range was calculated:

$$w_{max}^n = u_{max}^n - l_{min}^n \quad (4.5)$$

where w_{max}^n is the background range created for question n , u_{max}^n is the maximum standardised upper bound provided for question n across Round 1 and Round 2 by any participant, and l_{min}^n is the lowest standardised lower bound estimate provided for question n across Round 1 and Round 2 by any participant.

Finally, the average informativeness score of each participant per round was calculated by:

$$I_e^r = \frac{1}{N^r} \sum_{n=1}^N \left(\frac{w_e^{n,r}}{w_{max}^n} \right) \quad (4.6)$$

where, I_e^r is the average informativeness of participant e in Round r over all questions in Round r , $w_e^{n,r}$ is the width of the interval provided by participant e in Round r for question n , w_{max}^n is the background range for question n , and N^r is the total number of questions answered in Round r . Scores range between 0 and 1 – higher numbers relate to less informative individuals. To ensure that participants are not rewarded for not reporting any uncertainty when they are not certain of the true value of observed value x , this measure is considered in conjunction with the calibration measure. The performance-based metric analysis was undertaken in R using quantile aggregation code available on the open-science framework (Hemming et al. 2018b).

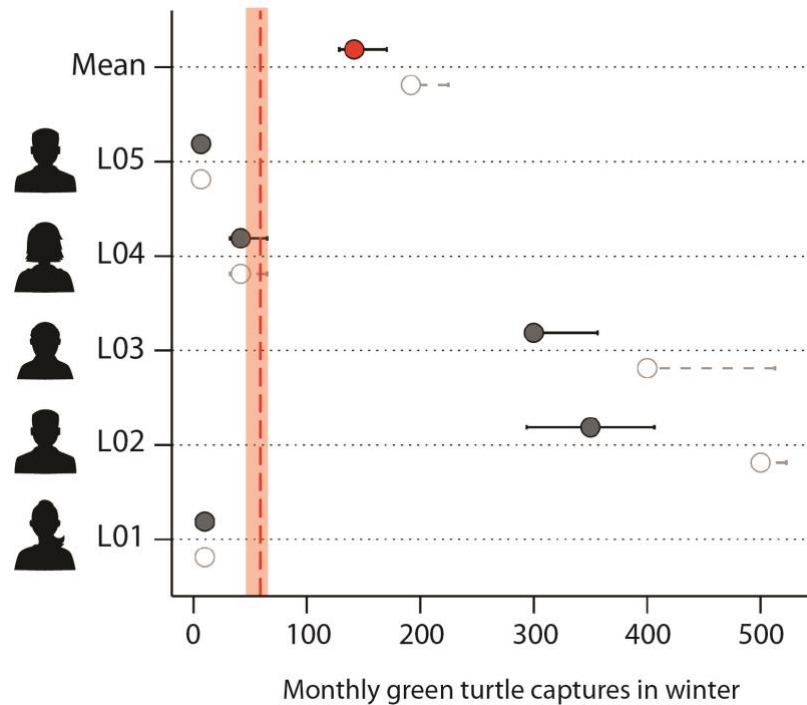


Figure 4.1. Respondents' elicitation estimates for monthly green turtle gillnet captures in winter that are used to explain the accuracy, calibration, and informativeness performance metrics. Participants present estimates (L01-L05) with Round 2 best estimates (grey circles) and associated credible intervals (horizontal lines). The group mean is the red circle. The red dotted line represents the bycatch rates estimated from the observer data and the light red band the uncertainty bounds. Participants (L01, L05) are the most informative (smallest credible interval) and their informativeness intervals do not capture the 'realised truth' (which if done over multiple questions would mean they are poorly calibrated). Participant (L02) is the least accurate (best estimate is furthest from the realised truth) and the least informative (largest credible interval). Participant (L04) has the most accurate estimate (closest best estimate to the realised truth), and their credible interval encompasses the realised truth (which if done over multiple questions will result in a good calibration score).

4.3 Results

Five respondents comprising 3 gillnet skippers and 2 not-for-profit employees participated in the elicitation procedure for the inshore-midwater fleet. The group comprised 4 males and 1 female. Respondent age was 27-50 years. Fishing experience for skippers was 11-17 years (Table 4.2).

Table 4.2. Expert elicitation respondent characteristics.

Variable	Category	Number
Stakeholder group	Gillnet skipper	3
	Not-for-profit scientist	2
Gender	Male	4
	Female	1
Age	26–40	3
	41–55	2
Years lived in San Jose	0–10	2
	11–20	0
	21–30	1
	31–40	2
Years fishing	0–10	2
	11–20	3

4.3.1 Elicited judgements for turtle captures

The group's green turtle confidence bounds were 129 – 227 individuals per month (Table 4.3). I used participants' monthly green turtle capture rates with gillnets to infer capture rates for the six-monthly summer (mean=850, range=771–1022) and winter (mean=1234, range=1105–1363) seasons. I then summed the seasonal estimates to obtain an annual capture rate (mean=2084, range=1876–2385; Table 4.3). As a supplementary analysis, participants' judgement of leatherback bycatch was also explored (Supporting Information).

4.3.1.1 Comparison of participant judgements with fisheries observer data

I analysed fisheries observer records from the inshore-midwater gillnet fleet in San Jose from August 2007 to March 2019. Over 461 inshore-midwater fishing trips, observers recorded the capture of 379 turtles in gillnets. Species proportions were 86.8% green sea turtles (n=329),

9.2% olive ridley turtles (n=35), 1.8% leatherback sea turtles (n=7), and 2.1% unidentified (n=8). Of the 379 turtles captured, 62% were released alive without visible injury, 28% alive with minor injuries, and 8% were returned dead (Table 4.4). Observer coverage for the fleet is low, representing approximately 1-4% of net deployments over the 11-year, 7-month monitored period (Supporting Information). As observer deployments occur on a volunteer basis with skippers, sampling selection bias is likely. No vessels were observed in the 2010-2012 fishing years (Table A2.1.7).

Table 4.3. Extrapolated mean estimates of green turtle captures in San Jose inshore-midwater gillnets in summer and winter, between expert elicitation and at sea observer datasets. Values are based on elicited monthly estimates of the efficacy of the bycatch reduction strategy scenario of gear switching from gillnets to potting or trolling, and at sea fisheries observer data obtained from the period August 2007–May 2019. Temp. Grp. = temporal grouping; winter represents the cold weather months of June to November, summer represents the warm weather months of December to May.

Temp. Grp.	Expert elicitation data (n=5)			Observer data (n=461)		
	Mean best (B)	Std. lower 90 CI (lsi)	Std. upper 90 CI (usi)	Mean	Min 90 CI	Max 90 CI
Monthly/winter	141.67	128.54	58.51	58.51	46.49	66.02
Monthly/summer	205.67	184.14	227.09	137.09	108.93	154.69
Total winter	850	771.25	1022.12	351.07	278.96	396.14
Total summer	1234	1104.85	1362.55	822.54	653.59	928.13
Annual	2084	1876.1	2384.67	1173.61	932.55	1324.28

The most parsimonious model for green turtle capture included the variables GRT, season (winter and summer), fishing year, soak time, and a random effect for skipper-vessel (Table 4.5). The skipper-vessel effect includes the effect of both the vessel and the skipper, the latter which cannot be measured or distinguished from the available data. There may also be a relationship between the skipper and vessel size. Larger vessels were more likely to capture turtles in a given trip than those with small capacities, after controlling for fishing effort. This may be a result of larger vessels having the ability to hold larger nets and stay at sea fishing for more extended periods, as well as covering a larger fishing area because they can carry more petrol and oil, larger quantities of ice for their catch, and more supplies for the crew. Fishing across a larger fishing area may result in larger vessels having access to different fishing grounds where there are more turtles present. Based on this model, I extrapolated the

observer data to produce a mean annual gillnet capture estimate of 1174 (range 933-1324) green turtle individuals (Table 4.3).

I ran two bootstrap hypothesis tests (each of 10,000 resamples with replacement) for the mean monthly estimates of green turtle gillnet captures within summer and winter fishing seasons. For both winter and summer; I found no statistically significant difference at the 95% confidence level in the mean monthly capture estimates of green turtle between the elicited data and the observed data (winter observed difference-in-means: 83.15, adj mean $\pm$ SD = 42.39 ± 32.59 ; $p=0.1177$; summer observed difference-in-means: 68.58, adj mean $\pm$ SD = 54.06 ± 41.22 ; $p=0.309$; Table 4.3).

Table 4.4 Turtle bycatches, and capture per unit effort in gillnets set by inshore-midwater vessels launching from San Jose in the period August 2007–May 2019, based on an at sea fisheries observer program, using trip as the unit of effort. CI = confidence interval.

Turtle species	n	Released without injury	Released injured	Dead	State unknown	Capture per unit effort/per trip (n=461)			
						Mean	SD	Min 90% CI	Max 90% CI
Green	329	199	100	23	7	0.71	1.98	0.53	0.89
Leatherback	7	6	0	1	0	0.02	0.12	0.01	0.03
Olive ridley	35	24	6	5	0	0.08	0.46	0.04	0.12
Unidentified	8	4	2	0	2	0.02	0.21	0.00	0.04
Total turtle bycatch	379	233	108	29	9	0.82	2.10	0.63	1.01

Participant L05 (not-for-profit) judged lower capture rates for green turtles than estimated from the observer data. Participant L04's (not-for-profit) judgement intervals encompassed the observer data for both seasonal estimates (Figure 4.2). By contrast, participants L02 and L03 (gillnet skippers) estimated significantly higher capture rates across both winter and summer seasons. Participants L02 and L03 adjusted their estimates downwards between Round 1 and Round 2 in the modified Delphi method, to be closer to the value estimated from the observer data. This indicates that new information from the discussion between elicitation rounds influenced participant L02 and L03's calibration and accuracy of judgement. Participant L01 (gillnet skipper) estimated closer to the realised truth and to the not-for-profit participants than the other two skippers (Figure 4.2).

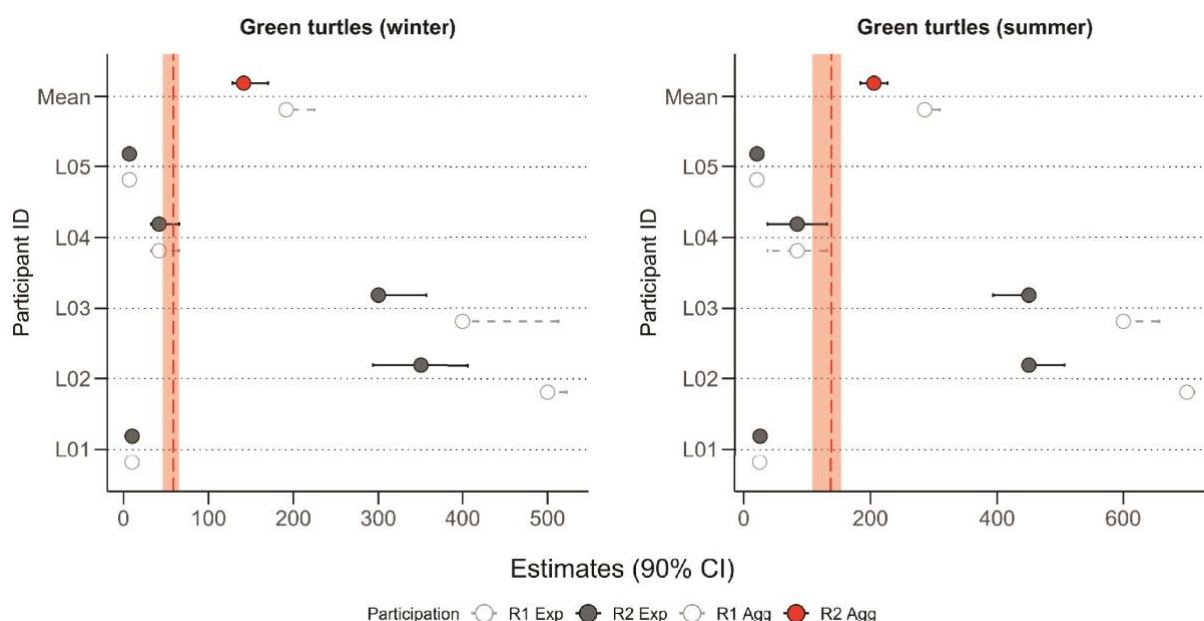


Figure 4.2. Respondents mean monthly estimates of green turtle captures in gillnets compared to extrapolated catch rates calculated from the observer data (red dotted line) with associated uncertainty bounds for the observer data (light red band). Elicited estimates are based on consideration of a possible gear switch from gillnets to trolling or lobster potting for all vessels in the San Jose inshore-midwater gillnet fleet. Monthly estimates were made for summer and winter fishing seasons. Experts assumed 100% compliance with the total gear switch scenario. Uncertainty bars have been adjusted to reflect 90% credible intervals for each expert's response.

Table 4.5. Coefficients of the best fit model for predicting probability of turtle bycatch.

<i>Green turtle bycatch</i>		Serial correlation-consistent standard errors		
Reference	Random effects	Intercept	Residual	n
Vessel	Std. dev	0.02837082	0.2556781	32
	Fixed effects	Coefficient	SE ₁	p-value
	Intercept	-0.014	0.026	0.59
GRT	4<8 GRT	0.036	0.030	0.2362
<i>reference = 0<4 GRT</i>				
	8<12 GRT	0.050	0.026	0.0646 *
	>12 GRT	0.104	0.083	0.2193
Year	Year (2011-2014)	0.153	0.022	0.0000 *
<i>reference = Year (2007-2010)</i>				
	Year (2015-2019)	0.012	0.014	0.3862
Season	Season (Summer)	0.036	0.012	0.0028 *
<i>reference = Season (Winter)</i>				
	Soak time	0.001	0.001	0.60

4.3.1.2 Performance metrics

Participant performance was evaluated by occupation groupings (skippers versus not-for-profit), comparing elicited estimates for the total gillnet ban to the capture rates calculated from the observer data (Figure 4.3). The not-for-profit employees were on average more accurate (lower ALRE score), better calibrated (their credible intervals encompassed the realised truth over more questions elicited), but less informative (they specified larger credible intervals) than the skippers. The skippers scored higher on informativeness than the not-for-profit employees, but lower on accuracy. Two of the five participants improved the accuracy of their estimates between the two elicitation rounds, and one improved informativeness. Participants did not improve the calibration of their estimates between the elicitation rounds (there was no increase in the number of realised truths captured between their upper and lower bounds). This is potentially reflective of overconfidence or attitudes towards risk from the skippers, leading to them submitting estimates with tight confidence bounds (high informativeness) that underestimate uncertainty (low accuracy).

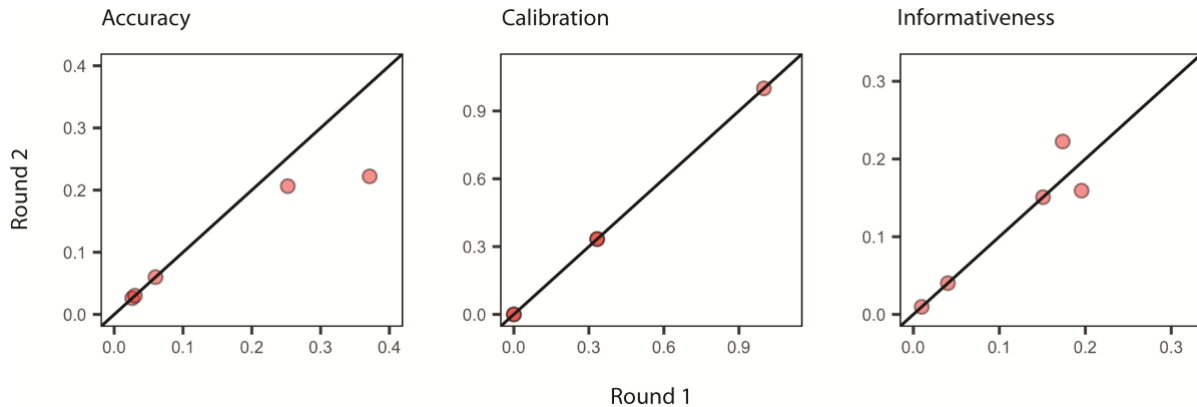


Figure 4.3. Scatterplots show the change in respondents estimates (n=5) between elicitation rounds by performance metrics. If dots fall below the line in the "accuracy" or "informativeness" plots, individuals improved their scores on these measures. In the "Calibration" plot, dots above the line indicate individuals who increased the number of realised truths captured between their upper and lower bounds (a score of 0.9 represents perfect calibration).

4.4 Discussion

The estimates of green turtle captures in the San Jose inshore-midwater gillnet fleet, obtained from both the observer data and the group estimates from the expert elicitation, indicate detrimental capture and bycatch rates for turtle populations like the endangered green turtle and the critically endangered East Pacific leatherback turtle population (assessed in Supporting Information) as both species are highly vulnerable to fishing pressure (Spotila et al. 2000; Lutcavage 2017). While the elicited estimates focused on capture rates, the observer data found 7% of captured green turtles died and 38% were returned to sea injured, indicating the potential for a high percentage of estimated captures to result in mortality (Table 4.4). Green and leatherback turtles are far ranging and traverse multiple nations' waters in their lifetimes. The southeast Pacific waters that these species swim through (Bailey et al. 2012; Eckert 2012) are fished by multiple small-scale fisheries where observer programmes are limited or not currently established (Salas et al. 2007; Sara 2011). For example, questionnaire-based surveys estimated that small-scale fisheries-related turtle mortality across 7 Ecuadorian harbours was 13,302 turtles per year (Alfaro-Shigueto et al. 2018). The IDEA protocol offers potential to improve data paucity on incidental capture and bycatch rates in data-poor fisheries such as those in the southeast Pacific by offering a decision-making process to more accurately quantify uncertainty and control for respondents' personal biases and heuristics.

The bootstrap hypothesis testing approach allowed me to compare the means of the two datasets despite small sample sizes. A high level of variation in total fleet size across observed fishing years meant that for observed years where no quantitative total fleet size estimates were available from shore-based surveys (Escudero 1997; Alfaro-Shigueto et al. 2010), I was required to use an interpolated approximation of fleet size. This uncertainty must be considered when interpreting the results. Despite the need to approximate fleet size, the methods used in the current study demonstrate that the IDEA protocol can provide broad estimates of protected species captures in small-scale fishery systems that are informative.

Both of the not-for-profit employees' judgements of green turtle captures were consistently closer to the observer data than the group mean. This finding contrasts with a number of Delphi-based elicitation studies that found that pooled group judgements

consistently outperform individuals (Burgman et al. 2011; Burgman 2015). These results may be reflective of two of the three gillnet skippers who were consistently overestimating when compared to the observer data. Due to the elicitation group's small sample size these estimations had a measurable effect on the pooled group means. Related to the small sample size, sample selection bias could also impact the results. Overestimation has been observed when gathering data from both small-scale fishers (O'Donnell et al. 2010) and scientific experts (Burgman et al. 2011; Oedekoven et al. 2015). While the 4-step elicitation method I employed is more likely to reduce overconfidence than 3-point procedures (Speirs-Bridge et al. 2010), it is possible that an overconfident attitude towards risk influenced several of the fishers' judgements (Figure 4.2). One of the three gillnet skippers (participant L01) estimated closer to the observer data and not-for-profit employees than the other two gillnet skippers. Therefore, the overestimating gillnet skippers' judgements could also be reflective of their actual experience, given the spatially and temporally dynamic nature of turtle captures.

Participant L01 was the only skipper in the study who was a member of a bycatch reduction cooperative currently trialled in San Jose by the not-for-profit conservation organisation with whom I was working. Exposure to conservation-oriented fishing practices aimed at reducing impact to sea turtles may have increased this fisher's awareness of turtle bycatch and contributed to this participant's estimates more accurately reflecting fleet-wide capture rates calculated from the observer data.

In addition to potential biases present in the respondents' estimations, it is possible that inferences made when extrapolating the observed capture rate to the wider fleet using capture per unit effort weightings from the GLMM did not accurately approximate turtle captures across the fleet. For example, estimates could be biased or inaccurate due to the rarity of positive turtle capture events, which can be sensitive to extrapolation from low percentage coverage rates because the data are often zero inflated (Babcock et al. 2003). The GLMM focused on the potential for a deployment effect (i.e. sample selection bias) as a result of a non-random assignment of observers on vessels within the inshore-midwater fleet. Observer programmes in which participation is voluntary, such as our current case study, are often more prone to deployment biases than programmes that require vessels to routinely take on board observers when fishing licenses are issued and in which observers are randomly assigned (Borges et al. 2004). GRT was selected a priori as a good variable to account for the potential deployment bias, which can arise due to difficulty in placing observers on the

smallest vessels, varying range distributions of vessels that results in different spatial and temporal overlap with turtle species, time at sea, and weather. As expected, both green and leatherback turtle capture estimates increased slightly with the capture per unit effort weighted by GRT class, compared to a straight extrapolation of the capture per unit effort rate by month (Supporting Information). There is also the possibility of an observer effect that results from fishers changing their behaviour as a result of an observer presence on board (Liggins et al. 1997). Because this effect occurs at the vessel level it can be hard to detect, especially when modelling a small amount of observer data as in the current study. The presence of an observer on board a vessel can cause skippers to fish away from their traditional sites, modify their fishing effort, operate their gear differently, retain catch that may have previously been discarded, or release bycatch that may have previously been retained. While observer effects have been found to be more distinct in fisheries with trip quotas (Gillis et al. 1995), few studies have attempted to disentangle deployment and observer effects on monitoring fishing trips. While the GLMM helped to account for potential non-random sample selection bias (Cotter & Pilling 2007), any bias from an observer effect ultimately must be addressed during data collection rather than post hoc during data analysis (Benoît & Allard 2009).

I successfully implemented the IDEA protocol in my case study fishery system. However, protocol adaptations were necessary due to the inshore-midwater gillnet skippers rarely overlapping with one another during the few days they spent on shore during my three month survey period. The methodological modification included holding two elicitation rounds facilitated through face-to-face interviews rather than a face-to-face group meeting or over email or web forum. Participants were provided with comprehensive comments and questions from the other participants, both between Round 1 and Round 2, and after Round 2, on printed paper in their native language (Spanish) and they then discussed these verbally with the facilitator (BIE). Continual discussion about specific questions was restricted as a result of the modified format. In addition, the gillnet skippers interviewed were not comfortable writing their responses, preferring to have the questions read aloud, followed by discussion of potential misinterpretation, verbally noting their answer, and then asking the facilitator to record their response. This may be due to some of the gillnet skippers in our case study fishery having difficulty in reading and writing. Scenarios preventing group meetings can be numerous in the field and while far less than ideal, the notes and facilitator were able to assist in clarifying uncertainties or misinterpretations held by respondents. Request were

made to record interviews, and respondents were encouraged by the facilitator to provide comprehensive explanations for their reasoning behind each estimate, so that others can understand the knowledge and rationale behind each respondent's estimate to the maximum extent possible and therefore better weigh it against their own. While the IDEA protocol is simple to understand and I was able to undertake it in this individualised way with resource users in our case study system, further investigation into possible local resource user-specific adaptations to modern elicitation protocols would be a beneficial area of future research.

This research has applied the IDEA protocol in a new context of conservation research and natural resource management, to estimate the total number of green turtles captured in a small-scale gillnet fishery and compare these estimates to capture rates calculated from observer data obtained from the same fleet. The analysis reveals high green turtle capture rates in the San Jose inshore-midwater gillnet fleet. I demonstrate that the IDEA protocol can be implemented to quantify uncertainty and control for personal biases and heuristics when interviewing respondents in small-scale fishing systems, and highlight that both observer data and elicitation estimates are approximations of an unknown truth. While the IDEA protocol was implemented successfully, minor methodological modifications were necessary to obtain participants' judgements. Future research could investigate how best to adapt the protocol to a range of local resource user contexts. Furthermore, comparing elicitation estimates to an observed value obtained from an observer programme provided informative data on participant performance when combined with a bootstrap hypothesis testing of means analysis. I encourage researchers and practitioners implementing elicitation studies with local resource users to draw on multiple sources of comparable data.

Chapter 5

Understanding the potential for information spread about bycatch reduction initiatives in small-scale fisheries

This chapter will be submitted to *PNAS*. The *PNAS* manuscript structure of introduction, results, discussion, methods have been maintained.

5.1 Introduction

The conservation and management of common-pool natural resources such as fisheries often involve behaviour-change interventions with resource users. These include the enforcement of rules, social marketing, and education campaigns (Dietz et al. 2003; Gutiérrez et al. 2011). Yet unlike other behavioural-change disciplines such as development studies, public health, and marketing, it is only recently that emphasis has been placed on understanding the social structure of the communities targeted for conservation interventions to predict how information flows through these networks (de Lange et al. 2019; Groce et al. 2019). Indeed, the degree to which information flows and behavioural changes propagate through networks is strongly dependent on their underlying structure (Watts & Strogatz 1998; Centola & Macy 2007; Centola 2018). This understanding can be crucially important in guiding dissemination of conservation messaging and the allocation of limited resources to interact with particular people to support the targeted uptake of more sustainably orientated behaviours (Barnes et al. 2016; Isaac & Matous 2017).

Social network analysis provides a robust analytical approach to quantify individual social interactions, assess the emergent structure, and potentially measure social processes (Brockmann et al. 2006; Prell & Bodin 2011). Within the context of conservation, it has

already proven useful for considering the establishment of common rules and norms among stakeholders (Meek 2013), enhancing conflict resolution strategies (Bodin & Crona 2009), potentially accelerating behavioural change (Matous & Wang 2019), and for identifying possible key players in communities in relation to specific conservation objectives (Nuno et al. 2014; Mbaru & Barnes 2017). Social network analysis can provide a number of individual-level social metrics and network-level metrics that describe how nodes (e.g., individuals, groups, communities) are socially interconnected to one another (with links representing relationships or interactions between nodes), that allow for associated statistics to be generated (Wasserman & Faust 1994). Individual-level social metrics can be considered simple (i.e., direct) or complex (i.e., indirect). Simple social metrics quantify an individual's own social associations to those they are directly connected to in the social network (e.g., at the dyadic-level, such as a conversation between two people), whereas complex social metrics look at higher-level network structure such as through indirect network links (e.g., 'friends of friends'). For example, when considering centrality measures, the number of links ('degree') is an individual-level, simple social metric, whereas how many 'degrees of separation' an individual is from the furthest other ('eccentricity'), and the ability for a node to bridge the network ('betweenness centrality') are individual-level, complex social metrics. Network-level metrics can also be calculated to inform about network structure. For example, the propensity for nodes to be linked to other nodes of similar centrality ('network assortativity'), or variance in individual centrality.

A potentially inaccurate assumption when analysing social network data in conservation science and natural resource management is that knowledge of the structure of the network (i.e., which individuals are socially linked to one another, and who may share information), is consistent across different contexts. This inconsistency implies that the social links measured in one information-sharing context will also be important for spreading the conservation information of interest in another closely related context. For example, it is understandable to assume that information shared between fishers about fishing would be predictive of a finer-scale yet closely related environmental outcome such as shark bycatch (Barnes et al. 2016). To date, little investigation into the fine-scale structural differences between information-sharing networks has been undertaken in conservation science and natural resource management (Groce et al. 2019). This absence may, in part, be due not only to the difficulty of collecting information-sharing data across multiple contexts. However, it is often analytically challenging to address how social networks differ from one another across

contexts over and above simple social metric, such as the total number of links, or the distribution of individuals' degree due to simple underlying differences or observational factors.

The use of network null models (permuted versions of the empirical datasets against which the observed dataset can be compared) is a method commonly applied when investigating hypotheses in datasets in which control groups are difficult to establish, exogenous treatments are unavailable, or observations may be missing or biased (for instance, particularly in non-human, animal network datasets (Whitehead 2008; Croft et al. 2011; Farine & Whitehead 2015). Null models allow the generation of expected patterns from the data in the absence of the process of interest, using routines that can include simulations of newly generated data based on the observed network, or more commonly, shuffling existing data to create expectations of randomised networks given specified constraints (permutations; Figure 5.1). Analysing social network data using network null models offers an ability to investigate multiple cross-contextual comparisons of information across a network, and can even elucidate the potential social causes of such differences (Firth & Sheldon 2016). There is significant potential for using null models with human social network data. For example, to account for sampling and observational biases that may emerge during data collection (Gavin et al. 2010). Despite this, the use of network null models to test hypotheses in human networks is currently limited (but see, for example, Newman and Park 2003 who apply null models to a network of board directors), and almost entirely absent in conservation science. Therefore, the potential of social network analysis to give insights into the structure and cross-contextual associations within human social systems, and whether these approaches provide useful information over and above the general structure of the network or simply knowing simple social metrics (such as the number of links each individual has), remains unexplored in conservation.

In fisheries, bycatch is the non-target portion of the capture that is discarded dead or injured to an extent that death will result, and may include species of conservation concern (Hall 1996). Bycatch and the incidental take of marine megafauna species, which are defined as large-bodied ocean dwellers like sea turtles, seabirds, marine mammals and sharks, remains as one of the most significant fishery issues in the world (Gray & Kennelly 2018). In small-scale fisheries, which encompass traditional, low-technology, low-capital fishing methods, bycatch poses unique problems and requires unique solutions because management and monitoring regulations are frequently underdeveloped, unenforced, or non-existent (Berkes et

al. 2001). Where formal institutional capacity is lacking, an effective strategy can be for fishers to partner with not-for-profits or state agencies in community co-management schemes (Cleaver 1999; Gutiérrez et al. 2011). In Peru, small-scale gillnet fisheries take significant amounts of protected species bycatch (Alfaro-Shigueto et al. 2011; Alfaro-Shigueto et al. 2018), with the critically endangered East Pacific population of leatherback sea turtles *Dermochelys coriacea* of particular conservation concern (Alfaro-Shigueto et al. 2007).

San Jose is a coastal fishing community in northern Peru with problematic turtle bycatch where a local not-for-profit currently undertakes a trial community co-management bycatch reduction scheme (Arlidge et al. 2020). This initiative intends to create direct incentives for bycatch reduction by giving price premiums to fish caught by vessels which follow best-practice bycatch reduction guidelines. Timely bycatch information is conveyed to fishers. The not-for-profit has a vision of expanding the community co-management scheme, first to more fishers within the target community, and second to similar communities along Peru's coast. This expansion could be more cost-efficient if the not-for-profit better understood how messages about the existence and aims of the bycatch-reduction initiative might spread.

With this aim in mind, I sought to better understand the social network of the San Jose coastal gillnetting community. I conducted a census survey of gillnet skippers in our target port, to elucidate the social network structure across a range of different information contexts that relate to fishing. The interventions trialled in our case study system are primarily intended to reduce the number of sea turtles captured in gillnets; therefore information-sharing about turtle bycatch is the study's primary interest. Bycatch reduction initiatives also potentially add value to catch (fishing finance), as well as relating to other fishing-related information shared, such as fishery regulations, vessel technology and maintenance, weather conditions, and crew management (Table 5.1). As such, I evaluate whether networks of information-sharing about turtle bycatch are structurally similar to networks for other information that relates to fishing.

In this study, I assess the structure of information-sharing networks across nine fine-scale contexts that relate to fishing, including turtle bycatch. Thus, I test the assumption that knowledge about other information-sharing social network contexts should be transferable to a related information-sharing context of interest (other fishing issues and turtle bycatch, in our case). I illustrate how null model analysis techniques may offer deeper insights into the fine-

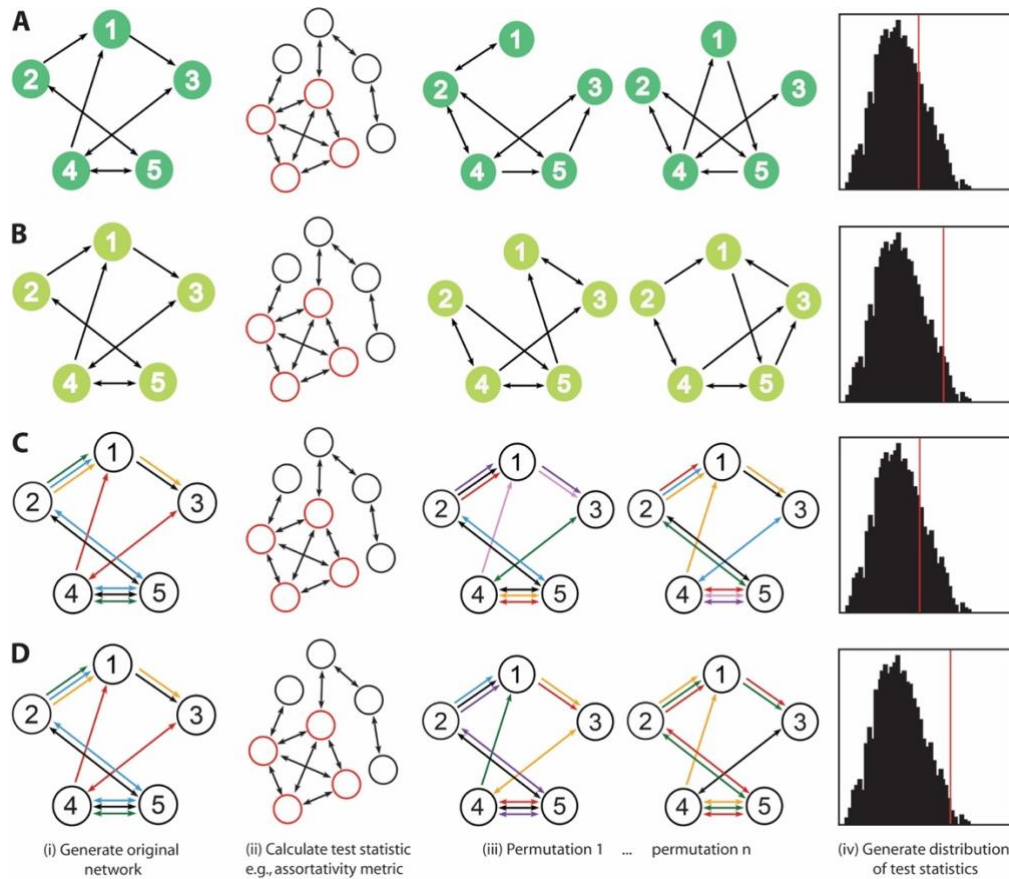


Figure 5.1. Schematic representation of edge-based permutation models with directed network data. Four main null model steps include (i) creating a social network from the observed data, (ii) calculating a test statistic, for example, a network-level metric like assortativity (high-degree nodes that are coloured red primarily connect to other high-degree nodes), (iii) randomising the observation data (typically with 1000 permutations), and (iv) recording the distribution of possible test statistics. Conclusions can then be drawn by comparing the observed test statistics to the distribution test statistics, and the P -value calculated. Throughout the edge swap permutations, the node positions remain the same, but the configuration of edges between nodes change based on select criteria. The four null model examples shown are all used in this paper's analysis. Edge permutation (A) allows the randomisation of all in-going links, whilst maintaining the number of nominations (out-going links) each individual made, (B) only allows the swap of links, by maintaining the number of nominees (in-going links) and nominations (out-going links) each individual made in this information-sharing context. The context permutation (C) maintains each dyadic nomination, but randomises the contexts that these nominations were made in (i.e., when individual X nominated individual Y for information sharing within three different contexts (represented by different coloured arrows), the context permutation allows these three nominations to be reassigned to any of the nine possible contexts), and (D) maintains each dyadic nomination, but randomises the contexts that these nominations were made in, while also controlling for the number of nominations that took place overall within each context (i.e., when individual X nominated individual Y for information sharing within nine different contexts, these three nominations were reassigned amongst the contexts in a way that was equal to the number of nominations in each context).

scale structure of social systems than could be gained from the use of simple centrality measurement methods, and provides new insight into comparing information-sharing networks within a social system of high conservation interest. Furthermore, I discuss these findings in relation to further research priorities in the use of network analysis for conservation science, particularly when predicting how new information and behaviours may spread socially.

Table 5.1. Fine-scale information-sharing contexts that relate to fishing.

Full name	Short name	Description	Broad categorisation
Turtle bycatch	T.Bycatch	Turtle bycatch encounters including live releases and mortalities in nets.	Process of fishing / Business and governance of fishing
Gillnet type & maintenance	Gear	Changes made to net configuration (shifting rigging configurations from surface drift net to mid-water drift net or bottom-set net), and net maintenance.	Process of fishing
Weather conditions	Weather	Ocean and weather conditions (e.g., wind, swell).	Process of fishing
Fish location & catch sites	Location	Where fish might be located and where they have been travelling to fish.	Process of fishing
Fishing activity	Activity	How many people fishing, who is fishing, who caught what.	Process of fishing
Vessel technology & maintenance	Tech	Existing and new technologies used on board the vessel (e.g., echo sounder, compass) and vessel maintenance (e.g., hull repairs, painting).	Process of fishing
Fishing regulations	Regs	Fishery policy and legislation.	Business and governance of fishing
Fishing finances	Finance	Market prices, loans, fines, penalties.	Business and governance of fishing
Crew management	Crew	The hiring and instructing of crew on board the vessel.	Business and governance of fishing

5.2 Results

I collected network data concerning fishing-related information-sharing for the gillnet skipper community in San Jose, Lambayeque, Peru (6°46' S 79°58' W) (see Methods). Our census

survey of $n=165$ represented 98.2% of the total gillnet skippers launching vessels from the case study port between July to September 2017; only 3 skippers declined to be interviewed (Figure 5.2a and table A3.1.1). As this study aimed to investigate the social structure amongst skippers, the study's data pertain to respondent-to-respondent networks only (which also allowed for consistent respondent numbers between network cross-contextual comparisons). Of the 165 skippers surveyed, 151 nominated at least 1 gillnet skipper from San Jose as one of the 5–10 people that they talk to most about fishing success, while 116 of the 165 respondents were nominated at least once by other respondents. This resulted in a total of 427 respondent-to-respondent nominations for one context or more (table A3.1.1). Respondents nominated between 1 and 8 other respondents (mean 2.8 outgoing links), for one or more information-sharing contexts. The average number of information-sharing contexts per nomination was 7.7 with a range of 1–9 (out of 9 contexts for which nominations were sought). Respondents received 1–15 nominations by other respondents (mean 3.7 incoming links), for one or more information-sharing contexts. Across the 9 different information-sharing contexts evaluated (Table 5.1), turtle bycatch was discussed between skippers the least often (in 61.6% of possible respondent-respondent links), whereas skippers most discussed fishing location and fishing activity (both in 97.9% of possible respondent-to-respondent links; Table 5.2).

The wider network of non-skipper outgoing links was not the primary focus of the current study, however, our analysis showed that the number of information-sharing links remained consistent between the respondent-to-respondent network and the wider network that includes non-skipper nominees. Across nine different information-sharing contexts evaluated, turtle bycatch remained the least discussed type of fishing information in the wider network (in 64.2% of possible nominations). Information about the weather and fishing activity were discussed the most (with 95.7% and 95% of possible links, respectively). Turtle bycatch and fishing regulations were the only two contexts that had a relative increase (both by 3%) in the amount they were discussed in the wider network, compared to the respondent-to-respondent network that contained only skippers (Table 5.2).

Table 5.2. Respondent-to-respondent network summary statistics. Respondents nominated 5 to 10 individuals that included other skippers in their community but also non-skipper community members that might be deemed valuable to their fishing success. This study only analysed respondent-to-respondent data, but the full network links across contexts are included in table section B.

(A) Respondent-to-respondent network data

	Number
Total no. of links across all contexts	3720
Total no. of links of one or more context	427
Total no. of eligible respondents for survey	168
Total no. of respondents surveyed	165
Total number of contexts	9
Mean number of contexts nominated per nominee	7.7
Mean incoming links of one or more context per respondent	3.7
Mean outgoing links of one or more context per respondent	2.8
Range of contexts nominated per nominee	1 to 9
Range of outgoing links of one or more context	1 to 8
Range of incoming links of one or more context	1 to 15

(B) Links across contexts

	Resp-resp	Full network
All	427	1102
Fish location & catch sites	418	1033
Fishing activity	418	1047
Weather conditions	415	1055
Gear type	411	1029
Fishing finances	411	1020
Captain hiring crew and managing them	342	868
Vessel technology & maintenance	311	807
Fishery regulations	304	822
Turtle bycatch	263	708

5.2.1 Structural differences between information-sharing contexts

I separately assessed the observed assortativity and node-level centrality of the turtle bycatch information-sharing networks and each of the other contexts of information sharing (Table 5.3). Across these contexts, I compared how the observed statistics differed from edge-

permuted versions of themselves (see Methods). I considered the observed statistic to be significantly different from that expected under the null models when it fell outside the 95% range of the distribution of the statistics generated by the permutations (i.e., equivalent to significantly different at $p < 0.05$ level in a two-tailed test).

5.2.2 Assortativity

For each of the information-sharing contexts, assortativity (propensity for a respondent to be connected to others who are similarly (dis-)connected) was examined, as this is a primary structural component of the network (Newman 2002; Newman 2003; Table 5.3). Social networks often show assortativity, which social factors often explain as age, language, race, or group size (Newman 2003; Firth et al. 2017). Further, the level of assortativity in a network is known to have important social implications, ranging from shaping which individuals interact (e.g., fishers peripherally positioned in a positively assorted network may only interact with other peripherally positioned fishers), to the operation and emergence of competition and cooperation (e.g., highly connected fishers may work together in a local fishing group), and the potential for simple contagions such as disease or information to spread given its starting point (e.g., if information about a bycatch reduction initiative is seeded with a well-connected fisher who is in close contact with multiple other well-connected fishers, then that information may flow more rapidly through the network than it would if it was seeded with a fisher with few social links on the networks periphery (Flack et al. 2006; Pastor-Satorras et al. 2015)).

I found that networks of turtle bycatch information-sharing nominations show no significant assortativity in comparison to the edge permutation null models (Observed stat: 0.038, edge null model 1: mean $\pm$ SD = -0.005 ± 0.059 ; $p = 0.512$, edge null model 2: mean $\pm$ SD = -0.011 ± 0.059 ; $p = 0.39$). As such, there was no evidence for a non-random tendency for highly nominated nodes to be disproportionately connected to other highly nominated nodes, nor for rarely nominated nodes to be disproportionately connected to other rarely nominated nodes. The turtle bycatch information-sharing networks differed markedly in this regard from all of the other information-sharing contexts' networks (Figure 5.2b), all of which had significantly higher assortativity scores than expected from edge permutation null model 1. In addition, all the other information-sharing contexts' networks had significantly higher assortativity scores than expected from edge permutation null model 2 apart from the 'weather' and 'technology' contexts which fell outside the top 5% of the null network

assortativity coefficients but were not significantly different in the two-tailed test (edge permutation model 2 two tailed $p=0.06$) (Figure 5.2c).

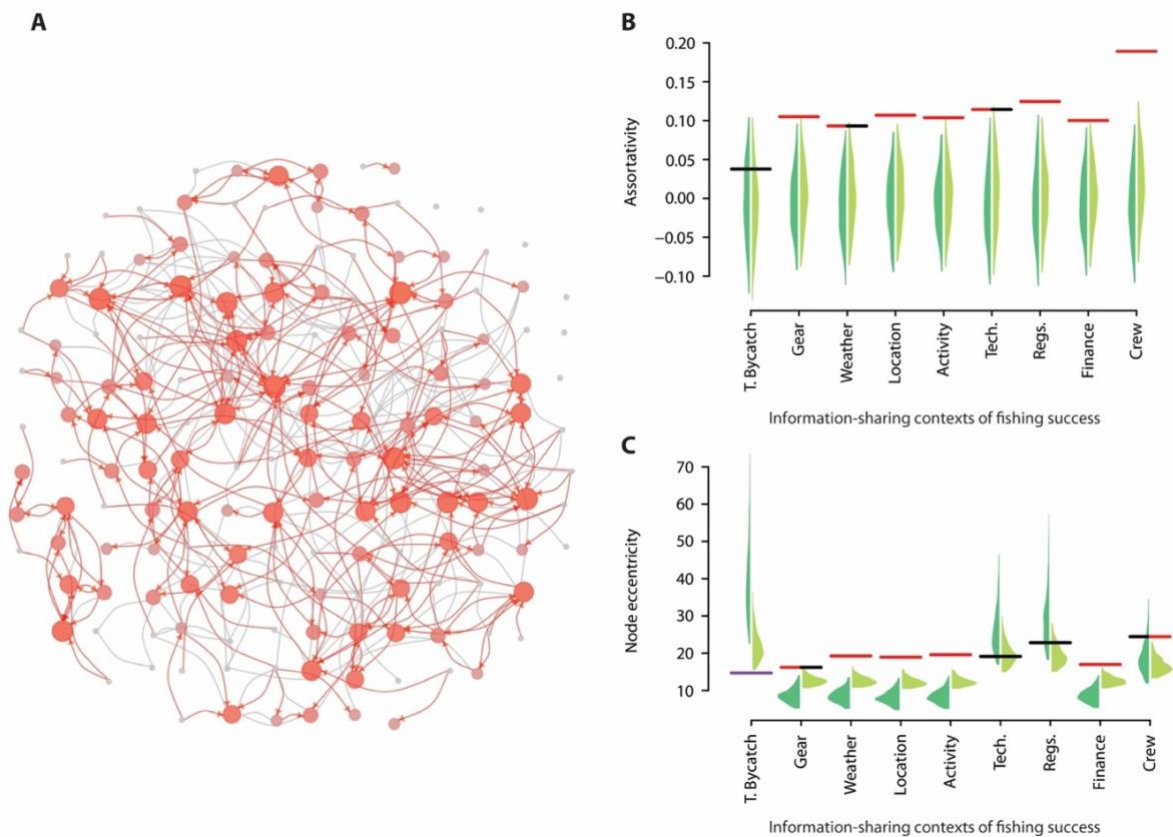


Figure 5.2. Structure of information-sharing in relation to turtle bycatch. **(A)** Illustrative network of the structure of information-sharing in relation to turtle bycatch. The nodes show each of the skippers and the adjoining lines show which dyads shared information in at least one context, and nominations within the turtle bycatch context is highlighted as a directed red arrow here (arrow points to the one that was nominated). Node size and shading shows the number of nominations each individual received for turtle bycatch information (largest and most red = most nominations, small and grey = no nominations). Layout was set as a spring layout of edges across any context (to minimise overlap) and then expanded into a circular setting. See Figure A3.1 for illustrative comparisons across contexts. **(B)** The observed In-Assortativity in comparison to the null distributions for the different information-sharing networks, and **(C)** the observed variance in the node eccentricity in comparison to the null distributions for the different information-sharing networks. Horizontal lines show the observed values from the actual networks (red = observed values are above the permutations, black = observed values are within the range of the permutations, purple = observed values are below the permutations). Polygon distributions show those generated by permutations (dark green = outgoing edge permutation that maintains the no. of nominations each individual makes, light green = edge swap that maintains the no. of nominations each individual makes and also the number of times each individual was nominated). Due to differences in network factors, direct comparisons between the observed values are not informative. For details on information contexts refer to Table 5.1.

5.2.3 Individual Centrality

The centrality of a node within a network represents how ‘connected’ or ‘important’ that particular node is, and therefore such metrics often gauge a node’s potential influence in a range of processes like increasing the spread of social contagions such as information or disease (Borgatti 2005). When considering the structure of the network as a whole, the variation in the centrality of the nodes within it is therefore very important (Amaral et al. 2000; Albert & Barabási 2002). For instance, if all nodes were of similar centrality then the network would hold a relatively uniform structure. In this case, the removal of a node would be likely to affect the overall structure similarly regardless of which node is chosen. Also, the spread of a new piece of information or disease would likely to take place at a similar rate regardless of the initial starting node. In contrast to this, if nodes show high variation in their centrality, the removal of one of the more central nodes would likely affect the overall structure more than removing a more peripheral one, and new information would more likely spread more quickly if this began in one of the central nodes in comparison to if it began in the peripheral nodes (Christley et al. 2005).

The variance in node centrality, considered for each information-sharing context network, provides a particularly informative and intuitive network measure in regards to the uniformity of the structure, its resilience to perturbations, and the influence of start-points on social contagions (Freeman 1978; Borgatti 2005; Borgatti et al. 2006). I aimed to consider node-level properties that depend on the structure of the social network whilst controlling for these simple characteristics (Table 5.3). Within networks of information-sharing, the furthest network distance between a node and all other nodes in the networks determine the maximum possible number of steps that a piece of information takes to reach a node. This node centrality metric is referred to as node eccentricity (Hage & Harary 1995). I found that sharing of information regarding turtle bycatch had significantly lower variance in node eccentricity than expected under the null models controlling for simple properties such as number of nominations and degree distributions (Observed stat: 14.71, edge null model 1: mean $\pm$ SD = 41 ± 13.5 ; $p < 0.01$, edge null model 2: mean $\pm$ SD = 22.66 ± 5.335 ; $p < 0.05$). Importantly, turtle bycatch information sharing was again unique in this sense (Figure 5.2c), as none of the other information-sharing contexts were significantly lower than expected under null permutations of themselves (table A3.1.2). In fact, six of the eight other contexts showed significantly higher variance in eccentricity than expected from a null model of their

own structure, which illustrates a particularly stark contrast from the turtle-bycatch information-sharing network. This means that across the gillnet skippers, there is less variation in individuals' centralities than expected in terms of turtle bycatch information sharing. In other words, gillnet skippers are more similar in how they share information about turtle bycatch with one another than expected, whilst this is not true for any other contexts of information sharing. This conclusion also held when considering other measures of centrality. For supplementary information, I examined the variance in betweenness (as an alternative measure of centrality; Figure. A3.1.3) and mean eccentricity for each network's nodes (rather than the variance; Figure. A3.1.4). I also investigated the observed variance in node eccentricity in comparison to the null distributions (generated from the context permutations; Figure A3.1.5) and the observed mean node eccentricity in comparison to the null distributions (Figure A3.1.6; see Supplementary Materials).

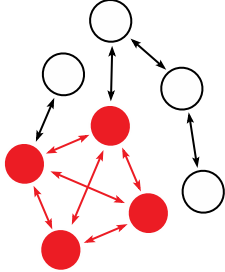
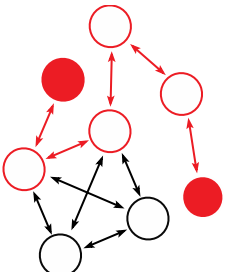
The findings demonstrated that the conservation-relevant context of primary interest (the turtle bycatch information-sharing context) generally held some structural dissimilarities to all the other contexts of information sharing.

5.2.4 Cross-contextual correlations of dyadic links

If individuals' social behaviour remains consistent across different aspects of their social lives, in terms of which individuals they form links with and the number of links they form, then the social networks across these contexts are expected to be correlated (Barrett et al. 2012; Firth & Sheldon 2015, 2016; Dunbar 2018). As individuals that share information to a particular context may also be more likely than a non-connected dyad (i.e., two skippers that know each other versus two that do not know each other) to share information about a different context, I expected that information-sharing networks across the assessed contexts would be correlated. Certain contexts may be strongly correlated to one another, however, whilst other contexts may be less correlated.

Respondents in our survey were asked to nominate individuals that they exchange useful information with about fishing and that they considered valuable to their fishing success. Respondents were then asked which contexts of information-sharing they talk to each other. I expected that information-sharing networks across the assessed contexts would be correlated with one another, assuming that dyads (pairs of skippers) which share information within a certain

Table 5.3. Network metrics used to assess information-sharing network structure. For network structure, red nodes (circles) and links (arrows) outline the represented metric in the network.

Metric	Network structure	Definition	Theoretical use in conservation-relevant systems	Example
Assortativity		A preference for nodes to attach to others that are similar in some way (e.g., high-degree)	Identifies individuals and pathways of individuals that could facilitate widespread diffusion of information about conservation initiatives in a community of conservation interest.	The authors use simulations of animal data to assess how variation in simple social association rules between individuals can determine their positions within emerging social networks. The results show that simple differences in group size cause positive assortativity and that metrics of individuals' indirect links can be more strongly related to underlying simple social differences than metrics of their dyadic links.
Node eccentricity		The furthest network distance between a node and all other nodes in the networks. The equivalent to the inverse of some definitions of 'node closeness'	Can inform whether or not information relevant to a conservation initiative is shared in an even or clustered manner throughout a community on interest. This can inform how social norms and personal beliefs might affect information flow, which in turn can allow for conservation practitioners to tailor interventions to particular perspectives about a harmful activity (e.g., bycatch).	Using social network analysis and several centrality measures including 'node closeness' (also equivalent to the inverse of some definitions of 'node eccentricity') the authors assess the structural nature and expanse of climate-based communication between professionals across sectors in the Pacific Islands region. Their results show a simultaneously diffuse and strongly connected network, with no isolated spatial or sectoral groups. The most central network members were shown to be those with a strong networking component to their professions.

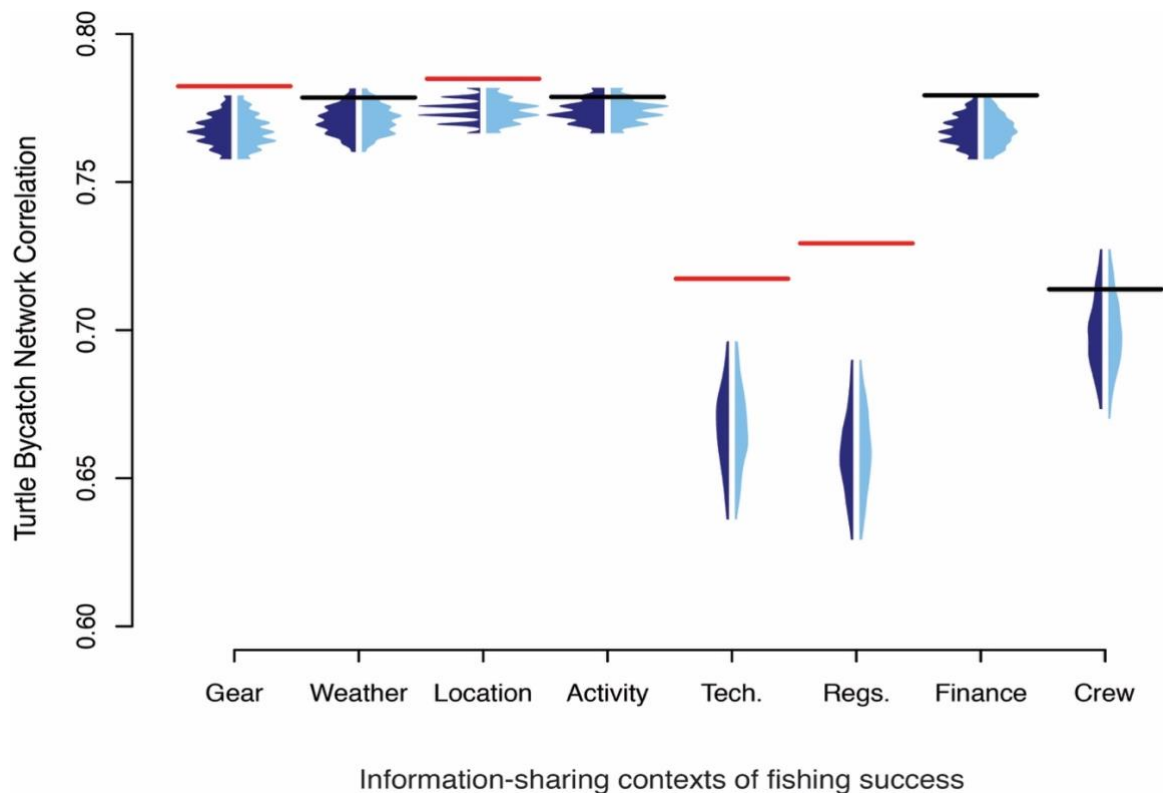


Figure 5.3 The observed correlation (and the correlations expected under the null models) between the turtle bycatch information-sharing network with all the other information networks. Horizontal lines show the observed values from the actual networks (red = observed values are above the permutations, black = observed values are within the range of the permutations, purple = observed values are below the permutations). Polygon distributions show those generated by permutations (dark blue = context swap that maintains the no. of nominations each individual makes and also the number of times each individual was nominated, but swaps the context these were made within whilst maintain the number of times each context was nominated as overall, light blue = conservative context swap that is the same as dark blue, but also maintains the number of contexts each dyad nominated each other for – but changes those contexts (same as a gbi permutation but on the dyad-by-context edges). Comparison between context can be made by comparing the distance between the observed values from the actual networks (horizontal lines) and their associated permutation distribution (polygon) to the distance between the observed and associated permutation for each context. Due to differences in network factors, direct comparisons between the observed values are not informative. For details on information contexts refer to Table 5.1.

context would be more likely to also share information in another context. As such, I expected all the other contexts to significantly predict information-sharing within the context of particular interest (turtle bycatch information). Indeed, turtle bycatch information-sharing networks were significantly correlated with all other contexts (unfolded corr; $r = >0.7$; standard $p < 0.01$). I also tested this observed correlation against that expected under the general social structure (context null model 1 - who gains information from whom overall;

Figure 5.1c) as well as controlling for the probability of nomination within each context (context null model 2; Figure 5.1d). Under these null models, I found that the dyadic directed links within the turtle bycatch information-sharing network were significantly more correlated with four other contexts of information sharing (regarding gear, locations, technology, and regulations – see Table 5.1) than expected under the general social structure (Figure 5.3). Although the turtle bycatch information-sharing network held the highest raw correlation with networks of information regarding fishing locations (unfolded corr; $r = 0.78$), the largest difference between the correlation expected under the null models and the observed correlation was with information sharing regarding fishing regulations (unfolded corr; $r = 0.78$; mean expected corr context null model 1 $r = 0.65$, mean expected corr context null model 2 $r = 0.65$), suggesting that the fishing regulations context was particularly predictive of turtle bycatch information links given the underlying social structure of the system.

5.3 Discussion

By combining a fine-scale survey of a small-scale fishing community with network null models I reveal that information-sharing networks about an issue of conservation concern (turtle bycatch) are structurally dissimilar from other information-sharing networks (Figure 5.2), more so than expected by simple differences in an individual's degree (how many people they are connected to). I also demonstrate that certain contexts can still be predictive of how information about turtle bycatch is shared between fishers, even more so than expected under the nomination structure of who nominated whom (Figure 5.3).

5.3.1 Structural differences between information-sharing contexts

I found that the turtle bycatch context did not show any assortment despite the positive assortment patterns across all other information-sharing networks (Figure 5.2b and table A3.1.1). This indicates that the usual mechanisms that drive assortment in the other contexts (and potentially social networks generally) are not at play in the turtle bycatch information-sharing network (Newman 2002; Newman 2003). This may suggest that a strategy of targeting high-degree nodes (individuals who share information with many others) with information about bycatch reduction may not be as effective as it would be in the other information-sharing networks (Figure 5.2b). When individuals interact with people who are similarly connected to them (assortment), they interact with individuals with access to similar

social resources. This is thought to be important for the spread of social contagions like information (Pastor-Satorras et al. 2015). For example, often conservation organisations can easily observe who appears to be generally well connected within a given social circle (such as skippers at a port), and who does not. If organisations wish to seed information about conservation interventions (e.g. turtle bycatch reduction strategies), they may start by discussing it with these well-connected people, on the assumption that they will be the most likely to pass it on to others who are also well connected. In our setting, the benefit from taking such an approach would not be as great as expected, because of the non-assortativity identified in the turtle bycatch information-sharing network through the use of the null model analysis. This non-assortativity illustrates the need to narrow down the context, rather than assuming that those who are well connected in one context are the best messengers for other contexts.

I also found that the turtle bycatch information-sharing network has less variance in node centrality than expected, i.e., a more uniform individual-level network structure (Figure 5.2c and Table 5.3). The low variance in node eccentricity indicates that the turtle bycatch network has a more homogenous network structure in contrast to the other contexts (and many observed social networks, where high variability in node centrality is common and can result in high-degree nodes forming (Watts & Strogatz 1998; Albert & Barabási 2002). This indicates that information about turtle bycatch will have less variation in the rate of diffusion throughout the San Jose skipper community, regardless of which skipper first started talking to other skippers in the community about the capture, compared to information-sharing in a network with higher variance in node eccentricity. For example, the weather, fishing locations, fishing activity, and finance.

As an addition to the above points, our findings of less variance in node centrality (Figure 5.2c) and less variance in mean eccentricity (Figure A31..4) in the turtle bycatch information-sharing network were also found when comparing to the context null models (Figure A3.1.5 and A3.1.6). This lower variance shows that the variance and mean eccentricity is lower than expected not just in comparing to the edge null models, but also lower than expected given the underlying social structure of who is connected to whom. This lower variance found when comparing to the context null models reinforces the hypothesis that the fine-scale structure of the network (beyond who talks to whom) is contributing to these patterns. For example, certain personal traits that skippers hold, such as whether or not

they would be willing to work with a local not-for-profit organisation to implement bycatch reduction strategies on their boats in future, may be contributing to skipper centrality within the network. This demonstrates a particularly interesting use of comparing results across various null models which randomise different processes (Croft et al. 2008; Firth et al. 2015). This information could be used to guide research into the underlying differences in attitudes, beliefs and knowledge between skippers. This in turn could inform the design of interventions which could be more tailored to particular perspectives about bycatch reduction and sustainable fishing practices more widely.

5.3.2 Cross-contextual correlations of dyadic links

Understanding correlations between networks allows for the assessment of skipper-to-skipper (dyadic link) information-sharing differences between multiple networks. Insight into these differences helps with identifying social contexts suited to conservation interventions, and more broadly, offers insight into the generalisability of network research (Matous & Wang 2019).

Using null model network-based approaches I demonstrate that across all contexts, the fine-scale structures of our information-sharing network are more similar than otherwise expected based on the number of links or even who is linked to whom. While this similarity provides assurance that in San Jose's gillnet skipper network, knowledge about a social network based on general information spread should be transferable into understanding how novel information should spread, the similarity also demonstrates that relying on simple network measures without the use of null model comparisons could potentially result in an improper assessment of network structure. I also show, through the use of null model permutations, the contexts that are most closely related to the specific context of conservation interest, offering greater understanding of how information flows relevant to the wider topic of information-sharing about fishing are structured and relate to one another (Figure 5.3). Both these points support the value of conservationist investing time and resources in more robust and comprehensive null model network-based analyses when gathering and assessing network data. More specifically, our results indicate that the fishing regulations network, followed by the vessel technology and maintenance, fishing gear, and fishing location networks, are more correlated with the turtle bycatch network structure than expected under the context null models (Figure 5.3). This finding gives insight into how fishers perceive

information relating to turtle bycatch. For example, the correlation between turtle bycatch and the fishing regulation network could be because fishers perceive turtle bycatch as something by which they must abide, similar to fishing regulations (related to the business and governance of fishing). This correlation is supported by supplementary structural analysis that shows that the turtle bycatch and regulation contexts are structurally dissimilar in relation to node variance to all the other contexts of information sharing (Figure A3.1.3). While these results begin to provide a more in-depth insight into how turtle bycatch information-sharing relates to other fishing-related information contexts and is perceived by fishers, further exploration is needed to determine the process underlying the structural differences identified.

5.3.3 Further work applying null models in social networks

Further research could investigate the drivers of the differences between turtle bycatch networks and other networks. In other words, how might individual-level traits influence the network structure of the information-sharing contexts? Structured network experiments offer significant potential in this area of research. To illustrate, studies in behavioural sciences manipulated the group structures of natural animal populations (by controlling individual access to experimental feeding stations) and demonstrated that experimentally imposed constraints carried over into patterns of association in other contexts (unrestricted, ephemeral food patches, nesting sites, and information transmission), hence causally linking different social contexts (Firth & Sheldon 2015; Firth et al. 2016).

Similar experimentation could be designed in the context of fishers and bycatch. For example, communication between particular skippers could be encouraged, and the resulting implications for information transfer subsequently assessed. Alternatively, half of the gillnet skippers in a fishing community could be targeted with information about the benefits of a turtle bycatch reduction initiative, and the network then resurveyed to measure whether the structure of the bycatch information-sharing network shifted to greater similarity to other fisheries contexts, which ones, and why?

Another major research need is understanding whether skippers individual attitudes and behaviours govern the social network links, or whether the social network positions govern individual skippers propensity to adopt positive attitudes and behaviours towards turtle bycatch? If the former, then interventions that seek to change the group-level social norm about bycatch could be effective as a way of bringing peripheral fishers into the

intervention, for example, through campaigns to engender pride in conserving turtles. Whereas if the latter is true, then interventions that seek to improve communication between peripherally located and more central fishers could be a more effective strategy; for example, setting up networking events to talk about the issues around bycatch. Therefore, further research that assesses certain nodes' network positions, followed by implementation of interventions in an experimental setting, and then reassessment of nodes' network positions would be informative. Lastly, research into the effectiveness or relationship between economic incentives versus social norms and social networks is required to improve understanding of how intrinsic and extrinsic motivations interact to shape behavioural intentions and decision making.

5.3.4 Conclusion

I quantified the underlying structure of a small-scale fishery social system across various contexts relating to fishing, and demonstrated how networks of information-sharing regarding a conservation-relevant topic (turtle bycatch) are structurally dissimilar from other social contexts that relate to fishing, and the extent to which dyadic links can be non-randomly predicted from other information-sharing networks. The results show how null models allow identifying the extent of structural differences, and provide information about which other contexts are best correlated with the conservation-relevant information sharing. Our findings highlight the need for further research on the use of network null modeling and cross-contextual comparisons to understand how information relating to a planned conservation intervention may spread through a network, which in turn could help inform messaging about conservation interventions that are salient to community members targeted for conservation interventions.

5.4 Materials and Methods

5.4.1 Study system

San Jose, Lambayeque, Peru ($6^{\circ}46'$ S, $79^{\circ}58'$ W) is home to 168 small-scale commercial, gillnet skippers that fish from the beach throughout the year. During months with warmer weather (and hence better fishing conditions), the number of skippers can more than double as

fishers arrive from inland areas seeking fishing work. Skippers typically operate with 1–4 crew (Alfaro-Shigueto et al. 2010).

Peruvian law defines small-scale fishing vessels as displacing a maximum of 32.6m³ Gross Registered Tonnage (GRT), up to 15m in length, and operated predominantly manually. San Jose’s small-scale gillnet vessels can be subdivided into two fleets. The first fleet comprises a class of open-welled boats known as ‘chalanas’, with a capacity range from 1–8 t. The second fleet comprises a predominantly larger vessel class known as ‘lanchas’, with small closed bridges ranging in capacity from 5–32 t Castillo et al. 2018. The survey interviewed actively fishing gillnet skippers on both chalana and lancha vessels.

The gillnet is the most common fishing gear used in Peru’s small-scale fishing fleet (Castillo et al. 2018). Many species are incidentally captured in gillnets in San Jose, including several species of turtle. Our chosen study population was actively fishing San Jose skippers deploying gillnet gear year-round, including those who owned and operated their vessels and those who skippered for others. Skippers were chosen as they are in charge of the fishing gear and crew when the boat is in the water and the gears deployed, and therefore their decisions are most influential in opportunities to reduce turtle bycatch (for example through better live release, or the use of LED lights on nets to reduce incidental captures; Ortiz et al. 2016). Skippers were deemed active if they fished from the San Jose fishing port with gillnets in the winter period of 1 July – 30 September 2017. The network was surveyed during the winter months as skippers actively fishing during these months are established fishers in the San Jose community throughout the year. I define gillnets as encompassing surface drift gillnets and fixed bottom gillnets in single or trammel net configurations. The total population (n=168) was determined using a combination of membership lists of the two main fishing groups in San Jose, lists of boats towed in and out of the water with tractors, and key informant interviews (see Supplementary Materials – Methods). Previous estimates of gillnet activity in San Jose recorded 95 gillnet vessels fishing in January–April 2004 (Alfaro-Shigueto et al. 2010), and 47 gillnet vessels fishing in November 1995–April 1996 (Escudero 1997).

5.4.2 Data collection

Social network data were collected using a structured questionnaire that I developed based on key informant interviews and relevant conservation science and social network analysis literature (Scott 2010; Barnes-Mauthe et al. 2013). Questionnaires were trialed with skippers

(n=8) in another fishing community 17 km down the coast from San Jose. Pilot study data were not included in this study's analysis. Respondents were interviewed in their native language (Spanish; Table A3.6). This research has Research Ethics Approval (CUREC 1A; Ref No: R52516/RE001 and R52516/RE002).

I surveyed with a fixed choice survey design, where respondents were asked to consider up to 10 individuals with whom they exchange useful information about fishing and whom they considered valuable to their fishing success. The decision to limit the number of skippers each respondent could specify was made for practical survey purposes as the network I surveyed was relatively large. The fixed-choice survey design also had the secondary benefit to help respondents understand what is required of them during the survey, as a free-choice survey design can result in subjective interpretations of the desired (Newman 2010). While the number of out-going links was limited to 10, there was no limit on the in-degree of links in the network (i.e., there was no limit to the number of times a skipper could be nominated by others), which was the main focus of our analysis. In classifying fishing-related information, I first classified 2 broad categories about which I expect gillnet skippers to exchange fishing related information. These include 1) the process of fishing, and 2) the business and governance of fishing. I then disaggregated these 2 broad categories into 9 fine-scale information-sharing contexts that relate to fishing, including: i) turtle bycatch, ii) gillnet type and maintenance, iii) weather conditions, iv) fish location and catch sites, v) fishing activity (how many people are fishing, who is fishing, who caught what), vi) vessel technology and maintenance, vii) fishing regulations (laws and rules), viii) fishing finances (market prices, loans, fines, penalties), and ix) crew management (Table 5.1). Respondents were then asked to highlight which context(s) of fishing-related information they discussed with each nominee. Contexts were randomised prior to interviewing each respondent using a random number generator. By investigating multiple contexts simultaneously, I had the added benefit of not letting the participant know that turtle bycatch information was of primary interest.

For each context, respondents were asked to consider people from San Jose that they share useful information about fishing with over the last 5 years; considering those that they thought may influence their fishing success. Respondents were asked to consider relationships that they have had with other vessel skippers, vessel owners, crew members, other fishery leaders, fishery management officials, members of the scientific community, boat

launching/landing support, fish sellers/market operators, family members, and any other stakeholders they fished or shared information with about fishing (see Supplementary Materials – Methods – Tables A3.2.1 and A3.3.1). Respondents were reminded that the shared information and names will remain anonymous and will not be revealed. I along with my research assistants highlighted that the information provided will help me understand how information that relates to fishing flows between fishers.

5.4.3 Determining population size

The total population (n=168) was determined by triangulating data obtained from membership lists of the two main fishing groups in San Jose, lists of vessels daily launching and landing logs, and key informant interviews. I restricted the network analysis to gillnet skippers – who owned their own vessel(s) or who skippered a vessel owned by someone else, and who launched and landed their vessels from the beach at San Jose, Lambayeque, Peru (6°46' S, 79°58' W). Gillnet skippers were required to be identified as actively fishing at least once during the winter period of 1 July – 30 September 2017 using one or more of the data sources used for the analysis.

There are two main at sea fishing groups in San Jose (the Maritime Union of Fishermen Society, and the Artisanal Fishermen and Hydrobiological Extractors Association). Following initial introductions made with both of the fishing groups leaders during which time I presented a description of the study and associated ethical clearance, I was granted access to the fishing groups membership lists, which contained information on gillnet skipper name, vessel name, and vessel unique identification (plate number). During our survey period, the fishers in San Jose were pushing and pulling their fishing vessels in and out of the water from the beach using large tractors that were driven by employees of a local company that specialised in providing this service. Subsequent information from San Jose in early 2019 indicates that this service is no longer provided due to legal implications imposed by recently implemented Government legislation. Skippers were charged a fee and the tractor drivers record each vessel (using the plate number) as they pushed each vessel out to sea and pulled each vessel back onto the beach following a fishing trip. The daily launching and landing logs were provided following a meeting with the company owner and the tractor drivers, during which time I presented the company owner a description of the study and associated ethical clearance. The daily launching and landing logs were cross referenced with

the list of active fishing group members and the list of actively fishing gillnet skippers was checked by several key informants during two key informant interviews held in San Jose in July 2017. Between 1 July – 30 September 2017 every actively fishing gillnet skipper (n=168) was identified and asked if they would like to partake in the interview; only three actively fishing gillnet skippers declined.

5.4.4 Statistical Analysis

5.4.4.1 Social network construction

A social network was created for each information-sharing context. In each network the nodes were the individuals, and the binary directed edges were the nominations by one node (sender) of another node (receiver) for this information-sharing context. All analysis was carried out in R (R Core Team 2019), with use of the `igraph` package (Csardi & Nepusz 2006) for visualising and processing the analysis, and carrying out the cross-contextual comparisons using the null models.

5.4.4.2 Structural differences across contexts

To investigate whether networks of information-sharing between individuals were similar across different contexts, I examined structural properties of the networks in terms of their assortativity and the variance and mean of individual centrality (Table 5.3). To account for the effect of basic characteristics of the contexts (number of links, degree distributions etc.) I compared these observed summary statistics to null models which allowed inference of structural differences and similarities over and above what would be expected from these simple differences using null models (Figure 5.1).

5.4.4.3 Network assortativity

The assortativity coefficient (Newman 2003) measures the extent to which central nodes are connected to other central nodes and peripheral nodes are connected to other peripheral nodes. Positive values demonstrate assortativity, with perfectly assorted networks scoring 1, and negative values representing disassortment. When nodes of similar centrality are randomly distributed in a network (i.e., fully disassorted), those networks do not always score -1 due to

the minimum value depending on the number of node types and the relative number of links within each group (Newman 2003).

For each of the information-sharing network contexts, I first calculated the assortativity by in-degree (the number of nominations each interviewed skipper received). This metric measures the extent to which ‘individuals that are highly nominated are disproportionately connected to others that are highly nominated’ and ‘individuals that are rarely nominated are disproportionately connected to others that are rarely nominated’. This is the primary assortativity measure of interest as in-degree provides the measure of which individuals provide information to others. However, as individuals differed in the number of nominations they made within each information-sharing context, I also calculated the assortativity by out-degree (the number of nominations each interviewed skipper made) to further examine whether individuals were also disproportionately connected to others who make a similar number of nominations as themselves. As social networks often show assortment by degree, I predicted that all the information-sharing networks would be positively assorted by nominations made and nominations received (i.e., highly nominating and nominated individuals would be closely associated with highly nominating and nominated individuals, whilst peripheral individuals would be more likely to be connected to one another).

5.4.4.4 Individual centrality

Across network science, various metrics measure the centrality of nodes. As such, when examining empirical networks, it is important to consider metrics which are relevant to the system. As I aimed to examine the use of social network analysis for conservation-relevant systems, I did not want to use simple node-level metrics that can be inferred without building social networks (e.g., using ‘degree’ is simply equivalent to counting the number of nominations an individual receives, and requires no knowledge of the network structure). Instead, I aimed to consider node-level properties that depend on the structure of the social network (Table 5.3). For this purpose, I used node eccentricity that measures how far a node is from the furthest other (Hage & Harary 1995). Furthermore, although this metric describes a node’s position within the wider network, the range of potential values it can take is not overly affected by permutations of the network structure in comparison to other more vulnerable metrics (e.g., betweenness, clustering) which are innately dependent on multiple

aspects of the set structure of the network and are intuitively expected to differ largely from permutations by default. Finally, this metric is also relatively fast to compute, which is particularly useful when calculating it for many iterations of null networks. As such, I computed the variation in eccentricity in ‘received nominations’ (in-eccentricity) for each of the information-sharing contexts.

5.4.4.5 Null models for structural differences

Drawing comparisons of network structure, correlations, and node positions across different networks (or networks within different contexts) requires particular consideration because the general basic structure of the network (such as number of links or degree distributions) has a large effect on the observed values obtained from standard summary statistics. This can be taken into consideration by comparing networks to null permutations (controlled randomisations) of themselves and recalculating the same summary statistics on the null networks as well. Through comparing the observed values of the summary statistics to the distribution of those statistics generated from the null networks, insight can be gained into the actual differences between observed networks across different contexts, over and above that expected from simple properties such as the number of links.

Therefore, when calculating summary statistics (assortativity, node-level centrality) of the networks in each of the information-sharing contexts, we (JAS and WNSA) also compared these to the values generated from permuting each of the contexts separately. Specifically, we carried out edge permutations. The first edge permutation simply allowed the randomisation of all in-going links, whilst maintaining the number of nominations (out-going links) each individual made within this information-sharing context (termed edge null model 1 - Figure 5.1a). The second edge permutation was a more conservative version of this, allowing swaps of links (which individuals nominated which other individuals in this information-sharing context) but maintaining the number of nominations each individual made in this information-sharing context (termed edge null model 2- Figure 5.1b). Separately, for each of the information-sharing contexts, 1000 permuted networks (of both of these permutation types) were generated and the distribution of the summary statistics were calculated for them.

5.4.4.6 Cross-contextual correlations

To reveal the extent to which the turtle bycatch information-sharing networks can be predicted from the other contexts, we examined the dyadic similarity between the different contexts of the information-sharing networks. We used context-based null models to compare the expected correlation between each context, and subsequently determined how the observed correlation between each context was driven by fine-scale structure over-and-above that expected from the general social structure of the system.

5.4.4.7 Assessing correlations

To examine the relationship between each network of dyadic information-sharing nominations, we calculated the correlation between the dyadic nominations on the unfolded network matrices. This approach is somewhat analogous to the Mantel test (that tests the correlation between two matrices), yet as the networks were directed (and non-symmetrical), this was applied to the entire matrix rather than the lower triangle part (but excluding the diagonals because ‘self-nominations’ were not possible). The calculated correlation statistic represented the similarity/dissimilarity in the directed dyadic nominations amongst contexts (who nominates whom), and these were compared to the distribution of the correlation statistic generated from the null models.

5.4.4.8 Null models for assessing cross-contextual correlations

The basic properties of each context, and the nomination structure in general, will have a larger deterministic influence on the cross-contextual correlations. For instance, considering a network of ‘any nomination in any context’, we would expect each individual context to hold a correlation equal to that of the number of nominations in each context (Figure A3.1.7). Similarly, contexts with similar numbers of nominations are more likely to be more correlated with one another than those with very different numbers of nominations. Simply carrying out edge-permutations, even conservative ones controlling for the number of nominations, or degree distributions, for example, would, by definition, randomise the underlying dyadic structure (who can nominate who) and thus mean all observed cross-contextual correlations would differ largely from expected under this null model just due to this alone. To infer the extent to which contexts are more, or less, similar than expected under the general dyadic

social structure, we carried out a cross contextual null model: For each dyadic nomination across any of the contexts, we randomised the contexts that these nominations were made within. For instance, when individual X nominated individual Y for information sharing within 3 different contexts, we allowed these 3 nominations to be reassigned to any of the contexts, but all 3 still in the direction of individual X nominating individual Y within these contexts. In this way, the overall dyadic nomination structure was maintained but the contexts within which these dyadic nominations took place within were randomised. Using this method (termed ‘context null model 1’ – Figure 5.1c), 1000 permuted networks were generated and the distribution of the expected cross-contextual correlations was recalculated using this.

As an even more conservative version of a cross-contextual null model, we created a new version of these permutations but also controlling for the number of nominations that took place overall within each context. For instance, when individual X nominated individual Y for information sharing within three different contexts, these three nominations were reassigned amongst the contexts in a way that was equal to the number of nominations in each context. For example, if context A had twice as many nominations in total as context B, reassigning a nomination between individual X and individual Y would be twice as likely to be reassigned within the context A than the context B. This was done by simply swapping individual context nominations between dyadic nomination pairs. This is similar to a group-by-individual permutations (Bejder et al. 1998) but where the rows of the matrix were set as the individual-to-individual dyadic nominations, and the columns were set as each of the information-sharing contexts. Using this permutation procedure (termed context null model 2 – Figure 5.1d), we generated 1000 permuted networks (with 100 swaps between each network and a burn-in of 2000 swaps; Figure A3.1.8) and then calculated the distribution of the expected cross-contextual correlations under this null expectation.

Chapter 6

Linking locally tailored solutions with broad concepts for biodiversity conservation

6.1 Research summary

The aims of the research presented in this thesis were twofold. The first aim was to develop and improve the theoretical basis that reconciles commercial production of renewable natural resources with biodiversity conservation through case studies focused on commercial fisheries and more widely across the renewable resource sectors. The second aim was to help develop broader systems thinking in fisheries management by emphasizing anticipation and prevention of adverse human impacts across the life cycles of incidentally captured marine megafauna species. The aims set were suitably broad for an investigation into a system-scale framework that seeks to track progress towards an agreed overarching objective, based on net conservation outcomes.

To work towards these aims, I set four research objectives. To explore the use of an approach for the mitigation of impacts on marine biodiversity (widely used in terrestrial EIAs) as an overarching framework for integrating the multiple elements of conservation goals and interventions towards a common goal. To explore specific challenges and trade-offs when attempting to implement a holistic management strategy for sea turtles in a data-poor, small-scale fishing system; particular attention was paid to obtaining the necessary data to understand the bycatch issue and implement appropriate management measures. To consider novel approaches to collecting and analysing information in data-poor fisheries scenarios to support a holistic recovery framework for sea turtles. To investigate innovative approaches to understanding social dimensions of conservation problems by mapping social network structure to support conservation intervention expansion and uptake.

The need to address the pervasive impact of human development on biodiversity in all sectors is clear (FAO 2018; Kok et al. 2018; IPBES [Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services] 2019). Taking a strategic and systematic

approach to measuring negative human impacts on the natural world and the efforts undertaken to manage those impacts offers the potential to better guide actions towards achieving a desired state of biodiversity. In Chapter 2, titled ‘A global mitigation hierarchy for nature conservation’, I explored the conceptual application of the biodiversity mitigation process across all human impacts to biodiversity. Yet arguably the primary focus was on the primary resource sectors – where impacts to living nature are the greatest, and the biodiversity mitigation approach is not yet applied. In taking a strategic approach to consider both positive and negative impacts on biodiversity across sectors and between scales, this research supports furthering systems thinking as a means to capture unaccounted biodiversity impacts and to better align humanity’s development initiatives with global biodiversity goals (Meadows 2008; Mahajan et al. 2019). The broad framing also highlights current failings that arise from implementing numerous project-scale biodiversity mitigation initiatives, while overlooking their cumulative impacts at the landscape or national scales (Maron et al. 2018a; Simmonds et al. 2019).

In fisheries, marine megafauna species such as turtles are particularly vulnerable from negative fishing impacts when the loss of sexually reproductive individuals occurs in populations (Finkelstein et al. 2008; Casale & Heppell 2016). Fishing-related mortality poses one of the major threats to all seven extant sea turtle species (Wallace et al. 2011). Nevertheless, many coastal fisheries around the world suffer from poor management, monitoring, and regulatory systems (Berkes 2003; Karr et al. 2017). These fisheries are particularly relevant to focus management efforts on as they can act as sinks for marine megafauna populations (Alfaro-Shigueto et al. 2018). In Chapter 3, titled ‘A mitigation hierarchy approach for managing sea turtle captures in small-scale fisheries’, I used the conceptual mitigation hierarchy as a framework to assess adverse impacts from the San Jose gillnet fishery on sea turtle populations, and to then evaluate management options to mitigate that impact. While this case study research is at the scale of a single fishery (two distinct gillnet fleets), I explored challenges in linking tailored fishery-specific solutions to desirable biodiversity outcomes at the Regional Management Unit (RMU) scale (Wallace et al. 2010a). This exploration was undertaken by considering management options that mitigate negative anthropogenic impacts to turtles beyond the geographic extent of the fishery and thereby contribute to a broader population recovery strategy (Dutton & Squires 2008). This research expanded on the mitigation hierarchy framework theory developed for fisheries management (Milner-Gulland et al. 2018) by integrating it with established decision-making processes in

fisheries management to support the assessment of risks (consequence $\times$ likelihood qualitative ecological risk assessment; Fletcher et al. 2003; Fletcher 2014) and evaluation of trade-offs between sustainability goals (management strategy evaluation; Smith 1993, 1994). The proposed framework proved useful for aligning thought, process, and tools, and for understanding the conservation issue better to inform management decision making.

In many small-scale fisheries around the world, the mortality rates of species of conservation concern remains a knowledge gap (Peckham et al. 2007; Alfaro-Shigueto et al. 2018; FAO 2018). When budgets are low and rapid assessments of bycatch are necessary, researchers and practitioners can use expert knowledge to inform decision making (e.g., de Oliveira Braga & Schiavetti 2013; Alfaro-Shigueto et al. 2018). To help improve the use of expert knowledge in conservation practice, in Chapter 4, titled ‘Evaluating elicited judgements of turtle captures for data-poor fisheries management’, I explored the use of the IDEA (“Investigate,” “Discuss,” “Estimate” and “Aggregate”) structured elicitation protocol for gathering data on bycatch rates of green sea turtles (Hanea et al. 2016b). I chose to use the IDEA structured elicitation protocol because it incorporates a four-step elicitation process to reduce overconfidence (Speirs-Bridge et al. 2010), encourages consultation with a diverse group of experts (Burgman et al. 2011), affords experts the opportunity to examine one another’s estimates and to reconcile the meanings of questions through discussion (Burgman 2015), and uses performance-based mathematical aggregation of judgements (Hanea et al. 2016a). These characteristics allowed for flexibility when applying the approach to local resource users in San Jose. I then drew comparisons between elicited estimates and fisheries observer data obtained from the same fishing system. This research provides a valuable contribution by demonstrating the use of structured elicitation methods when involving experts who do not have experience in providing quantitative estimates (in this study, the gillnet skippers) as well as illustrating the validation of expert judgment when the ‘truth’ from observational data is also uncertain. I also designed this research to support an assessment of an empirical quantitative assessment of net outcomes in fisheries in future.

Understanding the social dimensions of conservation problems is essential for tackling the current widespread loss of biodiversity that is occurring (Mascia et al. 2003; Milner-Gulland 2012; Veríssimo 2013). Chapter 5, titled ‘Understanding the potential for information spread about bycatch reduction initiatives in small-scale fisheries’, uses graph permutation techniques (null models) to analyse social network data and maps how the pathways of information about a conservation issue (turtle bycatch) are shared across networks of fishers.

The analysis then compared the mapped turtle bycatch information-sharing network to other closely related information-sharing networks that relate to fishing (Croft et al. 2011; Farine 2017). This research investigated the assumption that knowledge of the structure of the network (i.e., which gillnet skippers interact with one another, and who shares information with whom) is consistent across different, but closely related contexts (Barnes et al. 2016). The results show that not all closely related networks are similar in structure. More broadly, the research presents an analytical method to consider these differences, new to the fields of conservation science and natural resource management. Implementing this approach could improve understanding of social network structure, for designing and implementing behavioural interventions.

Of the multiple elements addressed throughout this thesis, I will pick out two themes of broader relevance to discuss in more depth in this final discussion. The first theme is calculating net outcomes in fisheries. To address this, I focus on the data collected and collated from my case study fishery, San Jose (Chapters 3, 4, & 5), and highlight how to develop further understanding across ecological, economic, and social dimensions. The second theme is that of trade-offs with consideration of biodiversity loss and social equity. Throughout the discussion, I will draw on the two major themes and discuss opportunities for future research.

6.2 Calculating net outcomes in fisheries

A natural next step beyond the research presented in this thesis is to integrate the baseline data gathered in the San Jose gillnet fishery (Chapters 3, 4, & 5) and develop a semi-quantitative assessment of the mitigation hierarchy equation we presented in Milner-Gulland et al. (2018), as an interim step towards a fully quantitative assessment of achievement of net outcomes in fishery systems. Initially, I intended to integrate the detailed vessel-level economic data which I had collected alongside the social network data with the baseline biological data (Chapter 4) and social data (Chapter 5) in a final data chapter. This would explore how changing the type of management strategies implemented, and the extent to which each management strategy is implemented impacts different targets. These targets could be focused towards ecological (e.g., sea turtle capture/bycatch rates), economic (e.g., based on conservation budgets of the local not-for-profit *ProDelphinus*), and social factors (e.g., ensuring an equitable distribution of the costs of conservation interventions across community members to ensure none are

disproportionally affected; either positively or negatively). However, this proved too ambitious for the timeframe of a DPhil.

Additional real-world empirical applications of the biodiversity mitigation process explored throughout the research presented in this thesis are needed. This empirical application of the framework is necessary across different impact types, scales, and sectors. Only then can understanding as to the efficacy and applicability of the proposed framework in practice be understood well enough to encourage widespread application.

Currently, research is underway into the application of the mitigation hierarchy framework in two other small-scale fishing systems to explore the mitigation of sharks and their relatives (Class Chondrichthyes; Booth et al. 2019; Booth et al. 2020; T. Gupta, pers. comm.). The published research from these studies, to date, offers an exciting insight into the flexibility of the framework across marine megafauna taxa and the potential applicability of the biodiversity mitigation process to fisheries, yet remains mostly conceptual in terms of empirical application.

The research presented in this thesis was the first case study application of the mitigation hierarchy framework for fisheries management, as proposed in Milner-Gulland et al. (2018). The other two case study applications, like the current study, are also focused on small-scale, coastal fishing systems (Booth et al. 2019; Booth et al. 2020; T. Gupta, pers. comm.). Small-scale fisheries pose a challenging issue to conservation, and any advances in furthering understanding of fishing-related impacts on marine megafauna and the means to better manage those impacts in these systems are a welcome addition. Nevertheless, to undertake an empirical and fully quantitative assessment of a net outcome goal in fisheries, a better understanding of the relationship between population growth rates of the bycatch species of management focus and those same species bycatch rates is needed.

6.2.1 Ecological net outcomes

Should all marine megafauna losses and gains be summed across a fishery or fishing system in practice, then complete at sea observer coverage is necessary for all vessels in a fishing fleet of management focus. Comprehensive coverage is not a precondition for implementing potential management actions that fall within the steps the framework (including offsets; Squires et al. 2018); yet from a practical standpoint, to sum all losses and gains the full documentation of capture at sea is necessary. Until recently, full observer coverage across

fisheries was largely cost-prohibitive, but the continuing development of remote electronic monitoring is making it increasingly feasible to observe all fishing events and bycatch incidents at the individual vessel level (Kindt-Larsen et al. 2011; Mangi et al. 2015; Bartholomew et al. 2018). For example, the remote electronic monitoring system integrated in the analysis presented in Chapter 3 has been designed and tested in the San Jose gillnet fishery. The small-scale fishing vessel remote electronic monitoring system offers potential for affordable at sea monitoring costs in coastal fisheries (Bartholomew et al. 2018).

With spatially explicit data of where individual vessels go and what they catch, even if this is a sub-section of the fishing fleet, extrapolations of fishing-related mortality can be made. In Chapter 4, I calculated turtle capture rates from extrapolating observer data and tested a structured elicitation protocol as an alternative, inexpensive methodology for gaining insight into marine megafauna bycatch rates. The elicited estimates yielded a broader range, yet when averaged no significant difference between the two data sources was found. The protocol appears useful for exploratory assessments of fishing-related mortalities in small-scale fishing systems. An elicitation exercise could feasibly be undertaken for all vessels within a coastal gillnet fishery, offering rapid and cheap preliminary insights into capture/bycatch rates at the vessels level. Once an understanding of vessel-level marine megafauna capture and mortality rates are understood, even if this is at a coarse scale of resolution, then an evaluation of the conservation benefit (i.e., capture/bycatch reduction potential), cost, and social impact can be calculated. For example, when an understanding of basic revenue is combined with where individual vessel's travel and where they capture turtles (even if only a broad spatial area is known), then the opportunity costs of a spatial area closure can be calculated (Ban et al. 2009; Adams et al. 2011).

To achieve NNL or better an offset is necessary. In terms of potential biodiversity offset options for sea turtles in the context of the San Jose case study system, I gave consideration to funding turtle nesting site protection on beaches as biodiversity gains have been demonstrated (e.g., Chaloupka 2003; Balazs & Chaloupka 2004; Dutton et al. 2005; Troëng & Rankin 2005). In all fisheries, but particularly in small-scale commercial and subsistence fishing systems, researchers and managers should consider the social and economic context of fishers operating in the fishery in question (Hall et al. 2007). Any form of tax or fee directly imposed on the fishers in many coastal fishing communities would likely have a deleterious effect on human wellbeing and livelihoods. A more feasible scenario is a bycatch tax on a large-scale, commercial fishery that takes turtle bycatch from the same East

Pacific RMU population as the San Jose gillnet fishery (Squires et al. 1995; Donoso & Dutton 2010; Squires & Garcia 2018b). The Chilean pelagic longline fishery was presented as a possible candidate in Chapter 3, because Donoso & Dutton (2010) it is one of the few studies highlighting sea turtle bycatch in commercial fisheries in the eastern Pacific commercial fisheries as well as taking both East Pacific leatherback and green turtles as bycatch. Funds from the sea turtle tax could go to nesting site protection or bycatch reduction initiatives in the San Jose gillnet fishery. For example, funding could be given to purchase LEDs to go on all gillnets or to fund the proposed gear transition from gillnets to handline trolling (Chapter 3).

The terrestrial protection of juveniles presents several challenges when attempting to offset for adult mortalities. The life-history traits of longevity, iteroparity, late sexual maturity, and low fecundity, mean that sea turtles, seabirds, cetaceans, and large shark species, are particularly at-risk from losing an adult from their population following negative anthropogenic impacts (e.g., capture and mortality in fisheries). In other words, marine megafauna life-history traits sharply limit their maximum attainable rates of population growth (Finkelstein et al. 2008). To illustrate some of the considerations and data necessary, I will draw on the most well-studied sea turtle species, the loggerhead sea turtle (*Caretta caretta*), even though this species has not been a focus of the current research. A stage-class population model calculated for loggerheads found that the mortality of one adult loggerhead was equivalent to 588 hatchlings (Crouse et al. 1987). Loggerhead turtles lay on average 110 eggs per clutch and typically each egg has an 80% or higher chance of hatching success (i.e., the number of hatchlings leaving their eggs; Miller et al. 2003). This rate drops slightly in terms of emergence success (i.e., the number of hatchlings reaching the beach surface; Miller 1985). During the day, the major lethal threats that emerged hatchlings face when traversing the beach to the water include high temperatures, predators (e.g., dogs and human hunters), and pollution (e.g., artificial lighting, debris, and petroleum contamination). During the night, the major lethal threats are predators and pollution (Bolten et al. 2011; Wallace et al. 2011). When quantifying the protection needed to equate to a biodiversity offset, one must consider the threats eggs in nests face prior to turtles hatching. Consideration of threats to unhatched turtles is needed because an understanding of how many unprotected nests are present at a given nesting site, as well as the nest to emerged hatchling ratio are necessary (Crouse et al. 1987; Dutton et al. 2005). Loggerheads, and all sea turtle species more generally, can have their eggs harvested, and nesting sites destroyed from shoreline stabilisation projects and

coastal construction (Wallace et al. 2011). All nesting sea turtles also face an increasing risk of climate change threats such as rising temperatures and sea levels that can affect hatchling sex ratios and inundate nests (Witt et al. 2010; Tomillo et al. 2015; Varela et al. 2019). Thus, considerable population data must feed the offset calculation.

If nesting site protection is a chosen option, funds could either be undertaken at unprotected southern nesting sites closer to the San Jose fishing system, as social acceptance of offsets is higher when the offsets take place closer to the impact site (Rogers et al. 2014; Rogers & Burton 2017). Olive ridley (*Lepidochelys olivacea*) have the closest nesting sites to San Jose with one nest report in El Ñuro, Piura, Peru (Kelez et al. 2009), approximately 375km from San Jose. Green turtles (*Chelonia mydas*) have been reported nesting in Los Pinos, Tumbes, northern Peru (Forsberg et al. 2012), approximately 466km from San Jose. No established nesting sites for leatherback turtles (*Dermochelys coriacea*) are present in Peru. With the closest leatherback nesting area to San Jose located in Ecuador (Eckert 2012). However, these sites may only be sporadically nested rather than established secondary nesting sites. Thus, further evaluation of any unprotected nesting sites close to San Jose is necessary. Alternatively, funds from a bycatch tax protection could focus on the most conservation ‘bang for the buck’ and direct conservation efforts to critical areas for the most threatened population captured in the fishery that are further away (e.g., a fund to purchase areas of critical nesting habitat for the East Pacific population of leatherback turtles in Costa Rica; Dutton & Squires 2008).

In Chapter 3, I considered other positive incentive-based mechanisms as possible management strategies in the form of an in-kind payment for ecosystem services (PES) scheme in which fishers donate their expertise and time as part of a conservation initiative and either receive a monetary or in-kind payment in exchange (Bladon et al. 2014). For example, in the Watamu Marine Park and Reserve complex in Malindi, Kenya, a not-for-profit called the Local Ocean Trust runs a conservation program that, in addition to conducting research, implements a community conservation education program and pays villagers performance payments for nest protection (Flintan 2002). The Local Ocean Trust has a tiered PES system in place. Individuals who report a nest receive a financial payment upon verification by the Local Ocean Trust, followed by a second financial payment upon verification of the nest hatching successfully (Flintan 2002). In San Jose, equivalent efforts could focus on a local conservation education program that shifts social norms away from the currently high rates of

turtle consumption (Alfaro-Shigueto et al. 2007; Alfaro-Shigueto et al. 2018). Such a scheme, funded by the biodiversity offset, could pay fishers a small fee when they call in leatherback turtle sightings at sea. Integration of a second payment made following the successful tagging of any reported leatherback turtles would provide an additional positive incentive to fishers and support local understanding of sea turtle movements and foster interest in their behavioural ecology.

6.2.2 Developing cost-effective conservation

Because of limited conservation budgets (James et al. 2001; Stroud et al. 2014), the consideration of project cost and the feasibility of implementing potential strategies have been extensively explored in the systematic conservation planning literature (Underhill 1994; Balmford et al. 2000; Sarkar et al. 2006; Wilson et al. 2006; McCarthy & Possingham 2007; Joseph et al. 2009). Many fisheries are also underfunded, and consideration to how managers can shift from one conservation management action to another at the opportune moment to achieve bycatch reduction goals cost-effectively is becoming increasingly common (Pascoe et al. 2011; Gjertsen et al. 2014; Squires et al. 2018; Tulloch et al. 2019). Nevertheless, to date, a comprehensive cost-effectiveness assessment that considers both direct and opportunity costs across the mitigation hierarchy has not been analysed in practice.

Theoretically, the mitigation hierarchy is considered in a step-wise, sequential manner during the ex-ante screening, scoping, and risk-cost analysis stages of the EIA process. The boundaries between each step of the mitigation hierarchy are based on a precautionary risk-based approach for biodiversity to maximise, to the greatest extent practicable, actions within each step while accounting for the risk from the remaining predicted impact before progressing to the next stage (CEQ [Council on Environmental Quality] 2000; BBOP [Business and Biodiversity Offset Programme] 2012). In this way, the mitigation hierarchy is not proposed as a one-way linear process and entails both feedback and adaptive management to optimise investments at each step of the process (CEQ [Council on Environmental Quality] 2000). In practice, actions spanning more than one step of the mitigation hierarchy are often implemented simultaneously and the EIA process in practice can differ substantially from the EIA process in theory (Sinclair 2018). In fisheries, implementation of a suite of interacting measures from several levels of the hierarchy may occur, which changes over time (Milner-

Gulland et al. 2018). This system dynamism can complicate calculations of effort, bycatch-relevant effort, and bycatch per unit effort (Bishop 2006).

In cases with lower risk, for example, where fishing impacts are occurring only to biodiversity that is of a lower conservation priority, such as a highly abundant non-target fish species, there is more flexibility in progressing through the mitigation hierarchy towards offsetting. Where fishing impacts are occurring to biodiversity that is a high risk, such as sea turtles, there should be greater rigour before proceeding to the next step (and in some instances, progression should not proceed, i.e., a “no exploitation” scenario) as the risk of failing to achieve the biodiversity goal is high. In fisheries, this could involve shutting a fishery down due to an extreme extinction risk to a species (e.g., high captures of eastern Pacific leatherback turtles; Spotila et al. 2000; Mazaris et al. 2017).

As well as gathering baseline capture data (Chapter 4) and gillnet skipper network data (Chapter 5), fine-scale economic data for individual operating costs for gillnet vessels in the San Jose gillnet fishery were collected. This data intended to integrate a cost-effectiveness assessment to evaluate specified capture and bycatch reduction targets, using the mitigation hierarchy framework to generate the most conservation benefit in a cost effective manner (Squires et al. 2018). A cost-effectiveness approach explores shifting from one step of the mitigation hierarchy to the next based on marginal costs across the entire hierarchy for a priced bycatch component, rather than undertaking conservation action to the greatest extent possible without regard for bycatch cost.

The mitigation hierarchy is not prescriptive in the degree of implementation of management strategies across each of the four action steps to achieve the biodiversity mitigation target specified. This flexibility comes with a need to highlight uncertainty and manage risk. Thus, any consideration of lowering costs for more conservation benefit must also consider lowering risk as well. To illustrate, in fisheries, several studies comparing bycatch management actions spanning more than one step of the mitigation hierarchy indicate actions lower down the mitigation hierarchy (i.e., post-harm compensatory actions) can achieve the same reduction in bycatch as avoidance-based measures, but for less cost. For example, nesting beach protection (offset) was identified as a more cost-effective means of achieving increases in leatherback populations than at sea conservation strategies (avoidance) in the US Hawaiian longline swordfish and California drift gill net fisheries (Gjertsen et al. 2014). Across trawl, net, and line fisheries of Australia, the most overall cost-effective

measures to reduce cetacean bycatch were trawl-net modifications (minimisation; Tulloch et al. 2019). However, most notably in the latter study, many locations in the national-scale analysis were identified where spatial closures (avoidance) were the most cost-effective conservation solution, despite higher management costs. This was because of their ability to lower risk as a result of effectiveness in limiting fishing gear interactions with cetaceans (Tulloch et al. 2019). Thus, these data do not undermine the need for a mitigation hierarchy as a way of organising thinking about bycatch reduction. Management strategy success can depend strongly upon both the degree of compliance with net outcome policies and upon underlying ecological parameters within the system they are applied (Bull & Milner-Gulland 2019; Tulloch et al. 2019), and uncertainties may also play an important role in shifting preferences towards preventative rather than compensatory measures. These results do, however, highlight that further exploration into efficient means for shifting between steps of the mitigation hierarchy framework is warranted.

An essential element that was not explored in detail in the main data chapters presented in this thesis is how to calculate and implement a bycatch tax to finance a biodiversity offset (i.e., for nesting site protection or bycatch mitigation in small-scale and subsistence fisheries that have high catch rates of marine megafauna). A bycatch tax corresponds to an environmental Pigouvian tax on an external cost (here the capture of marine megafauna e.g., sea turtles) to the target catch (e.g., specified species of finfish; Squires & Garcia 2018b). When an environmental Pigouvian tax is imposed or voluntarily implemented on a group of fishers, these fishers face an incentive to reduce bycatch to a specified level. In principle, the key is ensuring that the level that the tax is set equals the external cost, i.e. the cost (however valued and measured) of sea turtle bycatch or mortality (Segerson 2011; Squires & Garcia 2018). The bycatch tax itself, through the incentive to reduce bycatch, creates the first dividend and the conservation gains from the use of the collected tax revenues creates the second dividend, creating a “double dividend” Pigouvian tax.

A key question of implementation is the level of production to set the regulation: vessel, set of the gear, catch (landings), turtle interactions, or turtle mortality (Segerson 2011). Each creates a different set of incentives, where moving from vessels to sea turtle mortality improves the incentives but increases the uncertainty, since vessel and number and size of gear are known, number or length of sets of the gear, target catch/landings, and the number of sea turtle interactions are sometimes known through observer data, and sea turtle mortality unknown. A tax set on the vessel or gear itself rather than on their use is a lump-sum tax, and

as such does not create incentives to change fisher behaviour and decision-making at the margin (Squires & Garcia 2018b). A potential trade-off is created between incentives and uncertainty. Costs of implementation also arise the closer the tax is set to sea turtle mortality. Since sea turtle interactions can be rare events, especially when their populations are low, the uncertainty is compounded.

Approaches to addressing the uncertainty can vary (Segerson 2011). The current policy simply does not account for this uncertainty. Probabilistic limits are possible. Banking or borrowing across vessels and/or time can smooth the impacts and reduce uncertainty, although the asymmetry of impacts could be an issue. The Hawaiian large-scale pelagic longline fleet is regulated in this manner (Clarke et al. 2015). Since the number of allowed sea turtle interactions is fewer than the number of vessels in the fleet, a group limit is used. An overall bycatch limit, however, can lead to “race-to-fish” incentives without group management (Abbott and Wilen, 2009, Gjertsen et al., 2010; Segerson, 2011).

Issues of implementation arise through compliance, monitoring, and enforcement (Segerson 2011). The size of the tax and whether to set the tax as a lump-sum tax or as a tax on marginal behaviour, such as the number of interactions, target catch, or impact tax implementation. The first implementation of such a double dividend Pigouvian tax, by the California drift gillnet fleet, was a lump sum, voluntary tax (Janisse et al. 2010). The second implementation, through the International Seafood Sustainability Foundation, was a marginal tax set on the size of the target catch (in this case swordfish; Squires et al. 2018). In both cases, the size of the tax was essentially arbitrary, determined by what was feasible. Which type of tax to set, whether lump sum or marginal, may be settled by expediency of implementation, which was the case for these two taxes.

6.2.3 Integrating social perspectives into fisheries management

In fisheries management, along with assessing the maximum conservation expenditure given a given budget constraint, further exploration into the equitable distribution of the impact of conservation interventions and the associated socioeconomic effects on wellbeing across stakeholders are needed (Loomis & Ditton 1993; Lam & Pitcher 2012; Voss et al. 2014; Booth et al. 2019). Economic measures such as the Gini coefficient – a metric to indicate dispersion within a frequency distribution (Gini & Mutabilita 1912) - offer the possibility to compare ranked indices of fishers according to ecological (e.g., turtle bycatch rate), economic

(e.g., cheapest management strategy or group of management strategies to implement), and social (e.g., social influence measured by network centrality; Chapter 5) rankings. Metrics such as the Gini coefficient would help in determining the most equitable and effective means of allocating conservation effort to change behaviour across individuals in a fishery.

To illustrate this, consider the implementation of a hypothetical no-take marine protected area (MPA) across 25% of the geographic extent of the San Jose gillnet fishery system in the northern reaches (Figure 3.1). How might this avoidance measure affect individual fishers within the San Jose community? It is entirely feasible that the inshore gillnet (IG) fishers with low incomes operating in the northern area of the IG fleet's geographic extent may be disproportionately affected, either economically, socially, or both, by the proposed area closure, compared to fishers operating in the southern reaches of their fleet's geographic extent (Cinner et al. 2010; Cinner et al. 2012b; Halpern et al. 2013; Cinner et al. 2019). Thus, consideration must be given not only to the number of community members that management strategies must be applied to, to achieve the desired biodiversity mitigation goal, and at what cost. But also, to which community members are most economically or socially vulnerable, to ensure a more equitable and just distribution of conservation actions and their resulting beneficial outcomes to human wellbeing (Milner-Gulland et al. 2014; Woodhouse et al. 2015; Griffiths et al. 2018).

The social network analysis I implemented offers insight into the structure of multiple information-sharing networks about fishing across the San Jose gillnet skipper network and how they relate to one another. Future research could focus on testing the assumption that sub-optimal interventions would eventuate by targeting influential individuals outside of the information-sharing network directly related to the conservation intervention in question (Barnes et al. 2016; Chapter 5). Testing could simulate the transmission of information around the turtle bycatch network under various scenarios (Wu et al. 2004; Iyengar et al. 2011). For example, central individuals (based on the fishing site information network) could be the targets of bycatch information on the assumption that they can spread it to many people. Social network analysis can therefore support an understanding of how to allocate limited conservation resources across the mitigation hierarchy framework. Following such application, empirically testing predictions using randomised control trials between fishing communities in the field could allow for causal investigations into the efficacy of conservation interventions (Ferraro 2009; Banerjee et al. 2014, 2019).

6.3 Biodiversity loss and social equity

In Chapter 2, I highlighted that an important consideration for a global biodiversity conservation framework is the equitable distribution of costs and benefits between nations. I proposed that meeting any global biodiversity conservation goal through a global mitigation hierarchy framework could follow the United Nations Framework Convention on Climate Change's (UNFCCC) management of carbon emissions, with nations setting their own national goals and targets that then sum to achieve overarching planetary goals.

Maron et al. (2019) explore the equitable translation of country-level contributions towards a global net outcome target. The study evaluates the proportion of natural ecosystem between countries (using the human footprint dataset; Venter et al. 2016) and the variation in depletion of natural ecosystems (using the aforementioned Gini coefficient; Gini & Mutabilita 1912). The authors identified a wide variation in ecosystem depletion across countries. These results highlight that a range of fixed baseline net outcomes would be necessary to sum national contributions to an overarching net outcome goal. For example, country commitments could vary from net gain (for countries that need landscape scale restoration e.g., United Kingdom, France, Italy), to managed net loss (in rare circumstances where natural ecosystems remain extensive and the human development imperative is greatest e.g., Gabon, Niger, Suriname; Maron et al. 2019; Simmonds et al. 2019). This research begins to address the vital question of how humanity can achieve equitable biodiversity conservation at scale. Within a systematic framework for conservation (Chapter 2), consideration of the equitable distribution of conservation effort and the critical issue of historical natural impacts (national conservation debt) should be actively addressed.

When applying net outcome concepts to the scale of sea turtle RMUs (Chapter 3), the same approach could theoretically apply. Heterogeneity between different ecological and socioeconomic scenarios can be considered and integrated to systematically achieve the desired biodiversity outcome (Milner-Gulland et al. 2018; Booth et al. 2020). For example, certain large-scale commercial fishing fleets could be required to contribute a net gain towards sea turtle bycatch by contributing additional offsetting actions. For example, the turtle bycatch tax implemented by fishers and managers of the US California Drift Gillnet Fishery to fund turtles nesting site protection in Mexico (Janisse et al. 2010), if expanded to a more ambitious NNL or NG target, offers a possible template. Another example includes the proposed bycatch (Pigouvian) tax applied to the Chilean pelagic longline fleet for any

captures of leatherback turtles (Chapter 3; Donoso & Dutton 2010). Subsistence or small-scale commercial fisheries, for which development will support improved food security and poverty alleviation, rare allowances for a ‘managed net loss’ of harmful impact on specific turtle populations could be considered – provided gains and losses across all fisheries combine to achieve net population stability or net population gain for the sea turtle population in question (Chapter 3; Booth et al. 2020; Simmonds et al. 2019). In theory, this would allow for marine megafauna population recovery overall, despite (only in rare cases) select fisheries transitioning across a declining period of annual turtle captures towards an equal contribution with more developed fishery management frameworks in future (e.g., five or ten years). The benefit of such an approach is that there is an allowance for beneficial socioeconomic developments to occur in less developed fisheries management systems.

One of the most important factors when considering both a system- and global-level mitigation hierarchy approach is the operationalisation costs of tracking all losses and gains across all fisheries frameworks, and all human impact more broadly. Undoubtedly there would be substantial costs in such an approach. These costs would be associated with data collection, maintenance of data platforms, design and implementation of monitoring protocols, and managing incentive mechanisms (Bull et al. 2019). However, one of the reasons that an expansion of the mitigation hierarchy approach to a broad range of anthropogenic impacts was considered was that much of the institutional and legislative machinery for net conservation approaches are already in place. Currently, 133 parties to the Convention on Biological Diversity (CBD) either have the regulatory requirements for biodiversity impact mitigation measures with a net outcome objective, or are developing related policies (IUCN 2016b; Bull & Strange 2018; IUCN 2019). In fisheries, there is no formal specification of net outcome policies in the United Nations Convention on the Law of the Sea (UNCLOS; United Nations 1982), yet at least 77 countries now have had compensatory policies that enable the use of offsets in the marine environment (albeit mostly for coastal development rather than fisheries; Shumway et al. 2018).

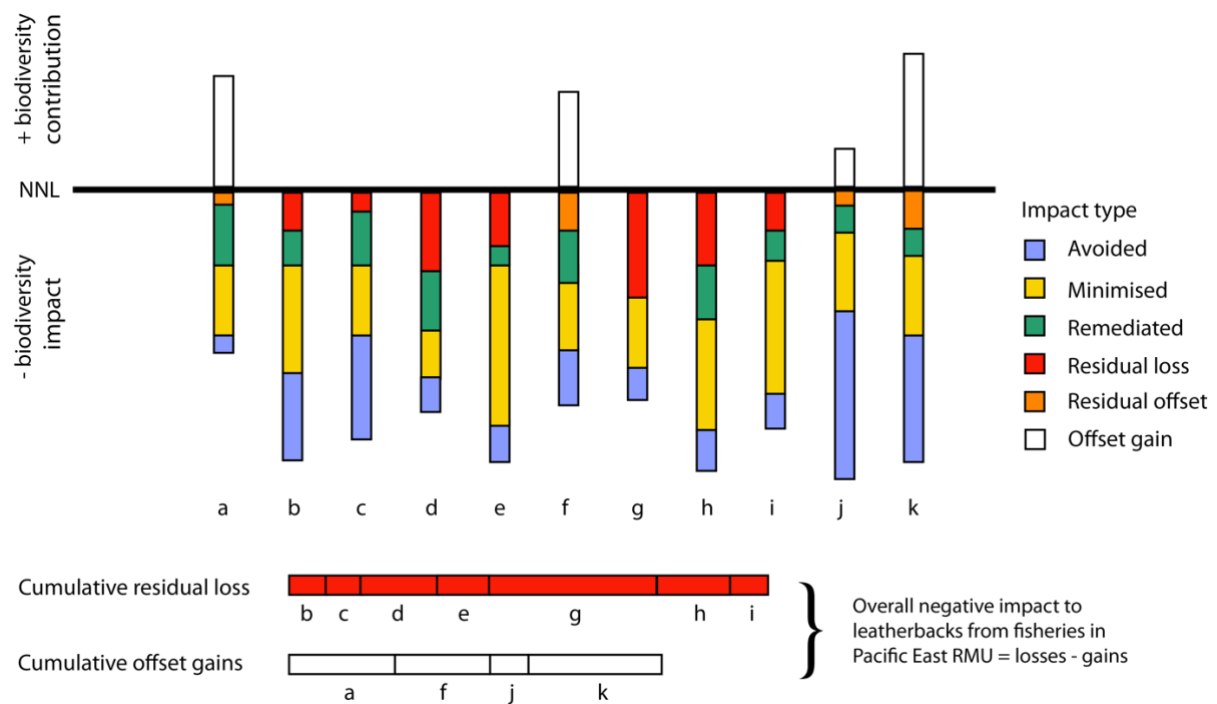


Figure 6.1. A hypothetical multi-fishery-level mitigation evaluation of the Pacific East RMU leatherback sea turtle population. Impacts and mitigation for the Pacific East RMU turtle population are represented here for 11 fisheries that sit within the boundaries of the leatherback turtle's Pacific East RMU (Wallace et al. 2010a). An RMU-scale population recovery plan could recommend different mitigation options depending on the local conditions. To illustrate the multilateral extent of a holistic recovery plan for the leatherback Pacific East RMU population, I present mitigation measures for one fishery operating in the exclusive economic zone of each country throughout the leatherback turtles East Pacific RMU distribution. In reality, multiple fisheries are operating within each country's waters, thus finer scale considerations of inter-country fishery interactions are also necessary. There is an assumption that all states set a fixed net positive reference scenario for population recovery. Countries a, f, j, and k are doing net gain at the individual fishing level. Countries b, c, d, e, g, h, and i are operating at a net loss at the project level. Country a's fishery has the lowest impact to leatherback turtles, countries j and k's fisheries, the highest. As a result of their high impact, countries j and k's fisheries are focused on avoidance. In contrast, country a is more focused on at sea minimisation and remediation measures, and biodiversity offsetting with some remediation measures also in place. The country i follows a similar method to the country a but does not implement any residual offset measure. Critically, countries must consider where to attempt biodiversity offset in a scenario such as this. For example, can they offset outside their nation, and if so, the necessary measures to prevent any form of double counting would need to be in place. Inspired by Kiesecker et al. 2009.

In Chapter 3, I found that because no management benchmarks for turtle protection were in place for the San Jose gillnet fishery, there was a need for fishery-level quantitative targets, metrics, and baselines that would contribute to broader turtle population recovery goals (across Pacific East RMUs). The management benchmarks which I chose for the San Jose gillnet fishery (to reduce turtle captures from 2020 levels by 15% every year for five years while maintaining total catch weight) remain somewhat arbitrary. Initially, I intended to

calculate an East Pacific Potential Biological Removal (PBR) for each sea turtle species captured in the San Jose gillnet fishery, but this proved too ambitious for the current research. PBRs can set an ecological threshold for a population that can be integrated into management measures. Should the mitigation hierarchy bycatch equation (Milner-Gulland et al. 2018) be solved for the San Jose gillnet fishery, and a net change in population growth rate achieved, that equation must adequately sum with other mitigation hierarchy equations across other fisheries and activities negatively impacting the focal sea turtle population. Only then would the population recovery goal for a transboundary marine megafauna species, such as the sea turtle species in this research, be quantitatively achieved (Figure 6.1).

6.4 Conclusions

This thesis is concerned with developing and improving the theoretical basis that reconciles natural resource management with biodiversity conservation. I have extended previous research by exploring the potential application of a broad scale biodiversity mitigation framework at two different scales: i) to support fisheries managers in the recovery of depleted marine megafauna populations, and ii) in taking a more strategic approach to assessing humanity's impacts on biodiversity more broadly. This involves a shift in focus away from top-down global biodiversity targets towards finding a process-based framework to achieve desired outcomes for biodiversity (Maron et al. 2018b; Bull et al. 2019). I have contributed to improving data gathering in data-poor fishery systems through my exploration of the IDEA structured elicitation protocol for gaining rapid insight into marine megafauna bycatch rates. Cross-contextual social network analysis has been introduced to conservation science and natural resource management as a novel approach to integrating social science research practices into management decision making in fisheries. As humanity seeks to deliver nature conservation alongside the development of human societies, there is a need for effectively tailoring solutions to local conditions using real-world data while considering broad-scale perspectives towards evaluating and managing anthropogenic impacts. This research contributes to an important and timely dialogue that seeks to shift emphasis away from piecemeal actions that prevent biodiversity loss, and instead adopt a strategic and proactive approach to restoring nature.

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Appendix 1

A1.1. Focus group discussion

Focus group discussion (FGD) were held in the ‘Asociación de Pescadores Artesanales de la Tercera Edad’ building in San Jose, Lambayeque, Peru (6°46’ S, 79°58’ W) during field surveys from 1 July to 30 September 2017. FGD were facilitated by two researchers, one of whom was from Peru and whom was experienced in working with coastal fishers along the nation’s coastline. The FGD estimating the San Jose inshore gillnet fleet’s geographic extent comprised 14 males and 1 female. Respondent age ranged from 22-58 years. Fishing experience for skippers ranged from 5-46 years. The FGD estimating the San Jose inshore-midwater fleet’s geographic extent comprised 4 males and 1 female. Respondent age ranged from 27-50 years. Fishing experience for skippers ranged from 11-17 years. Respondents were provided with refreshments and food during the FGDs.

Table A.1.1.1. Respondent characteristics by focus group discussion (FGD).

Variable	Category	FGD 1	FGD 2
Occupation	Skipper	13	3
	NGO scientist	1	2
	IMARPE officer	1	0
Gender	Male	14	4
	Female	1	1
Age	18-25	1	0
	26-40	5	3
	41-55	8	2
	56-70	1	0
Years lived in San Jose	0-10	1	2
	11 to 20	0	0
	21-30	5	1
	31-40	2	2
	41-50	5	0
	51-60	2	0
Years fishing	0-10	4	2
	11 to 20	3	3
	21-30	3	0
	31-40	3	0
	41-50	2	0

A1.1.1 Focus Group Discussion of turtle bycatch reduction options

The following strategies could be considered in future to reduce the number of turtles captured in gillnets in San Jose. Following information presented on each please rank on a scale of 1-5 how you would feel about using this strategy in your day to day fishing practices. A rank of 1 would indicate that you do not agree with the proposed turtle bycatch reduction strategy at all, whereas a rank of 5 would indicate that you are in total agreement with the proposed bycatch reduction strategy.

Total gillnet ban. Gear switching to lobster potting or hand line fishing (trolling)

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Spatial gillnet ban - prohibited driftnet fishing distance extended around Lobo de Tierra and Lobo de Afuera to 15 nautical miles offshore the islands. All year.

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Temporal gillnet ban between August to November. Gear switching to lobster potting or hand line fishing during the gillnet ban period every year.

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Spatial and temporal gillnet ban – gillnet ban shifting in space and time in relation to turtle movement.

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Offshore distance restriction – gillnetting only allowed to occur between 0-2 n.m. offshore

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Restrictions on soak time of gillnets (6 hours during daylight hours)

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Buoyless (buoys removed from float line) nets

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Using fixed (set) gillnets over drift gillnets

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

LED lights on gillnets

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Annual workshops on safe handling and release procedures, which includes the resuscitation of sea turtles

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Annual fee (bycatch tax) to fund turtle nesting site protection e.g., unprotected smaller nesting sites in Peru, Ecuador, Costa Rica, or Mexico (depending on species).

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Fishermen community enforcement in the 5-nautical mile Marine Protected Area around Lobo de Tierra and Lobo de Afuera [In-kind payment for ecosystem services program]

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Requirement for electronic monitoring device on all chalana/lancha *[delete as appropriate]* boats launching from San Jose

Do not agree with strategy at all				Total agreement with strategy
1	2	3	4	5
0	0	0	0	0

Table A1.1.1.1. Fishing area elicitation, specifying min, max, and average distances offshore from San Jose, North of San Jose, and South of San Jose. Respondents in the Focus Group Discussions were also asked for their certainty that the truth falls within the bounds of their estimate. Answers were asked to be given to the closest 5 nautical mile estimate possible, based on the maps provided.

	Winter			Summer		
	Offshore	North	South	Offshore	North	South
Min						
Max						
Average						
% interval						

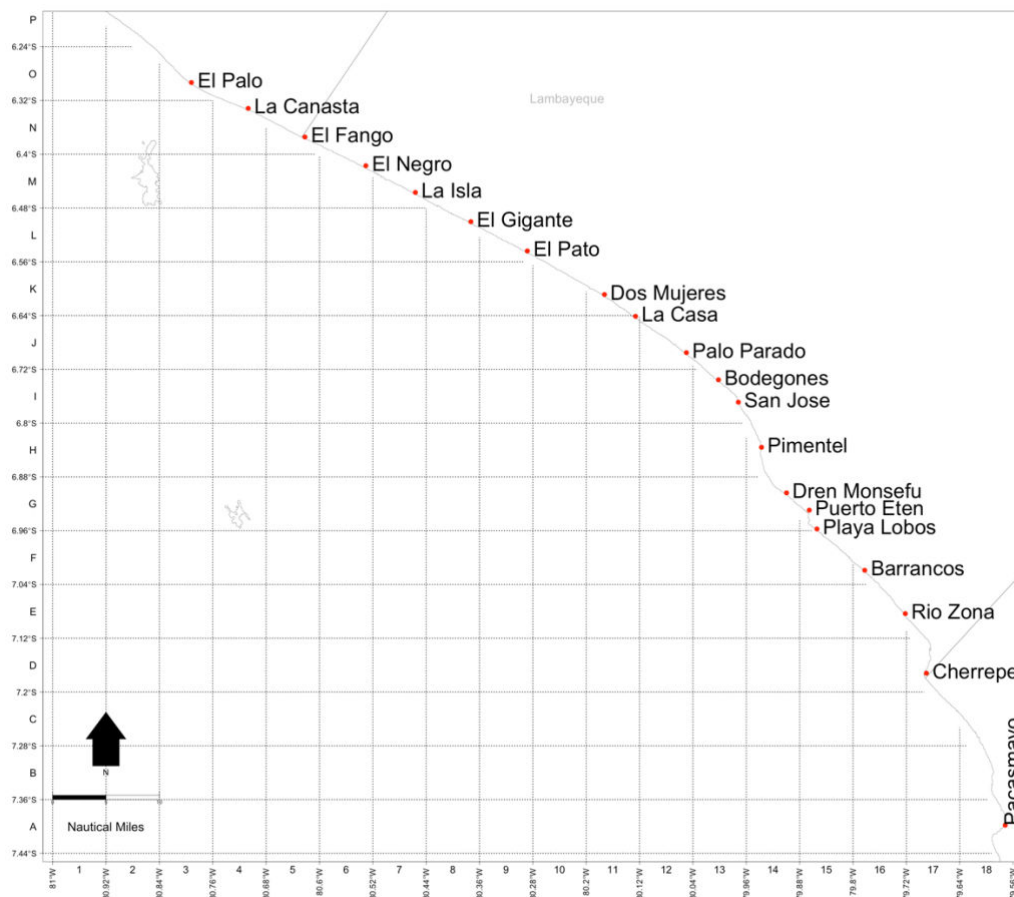


Figure A1.1.1.1. The San Jose inshore gillnet fleet map used to support elicitation of geographic extent of fishery.

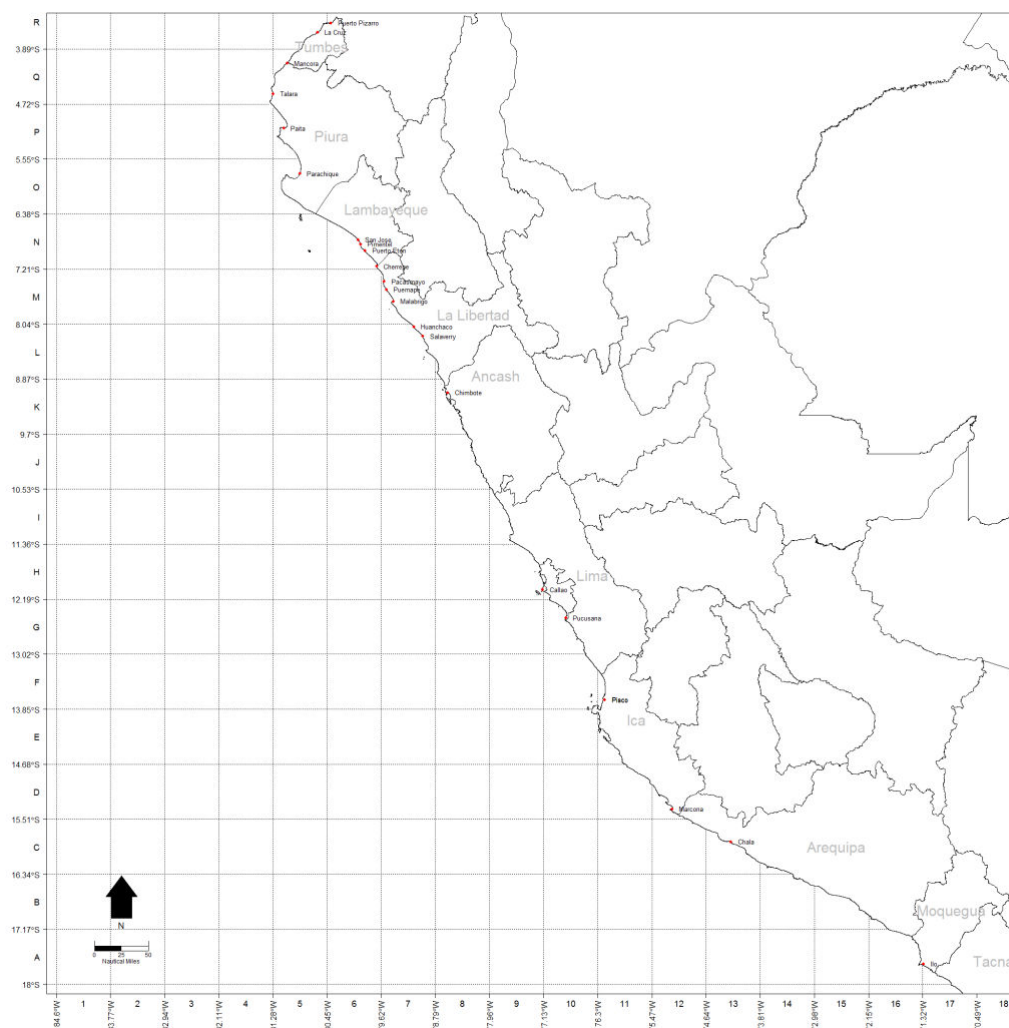


Figure A1.1.1.2. The San Jose inshore-midwater gillnet fleet map used to support elicitation of geographic extent of fishery.

A1.2. Additional risk assessment analysis

Table A1.2.1. Known seasonal changes in San Jose gillnet fishery characteristics by fleet. Summer fishing season = December – May, Winter fishing season = June – November. Fleet number is based on field census data of actively fishing gillnet skippers in San Jose that was obtained in the winter fishing season of the year 2017. IMG = inshore-midwater gillnet fleet, IG = inshore gillnet fleet.

Fleet	Season	Geographic extent	Fleet vessel number
IMG	Summer	27000 km ²	28
	Winter	31500 km ²	18
IG	Summer	1200 km ²	150+
	Winter	3700 km ²	150

Table A1.2.2. San Jose gillnet fishery characteristics by fleet. IMG = inshore-midwater gillnet fleet, IG = inshore gillnet fleet. Data are presented as mean $\pm$ standard deviation (SD). IMG sample size = 18, IG sample size = 150.

Fleet	Days per trip	GRT	Boat length (m)	Net length (m)	Outboard motor (horse power)	Management restrictions (effective overlap)
IMG	7.5 $\pm$ 1.6	8.9 $\pm$ 2.4	9.1 $\pm$ 1.8	1729.6 $\pm$ 611.7	-	Some at sea minimisation (LEDs) and remediation (REM and post-capture handling workshops)
IG	2.4 $\pm$ 1.4	3.7 $\pm$ 1.1	6.3 $\pm$ 1.3	1027.3 $\pm$ 327.1	52.4 $\pm$ 11	No

Table A1.2.3. San Jose gillnet fishery bycatch data by fleet. IMG = inshore-midwater gillnet fleet, IG = inshore gillnet fleet. BPUE data are presented as mean $\pm$ standard deviation (SD). Observer data are sourced from a volunteer programme over August 2007–May 2019 observing ~1-4% of the IMG fleet.

Fleet	Turtle species	Estimated potential overlap (fishery/turtle)	Turtle bycatch evidence	BPUE per trip	Observed released without injury	Observed injured releases	Observed deaths
IMG	Green	100%	Alfaro-Shigueto et al., 2010; Alfaro-Shigueto et al., 2018	0.71 ± 1.98	199	100	23
	Olive ridley	75%	Alfaro-Shigueto et al., 2010; Alfaro-Shigueto et al., 2018	0.08 ± 0.46	24	0	5
	Leatherback	100%	Alfaro-Shigueto et al., 2007; Alfaro-Shigueto et al., 2010; Alfaro-Shigueto et al., 2018	0.02 ± 0.21	6	6	1
IG	Green	100%	Alfaro-Shigueto et al., 2010; Alfaro-Shigueto et al., 2018	-	-	-	-
	Olive ridley	100%	Alfaro-Shigueto et al., 2010; Alfaro-Shigueto et al., 2018	-	-	-	-
	Leatherback	100%	Alfaro-Shigueto et al., 2007; Alfaro-Shigueto et al., 2010; Alfaro-Shigueto et al., 2018	-	-	-	-

Table A1.2.4. Olive ridley turtle bycatch evidence throughout the Pacific East regional management unit (RMU) distributions. MEX = Mexico, GTM = Guatemala, HND = Honduras, SLV = El Salvador, NIC = Nicaragua, CRI = Costa Rica, PAN = Panama, COL = Colombia, ECU = Ecuador, PER = Peru, CHL = Chile, EPO = East Pacific Ocean.

Turtle species	RMU (Pacific East) distribution	Bycatch evidence throughout East Pacific RMU	
Olive ridley	Baja California Sur Mexico to southern Peru, the eastern Pacific and northwest of Hawaii	MEX	Koch et al., 2006; Ruiz-Slater, 2006; Sara, 2011
		GTM	Cornelius and Robinson-Clark, 1986; Eckert and Eckert, 1997; Sara, 2011; Brittain et al., 2014; Brittain, 2016
		HND	Sotelo, 2010
		SLV	Sara, 2011
		NIC	Gutiérrez, 2009; Sara, 2011
		CRI	Araya, 2006; Sara, 2011; Whoriskey et al., 2011; Dapp et al., 2013
		PAN	Sara, 2011
		COL	Rojas and Zapata, 2006; Sara, 2011
		ECU	Sara, 2011; Alfaro-Shigueto et al., 2018
		PER	Alfaro-Shigueto et al., 2010; Rosales et al., 2010; Alfaro-Shigueto et al., 2011; Sara, 2011; Alfaro-Shigueto et al., 2018
		EPO	Wallace et al., 2010b

Table A1.2.5. Green turtle bycatch evidence throughout the Pacific East regional management unit (RMU) distributions. USA = United States of America, MEX = Mexico, GTM = Guatemala, HND = Honduras, SLV = El Salvador, NIC = Nicaragua, CRI = Costa Rica, PAN = Panama, COL = Colombia, ECU = Ecuador, PER = Peru, CHL = Chile, EPO = East Pacific Ocean.

Turtle species	RMU (Pacific East) distribution	Bycatch evidence throughout East Pacific RMU	
Green	Los Angeles south, sweeping down the coast of Chile and the Eastern Tropical Pacific out to 145 West	USA	Work and Balazs, 2002
		MEX	Koch et al., 2006; Ruiz-Slater, 2006; Sara, 2011; Mancini et al., 2012
		GTM	Eckert and Eckert, 1997
		HND	-
		SLV	Sara, 2011
		NIC	Sara, 2011
		CRI	López and Arauz, 2003; Araya, 2006; Sara, 2011; Whoriskey et al., 2011
		PAN	Sara, 2011
		COL	Rojas and Zapata, 2006; Sara, 2011
		ECU	Sara, 2011; Alfaro-Shigueto et al., 2018
		PER	Alfaro-Shigueto et al., 2010; Rosales et al., 2010; Alfaro-Shigueto et al., 2011; Sara, 2011; Alfaro-Shigueto et al., 2018
		CHL	Sara, 2011; Alfaro-Shigueto et al., 2018
		EPO	Wallace et al., 2010b; Seminoff et al., 2015

Table A1.2.6. Leatherback turtle bycatch evidence throughout the Pacific East regional management unit (RMU) distributions. USA = United States of America, MEX = Mexico, GTM = Guatemala, HND = Honduras, SLV = El Salvador, NIC = Nicaragua, CRI = Costa Rica, PAN = Panama, COL = Colombia, ECU = Ecuador, PER = Peru, CHL = Chile, EPO = East Pacific Ocean.

Turtle species	RMU (Pacific East) distribution	Bycatch evidence throughout East Pacific RMU
Leatherback	From the tip of Baja California Mexico south to Chile, out to 135W	USA Work and Balazs, 2002; Carretta et al., 2004; Eguchi et al., 2017
		MEX Martínez et al., 2007
		GTM Sara, 2011
		HND -
		SLV -
		NIC Sara, 2011
		CRI Sara, 2011
		PAN -
		COL -
		ECU Zarate, 2006; Sara, 2011; Alfaro-Shigueto et al., 2018
		PER Alfaro-Shigueto et al., 2007; Alfaro-Shigueto et al., 2010; Rosales et al., 2010; Alfaro-Shigueto et al., 2011; Sara, 2011; Alfaro-Shigueto et al., 2018
		CHL Donoso and Dutton, 2010; Sara, 2011; Alfaro-Shigueto et al., 2018
		EPO Spotila et al., 2000; Wallace et al., 2010b; Wallace et al., 2013

Table A1.2.7. Summaries of the management measures contained in each management strategy scenario evaluated. IMG = San Jose inshore-midwater gillnet fleet, IG = San Jose inshore gillnet fleet.

Characterisation (controls)	Scenario 1 (status quo)		Scenario 2 (protectionist)		Scenario 3 (incentive-based)	
Avoidance	IMG fleet	IG fleet	IMG fleet	IG fleet	IMG fleet	IG fleet
Total gear switch Gillnets to trolling/potting	No	No	Yes	Yes	No	No
Minimisation						
Effort restriction 50% reduction in gillnet soak time	No	No	No	No	Yes	No
Spatial management MPAs (spatio-temporal)	No	No	No	No	Yes	Yes
Gear controls LEDs on nets	Yes	No	Yes	No	Yes	Yes
Remediation						
Post-capture survival improvements Best practice handling release workshops	Yes	No	Yes	No	Yes	Yes
Compliance and monitoring Remote electronic monitoring	Yes	No	Yes	No	Yes	Yes
Biodiversity offset						
Bycatch (Pigouvian) tax ² - funds to secondary nesting site protection for turtles in Costa Rica e.g., leatherback, olive ridley	No	No	No	No	Yes	Yes

² A bycatch (Pigouvian) can be a double dividend tax, acting as both as an offset and minimisation strategy. Ideally proportional to the turtle bycatch mortality rate (bycatch/population size) on an eastern Pacific pelagic longline fishery. The tax minimizes bycatch by internalising the external costs of bycatch (for both consumers and producers as part of the tax is passed up the supply chain, depending upon the price elasticities of demand and supply). The first dividend is the welfare increase (including conservation) from minimisation through the bycatch tax and the second dividend, and an additional source of welfare increase (including conservation), comes from the offset (Squires et al. 2018).

Appendix 2

A2.1. Additional summary statistics - green turtle observer analysis

Table A2.1.1. Hausman test data where the null hypothesis is that the preferred model is random effects versus the alternative hypothesis that the preferred model is fixed effects. df = degrees of freedom.

Data: Green turtle bycatch (binomial) ~ GRT + Season + Year + Soak time + Net (km) + Crew. Panel data index = vessel identification.		
Chi squared = 6.6852	df = 6	p-value = 0.3509
alternative hypothesis: one model is inconsistent.		

Failing to reject the null hypothesis of random effects (against fixed effects) I proceeded with a Generalised Linear Mixed Model (GLMM) to integrate both fixed and random effect variables into our model.

Table A2.1.2. Results from the binomial generalised linear mixed model for predicting probability of green turtle catch, where vessel ID is a random effect (re). Models are ranked by Delta AIC scores, with Delta BIC scores also presented. df = degrees of freedom, GRT = gross registered tonnage. The chosen model is in bold text.

Green turtle catch

Rank	Model	df	Δ AIC	Δ BIC
1	GRT + Year + Season + Vessel (re)	2501	0	0
2	GRT + Year + Season + Crew + Vessel (re)	2500	1.4	7.2
3	GRT + Year + Season + Soak time + Vessel (re)	2500	1.7	7.6
4	GRT + Year + Season + Soak time + Crew + Vessel (re)	2499	3.1	14.8
5	GRT + Year + Season + Net length (km) + Crew + Vessel (re)	2499	3.4	15.0
6	GRT + Year + Season + Soak time + Net length (km) + Vessel (re)	2499	3.7	15.4
7	GRT + Year + Season + Soak time + Net length (km) + Crew + Vessel (re)	2498	5.1	22.6
8	GRT + Year + Vessel (re)	2502	78.9	67.2
9	GRT + Year + Crew + Vessel (re)	2501	80.6	74.7
10	GRT + Year + Soak time + Net length (km) + Crew + Vessel (re)	2499	84.4	90.3
11	GRT + Season + Net length (km) + Crew + Vessel (re)	2501	145.0	139.2
12	GRT + Season + Soak time + Net length (km) + Crew + Vessel (re)	2500	146.8	146.8
13	GRT + Vessel (re)	2504	150.8	127.5
14	GRT + Crew + Vessel (re)	2503	152.7	135.2
15	GRT + Net length (km) + Crew + Vessel (re)	2502	153.6	141.9
16	GRT + Soak time + Net length (km) + Crew + Vessel (re)	2501	155.5	149.6

Table A2.1.3. Summary of observer coverage across the inshore-midwater fleet by vessel size class. GRT = gross registered tonnage.

GRT class	Number of vessels	Number of trips	Number of sets
1<4	20	53	291
4<8	4	181	1099
8<12	7	208	1278
>12	1	3	17
Total	32	445	2685

Table A2.1.4. Extrapolated seasonal and annual green turtle catch estimates calculated from observer data without Generalised Linear Mixed Model (GLMM) class weightings. Temp. Grp. = temporal grouping; winter represents the cold weather months of June to November, summer represents the warm weather months of December to May.

Temp. Grp.	Mean	Min 90 CI	Max 90 CI
Winter	340.37	255.27	428.99
Summer	797.47	598.08	1005.10
Total net encounters p.a.	1137.85	853.35	1434.09

Table A2.1.5. Green turtle capture per unit effort / per trip weighted by vessel size class (gross registered tonnage). Gross registered tonnage for weightings were obtained from the Generalised Linear Mixed Model (GLMM). SE = standard error of the mean, CI = confidence interval.

GRT class	Coef.	SE	Green turtle weighted mean	Green turtle (-90 CI)	Green turtle (+90 CI)
1<4	0	0.000	0.71	0.56	0.86
4<8	0.036	0.030	0.75	0.60	0.75
8<12	0.050	0.026	0.76	0.61	0.76
>12	0.104	0.083	0.82	0.67	0.82

Table A2.1.6. Approximated inshore-midwater gillnet fleet size in San Jose. Actively fishing gillnet vessels in 2008 are based on expert opinion from researcher's surveying in San Jose that year (JAS & JCM). The fleet size was most recently recorded in a census survey in the winter of 2017 and key informant interviews provided estimates of the 2017 summer fleet size. Seasonal differences reflect the proportional difference identified from data the 2017 census data, which is supported by key informant interviews and focus discussion groups held in San Jose. An incremental decay of three vessels per year were applied from 2007 to 2019. Gillnet fleet size is declining as skippers and crew change from gillnets to squid jigging. The winter fishing season is June-November and the summer fishing season is December-May.

Year	Vessel Number (Summer)	Vessel Number (Winter)
2007	63	48
2008	60	45
2009	57	42
2010	54	39
2011	51	36
2012	48	33
2013	45	30
2014	42	27
2015	39	24
2016	36	21
2017	33	18
2018	30	15
2019	27	12

Table A2.1.7. Extrapolated green turtle capture estimates per season based on catch-per-unit-effort (CPUE) per trip. CPUE was weighted using Generalised Linear Mixed Model (GLMM) size class coefficients. Annual values were summed across gross registered tonnage weight classes. CI = confidence interval.

Year	Green turtle CPUE / Summer	Green summer (-90 CI)	Green summer (+90 CI)	Green turtle CPUE / Winter	Green winter (-90 CI)	Green winter (+90 CI)
2007	1199.54	953.15	1353.53	597.57	474.83	674.29
2008	1142.42	907.76	1289.08	560.22	445.15	632.14
2009	1085.29	862.37	1224.62	522.87	415.47	590.00
2010	NA	NA	NA	NA	NA	NA
2011	NA	NA	NA	NA	NA	NA
2012	NA	NA	NA	NA	NA	NA
2013	856.81	680.82	966.81	373.48	296.77	421.43
2014	799.69	635.43	902.35	336.13	267.09	379.29
2015	742.57	590.04	837.90	298.79	237.41	337.14
2016	685.45	544.65	773.45	261.44	207.74	295.00
2017	628.33	499.27	708.99	224.09	178.06	252.86
2018	571.21	453.88	644.54	186.74	148.38	210.71
2019	514.09	408.49	580.08	149.39	118.71	168.57
Mean	822.54	653.59	928.13	351.07	278.96	396.14

A2.2. Additional leatherback turtle analysis

A2.2.1. Elicited judgements for leatherback turtle capture rate with gillnets

In addition to eliciting participants' judgements for capture rates of green turtles, I also asked participants to quantify capture rates for leatherback turtles (*Dermochelys coriacea*) in gillnets set by inshore-midwater vessels operating from San Jose, Lambayeque, Peru (6°46' S, 79°58' W).

Participants' judgements for leatherback turtle captures were 4.8 – 15.2 individuals per month (Fig. S4.3). Skippers' judgements (best estimates) were higher for leatherback turtles than the not-for-profit employees. I then used participants' monthly turtle capture rates to infer seasonal capture rates, as well as calculating an annual capture rate for leatherback turtles by adding the summer and winter encounters together.

A2.2.2. Comparison of elicited judgements with at sea fisheries observer data

Following the analysis undertaken for green turtles presented in the main text, GLMMs were used to estimate the predictive power of vessel weight class for leatherback turtle catch while controlling for seasonal and annual temporal variations, fishing effort (gillnet soak time), and inter-vessel variation within the fleet as a random effect. GRT and a random effect for vessel resulted in the best model (Tables S4.1 and S4.2).

In contrast to green turtles, vessel size was found to be weakly negatively correlated to leatherback turtle catch, however, following correcting for serial correlation, no significance was identified (Table S4.2). The low and sporadic catch rate of leatherback turtles across the observer dataset (n=7) resulted in the GLMM model having little predictive power in terms of leatherback catchability in relation to our vessel size classes. I caution readers when interpreting the presented outputs in this analysis presented in supporting information. Based on this observer data and GLMM output, I extrapolated gillnet leatherback turtle capture estimates for the inshore-midwater gillnet fleet in San Jose as an estimated 19.18 (5 – 32) individuals per year (Table S4.3).

A2.2.3. Assessing participant performance

Participants' judgements were more precise at estimating catch rates for leatherback turtles than green turtles. Leatherback turtles are infrequently captured in this fishery (Table 1 – Main text); it is possible that these fishers were able to recall these rare capture events with more precision than for green turtles, which are more frequently captured, due to the lasting impression that encountering this species leaves. Leatherback turtles are more easily differentiated in size and by their distinct soft leather-like shell from the other hard-shelled sea turtles that are captured in the San Jose fishing system (green, hawksbill, and olive ridley turtles; Alfaro-Shigueto et al. 2011; Alfaro-Shigueto et al. 2018). Indeed, good recollection of rare capture events is reflected in the findings of other studies eliciting local knowledge for species counts (van der Hoeven et al. 2004; Brittain et al. 2020). Participant L05 (not-for-

profit) submitted accurate leatherback turtle judgements despite very tight confidence bounds. Participant L01 (gillnet skipper) accurately estimated leatherback turtle captures in winter.

Table A2.2.3.1. Results from the binomial generalised linear mixed model for predicting probability of leatherback turtle catch, where vessel ID is a random effect (re). Models are ranked by Delta AIC scores, with Delta BIC scores also presented. df = degrees of freedom, GRT = gross registered tonnage. The chosen model in bold text.

<i>Leatherback turtle catch</i>				
Rank	Model	df	Delta AIC	Delta BIC
1	GRT + Vessel (re)	2504	0	0
2	GRT + Crew + Vessel (re)	2503	1.528	7.367
3	GRT + Year + Vessel (re)	2502	2.777	14.454
4	GRT + Net length (km) + Crew + Vessel (re)	2502	3.247	14.924
5	GRT + Year + Season + Vessel (re)	2501	3.628	21.143
6	GRT + Season + Net length (km) + Crew + Vessel (re)	2501	3.882	21.397
7	GRT + Year + Crew + Vessel (re)	2501	4.408	21.924
8	GRT + Year + Season + Crew + Vessel (re)	2500	5.179	28.533
9	GRT + Soak time + Net length (km) + Crew + Vessel (re)	2501	5.24	22.755
10	GRT + Year + Season + Soak time + Vessel (re)	2500	5.614	28.968
11	GRT + Season + Soak time + Net length (km) + Crew + Vessel (re)	2500	5.867	29.221
12	GRT + Year + Season + Net length (km) + Crew + Vessel (re)	2499	6.894	36.086
13	GRT + Year + Season + Soak time + Crew + Vessel (re)	2499	7.162	36.354
14	GRT + Year + Season + Soak time + Net length (km) + Vessel (re)	2499	7.309	36.501
15	GRT + Year + Soak time + Net length (km) + Crew + Vessel (re)	2499	8.209	37.401
16	GRT + Year + Season + Soak time + Net length (km) + Crew + Vessel (re)	2498	8.877	43.908

Table A2.2.3.2. Best fit model for predicting probability of leatherback turtle catch chosen following AIC and BIC ranking criteria.

<i>Leatherback turtle bycatch</i>				
Reference	Random effects	Intercept	Residual	n
Vessel	Std. dev	1.05E-02	4.85E-02	32
	Fixed effects	Coefficient	SE ₁	p-value
	Intercept	0.006	0.007	0.41
GRT				
<i>reference = 0 < 4 GRT</i>	4 < 8 GRT	-0.009	0.008	0.2732
	8 < 12 GRT	-0.008	0.006	0.2281
	> 12 GRT	-0.010	0.016	0.5314
	Net (km)	0.001	0.002	0.5962
	Crew number	0.001	0.001	0.5032

₁ Serial correlation-consistent standard errors

Table A2.2.3.3. Extrapolated seasonal and annual reductions in leatherback turtle captures with small-scale fishery gillnets set from vessels launching from San, Jose, Peru, based on elicited monthly estimates of the efficacy of the bycatch reduction strategy scenario of gear switching from gillnets to potting or trolling, and at sea fisheries observer data obtained from the period August 2007–May 2019. Temp. Grp. = temporal grouping; winter represents the cold weather months of June to November, summer represents the warm weather months of December to May. CI = credible interval. Note that no weighting by GLMM coefficients were applied to the extrapolated bycatch rates calculated from the observer data.

Temp. Grp.	Expert elicitation data			Observer data		
	Mean best (B)	Std. lower 90 CI (lsi)	Std. upper 90 CI (usi)	Mean	Min 90 CI	Max 90 CI
Monthly/winter	7.17	4.76	9.48	2.24	0.59	4.00
Monthly/summer	12.03	9.65	15.16	3.20	0.84	5.70
Total winter	43	28.57	56.89	5.74	1.52	10.23
Total summer	72.2	57.92	90.95	13.44	3.55	23.98
Annual	115.2	86.49	147.84	19.18	5.07	34.21

Table A2.2.3.4. Leatherback turtle catch-per-unit-effort (CPUE) per trip weighted by vessel size class (gross registered tonnage). Gross registered tonnage for weightings were obtained from the Generalised Linear Mixed Model (GLMM). SE = standard error of the mean, CI = confidence interval.

GRT class	Coef.	SE	Leatherback turtle weighted mean	Leatherback turtle (-90 CI)	Leatherback turtle (+90 CI)
1<4	0.000	0.000	0.015	0.01	0.02
4<8	-0.009	0.008	0.006	0.00	0.02
8<12	-0.008	0.006	0.007	0.00	0.02
>12	-0.010	0.016	0.005	0.00	0.01

Table A2.2.3.5. Extrapolated seasonal and annual leatherback turtle catch estimates calculated from observer data without Generalised Linear Mixed Model (GLMM) class weightings. Temp. Grp. = temporal grouping; winter represents the cold weather months of June to November, summer represents the warm weather months of December to May.

Temp. Grp.	Mean	Min 90 CI	Max 90 CI
Winter	7.28	2.78	12.07
Summer	17.06	6.52	28.29
Total net encounters p.a.	24.33	9.30	40.36

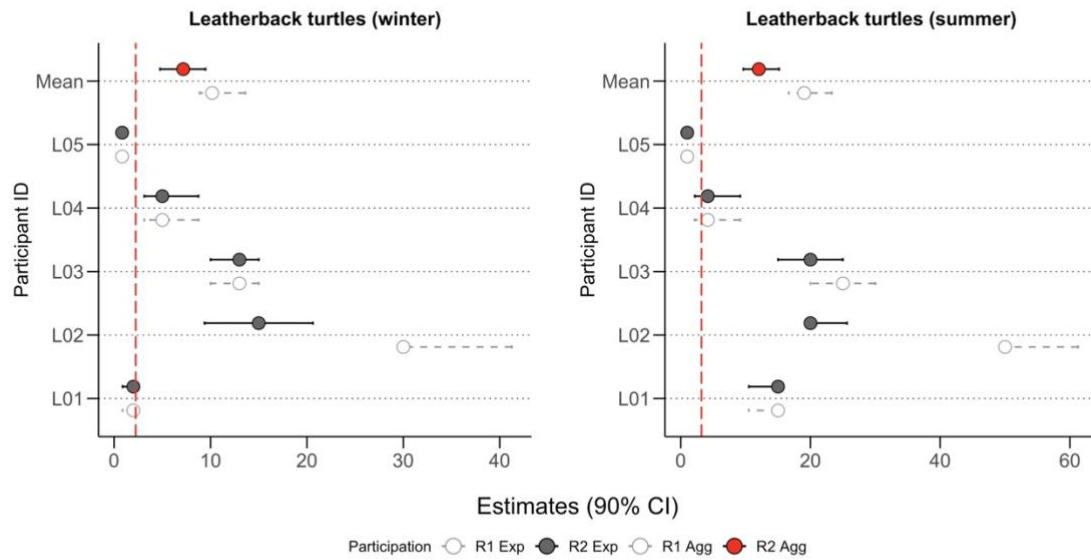


Figure A2.2.3.1. Comparing elicited seasonal estimates of the number of leatherback turtles saved from encountering gillnets set by the inshore-midwater fleet as a result of a total gear switch from gillnets to a fishing gear that results in no turtle bycatch (such as lobster potting or trolling) to current bycatch rates calculated from voluntary, at sea human observer data in San Jose, Peru. Elicited estimates are divided into cold weather months and warm weather months fishing seasons. Experts assumed 100% compliance with the total gear switch scenario. Uncertainty bars have been adjusted to reflect 90% credible intervals for each expert's response. Red dotted line shows the extrapolated estimates of turtle catch from the observer data.

Table A2.2.3.6 Extrapolated leatherback turtle capture estimates per season based on catch-per-unit-effort (CPUE) per trip. CPUE was weighted using Generalised Linear Mixed Model (GLMM) size class coefficients. Annual values were summed across gross registered tonnage weight classes. CI = confidence interval.

Year	Leatherback turtle CPUE / Summer	Leatherback summer (-90 CI)	Leatherback summer (+90 CI)	Leatherback turtle CPUE / Winter	Leatherback winter (-90 CI)	Leatherback winter (+90 CI)
2007	19.60	5.18	34.97	9.77	2.58	17.42
2008	18.67	4.93	33.30	9.16	2.42	16.33
2009	17.74	4.68	31.64	8.55	2.26	15.24
2010	NA	NA	NA	NA	NA	NA
2011	NA	NA	NA	NA	NA	NA
2012	NA	NA	NA	NA	NA	NA
2013	14.00	3.70	24.98	6.10	1.61	10.89
2014	13.07	3.45	23.31	5.49	1.45	9.80
2015	12.14	3.21	21.65	4.88	1.29	8.71
2016	11.20	2.96	19.98	4.27	1.13	7.62
2017	10.27	2.71	18.32	3.66	0.97	6.53
2018	9.34	2.47	16.65	3.05	0.81	5.44
2019	8.40	2.22	14.99	2.44	0.64	4.35
Mean	13.44	3.55	23.98	5.74	1.52	10.23

Appendix A2.3 Elicitation data survey

Type 2: Full Oral Information Giving and Consent Seeking Process

Record this consent process using a digital recorder (if participant has consented to this) or by using a Record of Consent Form.

[Oral information giving stage]

Hello, my name is Bruno Ibañez Erquiaga and this is my colleague William Arlidge. We wondered if you'd be interested in being involved in a short follow-up survey that follows on from the survey you took part in last year. This work is part of William's PhD research at the University of Oxford in the Department of Zoology. This research is part of his degree on managing fisheries sustainably. Can I tell you more about the study? *[Await confirmation]*

We want to investigate how information flows between fishermen who use nets in San Jose in order to understand how best to introduce sustainable fishing practices in future and to understand the cost of fishing in the Peru surface gillnet fishery.

Last year we invited you to a meeting with other randomly selected chalana skippers from San Jose to hear your thoughts on various strategies to reduce the number of turtles captured in your nets and then asked you to rank each strategy on a scale from 1 to 5, with 1 indicating you strongly disagreed with the idea, and 5 indicating that you agreed with it. We also asked you to estimate the distances from San Jose that you and the other chalana/lancha *[delete as appropriate]* fishermen in San Jose operate.

We are now interested in understanding in more detail how many turtles you think might be saved by each of the suggested strategies we asked you to rank last year.

If you choose to be a part of this project, here is what will happen:

I will give you a survey that will last around 45 minutes where I will first show you some summery data from last year's survey, hear your thoughts on whether this summary information is correct, then ask you to estimate the max, min, and average number of turtles you think will be saved by each strategy, followed by asking for a percentage estimate of how sure you are about your estimates.

The answers you give will contribute to data for William's DPhil thesis and several academic publications. The personal information you will share with us will not be passed to any third party. Data protection will be of the highest priority. There will be no disclosure of any data that could place you at risk of criminal or civil liability and all data will be anonymised, held on a secure server and treated in the strictest confidence.

This research is anonymous, which means that in any publications, your name will not be used, unless you insist on the opposite.

In order to mitigate any potential risks, we will ensure that none of the information that you provide will be passed on to any third party, all information will be stored on an encrypted hard drive on William's computer and no names will be used in any publications.

Taking part is completely voluntary and we can stop any time you like without giving a reason and without any negative consequences.

With your permission, we would like to make an audio recording of our discussion to make sure we are getting an accurate record of your thoughts. Alternatively, I can take notes in my notebook. Which would you prefer? William may want to re-contact you to clarify information you gave me in your interview. In that case, we will ask you if you have time to answer some more questions.

William will safely store your data on his university laptop with Whole Disk Encryption that will be kept with him throughout the field work period and the remaining period of his studies at the University of Oxford. At the end of the project, all research data will be preserved and saved for a minimum of three years (as per the University of Oxford's policy). Data will be stored at the University of Oxford via the Bodleian Library ORA-Data archive. It will not be in the public domain and their use will be restricted to specific purposes.

If you agree to take part in this project, the research will be written up as a thesis.

On successful submission of the thesis, it will be deposited both in print and online in the University archives, to facilitate its use in future research.²

The research will also be published in academic journals.

If you have any complaints or concerns please feel free to contact William in the first instance. His mobile is [provided]. You can also reach him at william.arlidge@zoo.ox.ac.uk.

This research project has been reviewed and approved by a University of Oxford ethics committee. If, after contacting William or myself with any concern, you remain unhappy and wish to make a formal complaint, please contact the ethics committee. Their email is ethics@socsci.ox.ac.uk. William will also give you their postal address and this project's ethics reference number.

You must be 18 years or over to participate in this study.

Do you have any questions?

[Oral consent seeking stage, after participant has had sufficient time to think about whether s/he wants to take part]

Do you give your permission for me to interview you? Do you give me permission to audio record you? Do you give your permission for me to re-contact you to clarify information?

Are you happy for me to collect and detail sensitive personal data?

Are you happy to take part?

Ok, thanks, in which case let's start.

[Expert elicitation stage]

We're really grateful for your help in getting this information. We are interested in how many green and leatherback turtles will be saved, for each of the management strategies that you

² Oxford students following D.Phil., M.Litt. and M.Sc. (by Research) courses should refer to http://www.bodleian.ox.ac.uk/ora/oxford_theses.

previously ranked. This is the actual number of turtles that would otherwise been captured in nets in a given month in winter and summer. We're interested in the number of turtles from the whole San Jose *chalana/lancha* [delete as appropriate] gillnet fishery, not just your boats, and we're interested in the number of turtle captures in nets.

Just to be clear, I am an independent student. I am not proposing that any of these measures actually should be implemented. I am trying to understand the impacts of each of these types of approach on both skippers and turtles. We asked last time about your views on how these would affect YOU. Now we are asking about the TURTLES. So please do answer honestly, and please set aside for the moment any views you have about the feasibility or acceptability of a given approach, and think just about their effects on turtle mortality.

For each of the approaches I am going to start by asking you for your best estimate of the minimum number of turtles that would not captured using the approach, from the whole fleet, in Winter. This is assuming that EVERYONE COMPLIES with the method – i.e., we're thinking about the intrinsic benefit that the approach could have for turtles based on your experience, not about how it would work out in practice.

Next, I am going to ask you about the maximum number of turtles you think could be [saved] using this method, in winter. The difference between minimum and maximum could be because of different turtle distributions in different years, or just by chance how many turtles happen to swim into a net in a given season.

Then I will ask you your best guess, about what the number would be in a "typical" winter. This is not necessarily the middle of the two numbers of course.

And finally, for each one of these approaches I will ask you how certain you are about your estimates. For some of them it might be that your estimates are based on quite a lot of guesswork, and for others you might be pretty sure, based on your experience. I'll ask you to give me this as a %, where 100% means you're certain that the true number for your estimate will fall within your given minimum to maximum range, and you think your best guess at a typical year is very close to the truth. And 50% is when you are quite unsure and really you don't know at all.

As a trial, I'd like you to estimate how many beers you think my New Zealand colleague, William, could drink in an evening with you, without falling over? What's your minimum estimate, maximum, best guess? And given your experiences with him, how sure are you, between 50% and 100%, that the average number of beers William can drink without falling over will be within your minimum and maximum interval?

[Question sets for the focus group were randomised using a random number generator]

To estimate the mean number of green turtle species saved by a particular bycatch reduction strategy (e.g., lights on gillnets) per month in winter, one would ask the following:

Realistically, what do you think could be the lowest number of green turtles that could be saved using lights on gillnets per month in winter? Realistically, what do you think could be the highest number of green turtles that could be saved using lights on gillnets per month in winter? What is your best estimate (the most likely value) of green turtles that could be saved using lights on gillnets per month in winter? For the interval created (lower and upper bound),

what is the probability between 50% and 100% that the number of green turtles saved by using lights on gillnets per month in winter will fall within this interval?

REMEMBER: We're interested in the number of turtles from the *whole San Jose chalana/lancha [delete as appropriate] gillnet fishery*, not just your boat(s), and we're interested in the number of *turtle captures* in nets. We don't want you to think about the inshore/inshore-midwater *[delete as appropriate] gillnet vessels, purse seine, or squid [pota] vessels*.

Q1: Spatial gillnet ban - prohibited driftnet fishing distance extended around Lobo de Tierra and Lobo de Afuera to 15 nautical miles offshore the islands. All year.

	Winter		Summer	
	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q2: Temporal gillnet ban between August to November. Gear switching to lobster potting or hand line fishing during the gillnet ban period every year.

	Winter		Summer	
	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q3: Spatial and temporal gillnet ban – gillnet ban shifting in space and time in relation to turtle movement.

	Winter	Summer
--	--------	--------

	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q4: Offshore distance restriction – gillnetting only allowed to occur between 0-2 n.m. offshore

	Winter		Summer	
	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q5: Total gillnet ban. Gear switching to lobster potting or hand line fishing

	Winter		Summer	
	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q6: Restrictions on soak time of gillnets (6 hours during daylight hours) [inshore-midwater group only]

	Winter	Summer
--	--------	--------

	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q7: Buoyless (buoys removed from float line) nets

	Winter		Summer	
	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q8: Using fixed (set) gillnets over drift gillnets

	Winter		Summer	
	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q9: LED lights on gillnets

	Winter		Summer	
	Green	Leatherback	Green	Leatherback

What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q10: Annual workshops on safe handling and release procedures, which includes the resuscitation of sea turtles

	Winter		Summer	
	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q11: Annual fee (bycatch tax) to fund turtle nesting site protection e.g., unprotected smaller nesting sites in Peru, Ecuador, Costa Rica, or Mexico (depending on species).

	Winter		Summer	
	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q12: Fishermen community enforcement in the 5-nautical mile Marine Protected Area around Lobo de Tierra and Lobo de Afuera [In-kind payment for ecosystem services program]

	Winter	Summer
--	--------	--------

	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Q13: Requirement for electronic monitoring device on all chalana/lancha [*delete as appropriate*] boats launching from San Jose

	Winter		Summer	
	Green	Leatherback	Green	Leatherback
What is the lowest the value could be?				
What is the highest the value could be?				
What is your best estimate (the most likely value)?				
How confident are you that the interval you provided contains the truth (provide an answer in the range of 50-100%)?				

Appendix A2.4. Elicitation Data

Only the elicitation data from the total gillnet gear ban strategy was integrated into the Chapter 4 analysis.

Question 1: How many green turtles in winter per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required?

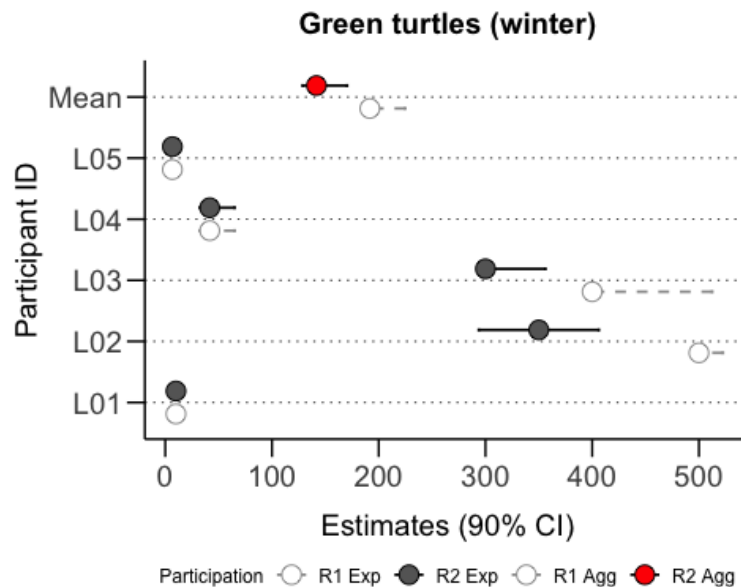


Figure A.2.4.1. Round 1 and 2 estimates for Question 1: How many green turtles in winter per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required? Intervals have been standardised to 90%. The graph shows estimates for each expert in Round 1 (R1 Exp), and Round 2 (R2 Exp) and the aggregations in Round 1 (R1 Agg), Round 2 (R2 Agg).

Table A2.4.1. Round 1 and 2 estimates for Question 1: How many green turtles in winter per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required? Intervals have been standardised to 90%. Conf = confidence bound estimate.

ID	Stakeholder group	Round	Lower	Upper	Best	Conf	Respondent comments
Mean	Mean	NA	189.79	224.85	191.67	90%	NA
Mean	Mean	NA	128.54	170.35	141.67	90%	NA
L01	Gillnet skipper	1	10.00	16.43	10.00	90%	I think all turtles that are usually captured in San Jose nets would be saved with this strategy. We don't fish as much in winter as we do in summer, so I am considering the differences in how much we fish between seasonal periods.
L01	Gillnet skipper	2	10.00	16.43	10.00	90%	I don't have any changes to make to my original estimates.
L02	Gillnet skipper	1	500.00	522.50	500.00	90%	Green turtles are captured in the highest numbers in our nets. I think this trend is consistent across the fleet.
L02	Gillnet skipper	2	293.75	406.25	350.00	90%	I am readjusting my estimate down as I had a really high monthly turtles saved, I was thinking too much about fishing further north and not considering more southern inshore-midwater boats launching from San Jose.
L03	Gillnet skipper	1	400.00	512.50	400.00	90%	I think that all the turtles that are usually captured in nets would be saved if there was a total ban and we switched to these fishing methods.
L03	Gillnet skipper	2	300.00	356.25	300.00	90%	When I consider that more green turtles are often caught when we head north rather than south, I'm going to readjust my estimate down as captures may not be evenly spaced
L04	Not-for-profit	1	32.29	65.10	41.67	90%	No comment.
L04	Not-for-profit	2	32.29	65.10	41.67	90%	I don't have any changes to make to my original estimates.
L05	Not-for-profit	1	6.67	7.74	6.67	90%	Here I am thinking about the total number of turtles likely captured by the San Jose fleet.
L05	Not-for-profit	2	6.67	7.74	6.67	90%	I don't have any changes to make to my original estimates.

Question 2: How many green turtles in summer per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required?

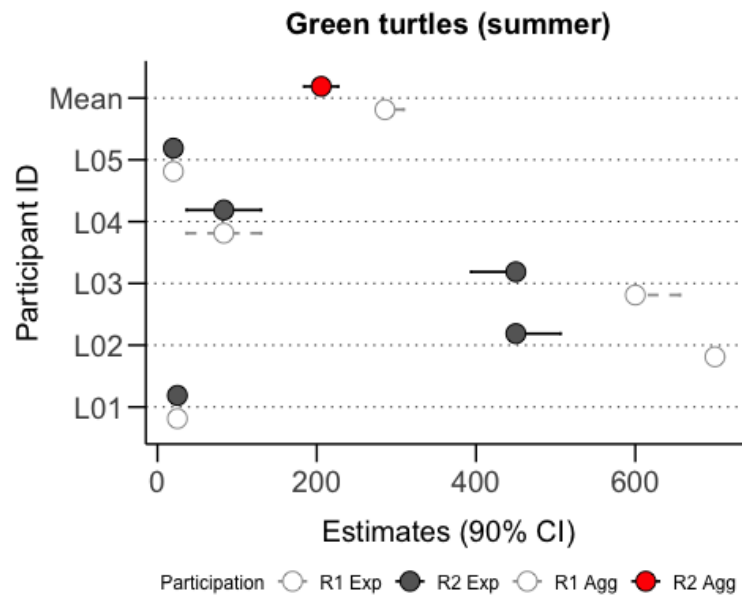


Figure A2.4.2. Round 1 and 2 estimates for Question 2: How many green turtles in summer per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required? Intervals have been standardised to 90%. The graph shows estimates for each expert in Round 1 (R1 Exp), and Round 2 (R2 Exp) and the aggregations in Round 1 (R1 Agg), Round 2 (R2 Agg).

Table A2.4.2. Round 1 and 2 estimates for Question 2: How many green turtles in summer per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required? Intervals have been standardised to 90%. Conf = confidence bound estimate.

ID	Stakeholder group	Round	Lower	Upper	Best	Conf	Respondent comments
Mean	Mean	NA	275.39	309.34	285.67	90%	NA
Mean	Mean	NA	184.14	227.09	205.67	90%	NA
L01	Gillnet skipper	1	20.50	25.00	25.00	90%	My summer estimates are higher than my winter estimates as we are generally fishing more days when the sea is not so rough and they do not close the beach due to danger from waves.
L01	Gillnet skipper	2	20.50	25.00	25.00	90%	I don't have any changes to make to my original estimates. I don't catch that many green turtles in my nets. I think some of these other estimates are too high.
L02	Gillnet skipper	1	700.00	711.25	700.00	90%	I am considering how many green turtles encounter my nets and multiplying out. Sometimes we get a single haul with between 30-40 green turtles in it. These are all released, but in summer numbers can be high. In winter captures are lower as we fish less.
L02	Gillnet skipper	2	450.00	506.25	450.00	90%	The same as my green turtle winter estimate - I am going to readjust my estimate down as I think my first estimates were too high due to not considering how turtle captures are often lower when we fish further south.
L03	Gillnet skipper	1	600.00	656.25	600.00	90%	This strategy would be highly effective for reducing turtle encounters with nets; I don't think any turtles would be captured using handlines or potting and we often encounter these in nets, so I imaging across the fleet this would be reasonably high numbers.
L03	Gillnet skipper	2	393.75	450.00	450.00	90%	It seems like some other skippers may capture lower numbers of green turtles than I do, so perhaps I was overestimating. I would like to readjust.
L04	Not-for-profit	1	36.46	130.21	83.33	90%	No comment.
L04	Not-for-profit	2	36.46	130.21	83.33	90%	I don't have any changes to make to my original estimates.
L05	Not-for-profit	1	20.00	24.00	20.00	90%	Thinking about the total number of turtles likely captured by the San Jose fleet.
L05	Not-for-profit	2	20.00	24.00	20.00	90%	I don't have any changes to make to my original estimates.

Question 3: How many leatherback turtles in winter per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required?

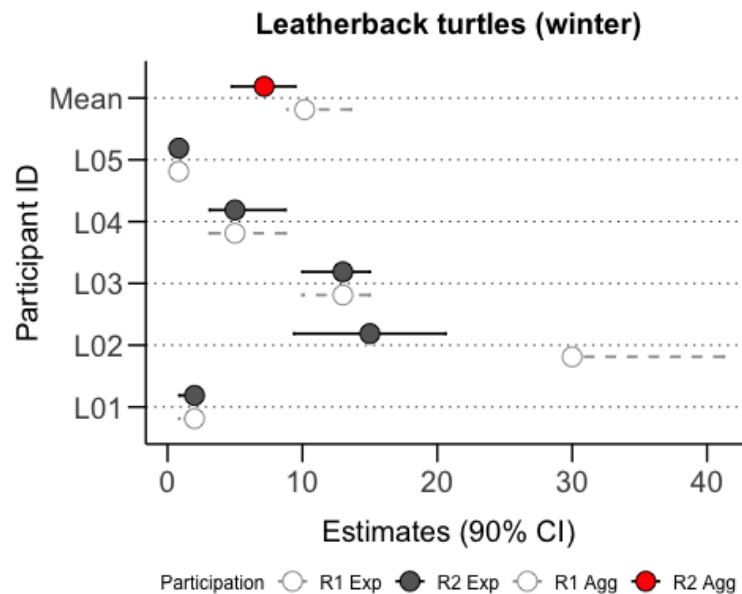


Figure A2.4.3. Round 1 and 2 estimates for Question 7: How many leatherback turtles in winter per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required? Intervals have been standardised to 90%. The graph shows estimates for each expert in Round 1 (R1 Exp), and Round 2 (R2 Exp) and the aggregations in Round 1 (R1 Agg), Round 2 (R2 Agg).

Table A2.4.3. Round 1 and 2 estimates for Question 3: How many leatherback turtles in winter per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required? Intervals have been standardised to 90%. Conf = confidence bound estimate.

ID	Stakeholder group	Round	Lower	Upper	Best	Conf	Respondent comments
Mean	Mean	NA	8.89	13.61	10.17	90%	NA
Mean	Mean	NA	4.76	9.48	7.17	90%	NA
L01	Gillnet skipper	1	0.88	2.00	2.00	90%	All turtles that are usually captured in nets would be saved with this strategy
L01	Gillnet skipper	2	0.88	2.00	2.00	90%	I don't have any changes to make to my original estimates.
L02	Gillnet skipper	1	30.00	41.25	30.00	90%	No comment.
L02	Gillnet skipper	2	9.38	20.62	15.00	90%	I am considering the captures of laud (leatherback turtles) that I hear about and then extrapolating out. Captures definitely occur.
L03	Gillnet skipper	1	10.00	15.00	13.00	90%	All turtles that are usually captured in nets would be saved if there was a total ban, I would think a maximum of 15 turtles per month in winter would be a good estimate.
L03	Gillnet skipper	2	10.00	15.00	13.00	90%	I don't have any changes to make to my original estimates.
L04	Not-for-profit	1	3.12	8.75	5.00	90%	No comment
L04	Not-for-profit	2	3.12	8.75	5.00	90%	I don't have any changes to make to my original estimates.
L05	Not-for-profit	1	0.43	1.03	0.83	90%	Thinking about the total number of turtles likely captured by the San Jose fleet
L05	Not-for-profit	2	0.43	1.03	0.83	90%	I don't have any changes to make to my original estimates.

Question 4: How many leatherback turtles in summer per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required?

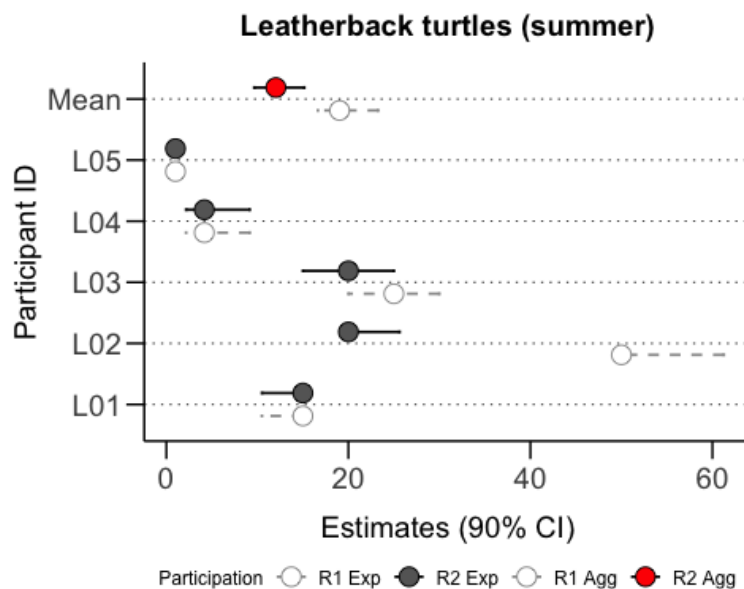


Figure A2.4.4. Round 1 and 2 estimates for Question 4: How many leatherback turtles in summer per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required? Intervals have been standardised to 90%. The graph shows estimates for each expert in Round 1 (R1 Exp), and Round 2 (R2 Exp) and the aggregations in Round 1 (R1 Agg), Round 2 (R2 Agg).

Table A2.4.4. Round 1 and 2 estimates for Question 3: How many leatherback turtles in summer per month would be saved using a total gillnet ban, with gear switching to lobster potting or handline fishing required? Intervals have been standardised to 90%. Conf = confidence bound estimate.

ID	Stakeholder group	Round	Lower	Upper	Best	Conf	Respondent comments
Mean	Mean	NA	16.65	23.28	19.03	90%	NA
Mean	Mean	NA	9.65	15.16	12.03	90%	NA
L01	Gillnet skipper	1	10.50	15.00	15.00	90%	I think laud (leatherback turtles) in particular wouldn't be captured by handlines, there would be little overlap as handline fishers don't venture too far from the coast.
L01	Gillnet skipper	2	10.50	15.00	15.00	90%	I don't have any changes to make to my original estimates.
L02	Gillnet skipper	1	50.00	61.25	50.00	90%	No comment.
L02	Gillnet skipper	2	20.00	25.62	20.00	90%	I am readjusting my estimate down. When I consider how many leatherback turtles I hear about, I think that these estimates were too high at first. I think these occur more in summer. We are meant to let the local IMARPE officer know if we catch them
L03	Gillnet skipper	1	20.00	30.00	25.00	90%	No comment.
L03	Gillnet skipper	2	15.00	25.00	20.00	90%	Considering that I rarely hear about laud (leatherback turtles) captures and looking at the other estimates, I think I was too high with my first estimate
L04	Not-for-profit	1	2.17	9.17	4.17	90%	No comment.
L04	Not-for-profit	2	2.17	9.17	4.17	90%	I don't have any changes to make to my original estimates.
L05	Not-for-profit	1	0.60	1.00	1.00	90%	Thinking about the total number of turtles likely captured by the San Jose fleet
L05	Not-for-profit	2	0.60	1.00	1.00	90%	I don't have any changes to make to my original estimates.

Appendix 3

A3.1. Structural differences between information-sharing contexts

A3.1.1. Assortativity

The analysis of network assortativity (presented in the main text) found that networks of turtle bycatch information-sharing nominations show no significant assortativity in comparison to both the edge permutation null models (Figure 5.2b). Individual gillnet skippers had a propensity to be disproportionately connected to other gillnet skippers who had nominated a similar number of people as they had (out-assortativity). Although none of the information-sharing contexts were significantly different from the edge null models in their out-assortativity, the sharing of information regarding turtle bycatch was the only context that was slightly lower than expected, whilst all other contexts were higher than expected (Figure A3.1.2 and Table A3.1.2). The lack of significant differences here is probably due to the relatively low variance in out-going links in comparison to in-going links (i.e., due to the questionnaire set-up the number of nominations an individual could make was limited – see Methods in main text), and is most likely driven by a carry-over of the strong patterns evident in the in-going nomination assortativity.

The analysis shows that the random assortment in the turtle bycatch context is most likely a result of more complex dyadic-level behaviour patterns driving each individual's attitudes and behaviours. This is because the assortment statistic itself is the level of like-to-like connectivity given the total number of links. The edge permutations (edge null model 2) also (a) directly control for the number of out-going and in-going links in each context (Figure 5.2c), and (b) still find that assortment is not significantly different in the turtle context, but significantly differently in the other contexts. These comparisons are over and above that which would be expected from the differences in the number of links, or even the degree distributions, specific to each context.

A3.1.2. Individual Centrality

When considering the variance in betweenness (as an alternative measure of centrality; fig. S3), or the mean eccentricity of each network's nodes (rather than the variance; Figure A3.1.4), I found that the observed statistics from all contexts (including turtle bycatch) were lower (and mostly strongly significantly lower) than the statistics generated from edge null model 1. This is most likely due to the random reassignment of in-going links in this permutation causing (i) the assignment of in-going links to nodes which are originally disconnected in this context and thus increasing the mean and (ii) the randomisation of the in-going degree distribution increasing the betweenness variance.

Seven of the nine information-sharing contexts fell within the expected range of both the edge model permutations for node eccentricity (how far an actor is from the furthest other), the only exceptions is bycatch and fishing activity. I found that the observed variance in node eccentricity (Figure A3.1.5) was lower than expected for information sharing regarding turtle bycatch, in comparison to the null distributions (generated from the context

permutations), which had higher than expected observed variance in node eccentricity. The opposite was true for fishing activity. The observed mean node eccentricity (Figure A3.1.6) followed a similar pattern to the variance in node eccentricity, with information sharing regarding turtle bycatch is the only context that was lower than expected in comparison to the null distributions. Mean node eccentricity for information sharing regarding fishing activity illustrated the greatest contrast to the turtle bycatch context with higher than expected observed statistics. This supplementary analysis demonstrates that the turtle bycatch information-sharing context holds some structural dissimilarities in mean node eccentricity, not only when compared to the edge null models (Figure 5.3), but also given the underlying social structure of who is connected to who within the network.

A3.1.3. Cross-contextual correlations of dyadic links

Along with focussing on the ability of each context to predict to turtle bycatch information-sharing links, I also considered the correlation between all contexts and how these differed from the correlations expected under the context permutation null models (Figure A3.1.9). I found that the dyadic directed links within the ‘technology’ information-sharing network was more correlated with all the other contexts than expected under the general social structure of the network. This suggests that the technology information-sharing network was particularly predictive of fishing activity in general.

As expected, when comparing the correlations to those generated from edge-permutations (rather than context permutations), the observed statistics were vastly different even though these permutations were controlling for the number of nominations, degree distributions etc. due to randomising the underlying dyadic social structure (in terms of who can nominate who) (Figure A3.1.10).

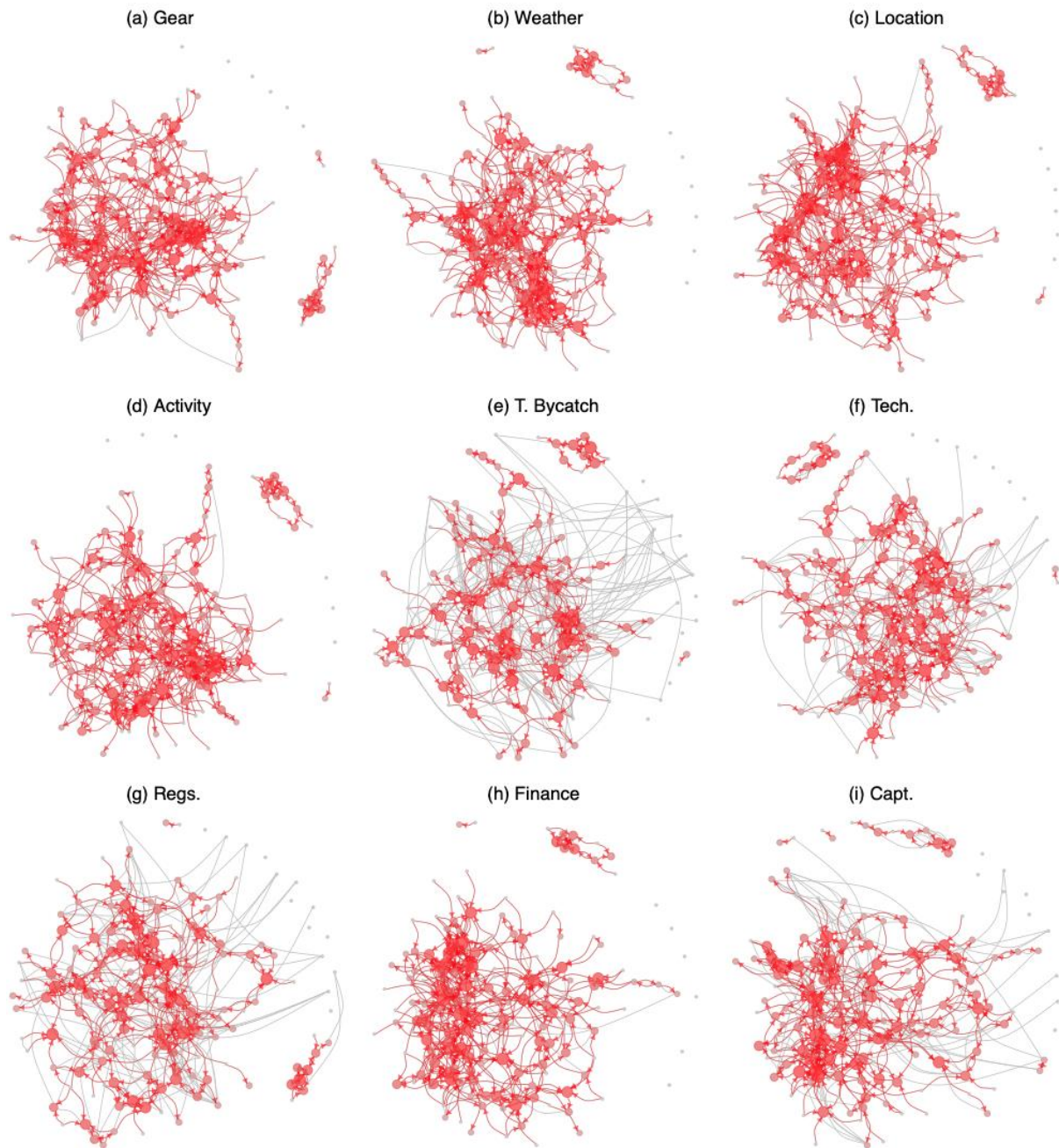


Figure A3.1.1. Illustrative network of the structure of information sharing across contexts. The nodes show each of the skippers and the adjoining lines show which dyads shared information in at least one context, and nominations within the focal context (as indicated by heading) is highlighted as a directed red arrow here (arrow points to the one that was nominated). Node size and shading shows the number of nominations each individual received for the focal context (largest and most red = most nominations, small and grey = no nominations). Layout was set as a spring layout of edges within each focal context (to minimise overlap).

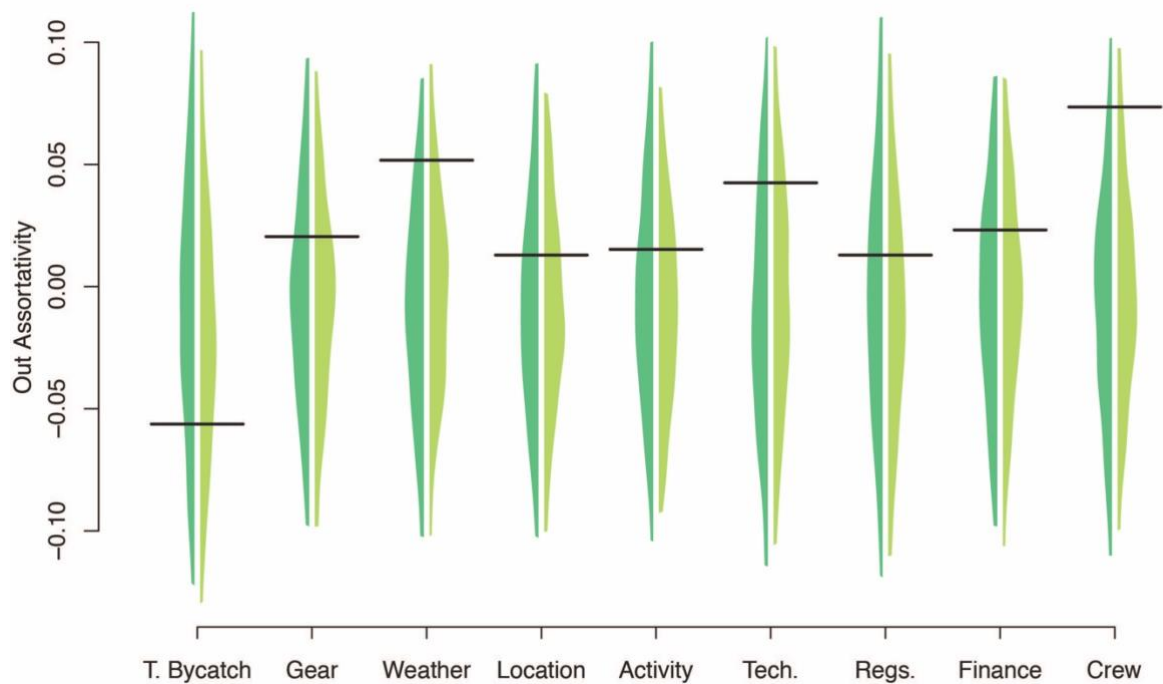


Figure A3.1.2. The observed assortativity coefficient for outgoing links in comparison to the null distributions for the different information-sharing networks. Horizontal lines show the observed values from the actual networks (red = observed values are above the permutations, black = observed values are within the range of the permutations, purple = observed values are below the permutations). Polygon distributions show those generated by permutations (dark green = outgoing edge permutation that maintains the no. of nominations each individual makes, light green = edge swap that maintains the no. of nominations each individual makes and also the number of times each individual was nominated). Outgoing links also show the same pattern seen in figure 1 (i.e., the turtle bycatch network is the only information network measured which is not assorted) but with no significant difference. For details on information contexts refer to Table 1 in the main text.

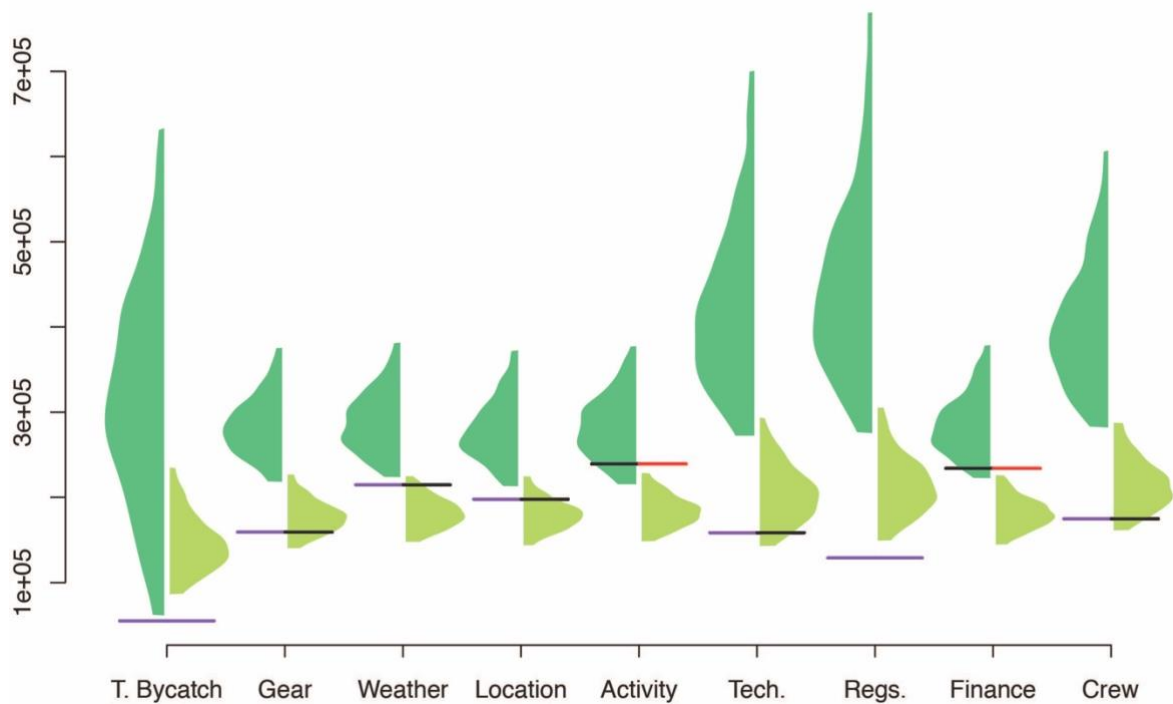


Figure A3.1.3. The observed variance in node betweenness in comparison to the null distributions for the different information-sharing networks. Horizontal lines show the observed values from the actual networks (red = observed values are above the permutations, black = observed values are within the range of the permutations, purple = observed values are below the permutations). Polygon distributions show those generated by permutations (dark green = outgoing edge permutation that maintains the no. of nominations each individual makes, light green = edge swap that maintains the no. of nominations each individual makes and also the number of times each individual was nominated). Here a similar pattern to the assortativity coefficient is also seen. For details on information contexts refer to Table 1 in the main text.

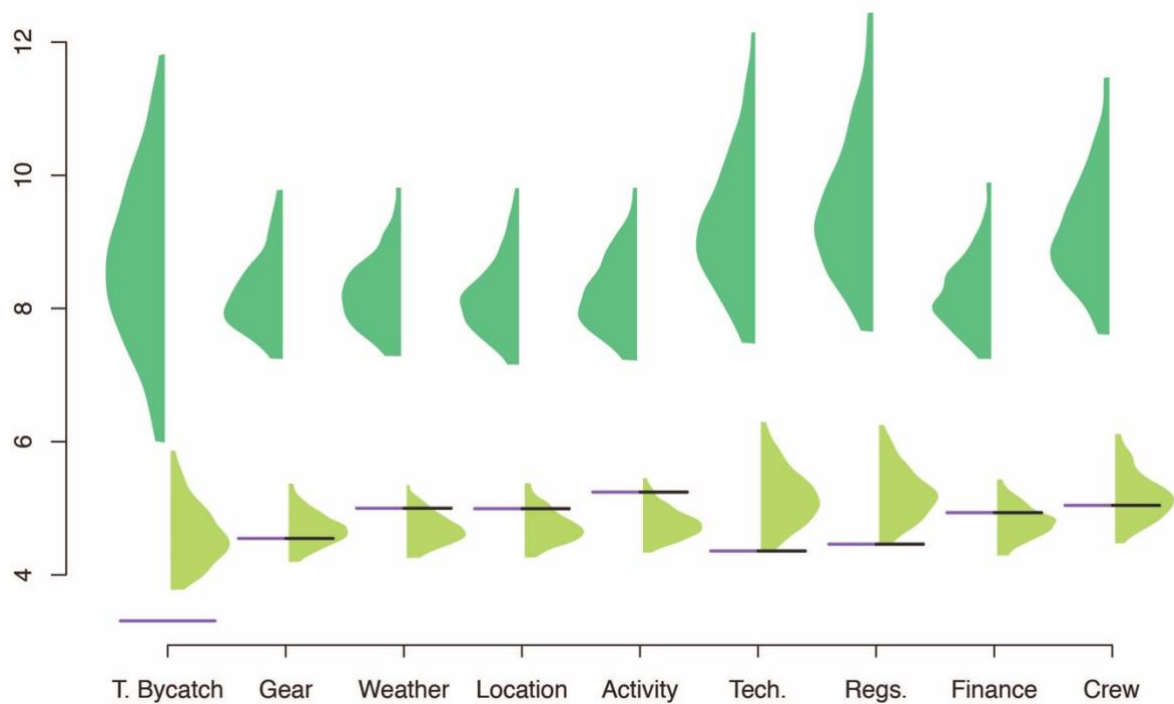


Figure A3.1.4. The observed mean node eccentricity in comparison to the null distributions for the different information-sharing networks. Horizontal lines show the observed values from the actual networks (red = observed values are above the permutations, black = observed values are within the range of the permutations, purple = observed values are below the permutations). Polygon distributions show those generated by permutations (dark green = outgoing edge permutation that maintains the no. of nominations each individual makes, light green = edge swap that maintains the no. of nominations each individual makes and also the number of times each individual was nominated). Here a similar pattern to the assortativity coefficient is also seen. For details on information contexts refer to Table 1 in the main text.

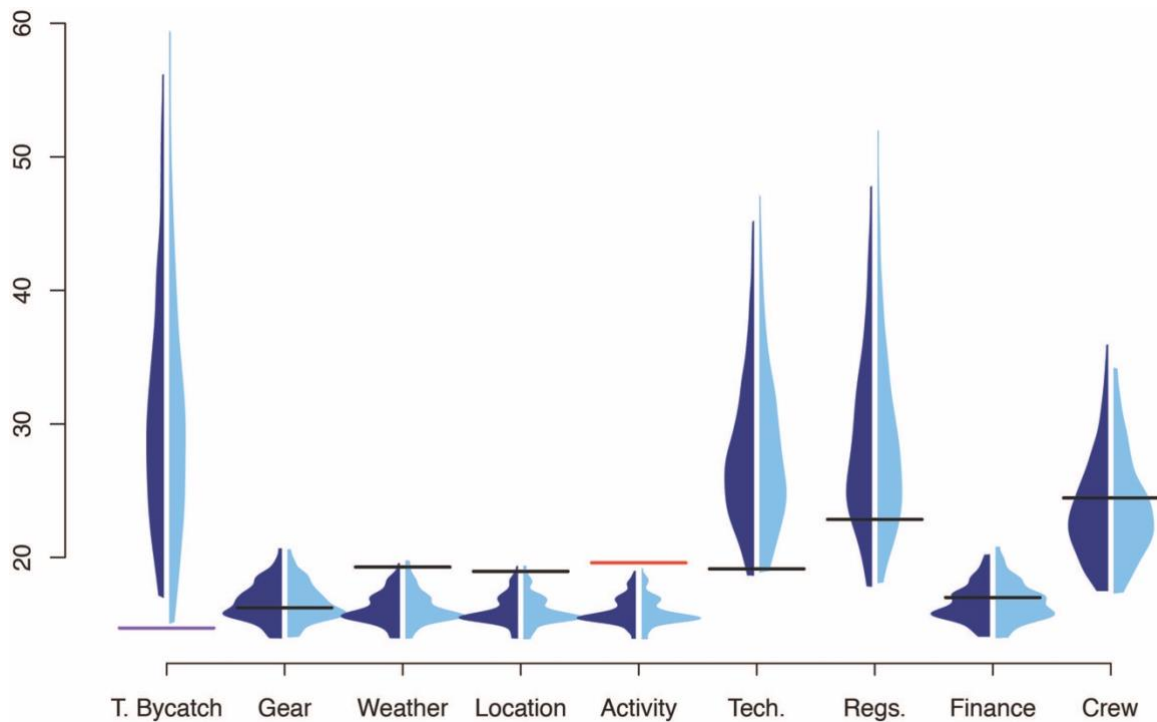


Figure A3.1.5. The observed variance in node eccentricity in comparison to the null distributions (generated from the context permutations) for the different information-sharing networks. Horizontal lines show the observed values from the actual networks (red = observed values are above the permutations, black = observed values are within the range of the permutations, purple = observed values are below the permutations). Polygon distributions show those generated by permutations (dark blue = context swap that maintains the no. of nominations each individual makes and also the number of times each individual was nominated, but swaps the context these were made within whilst maintain the number of times each context was nominated as overall, light blue = conservative context swap that is the same as dark blue, but also maintains the number of contexts each dyad nominated each other for – but changes those contexts (same as a gbi permutation but on the dyad-by-context edges). For details on information contexts refer to Table 1 in the main text.

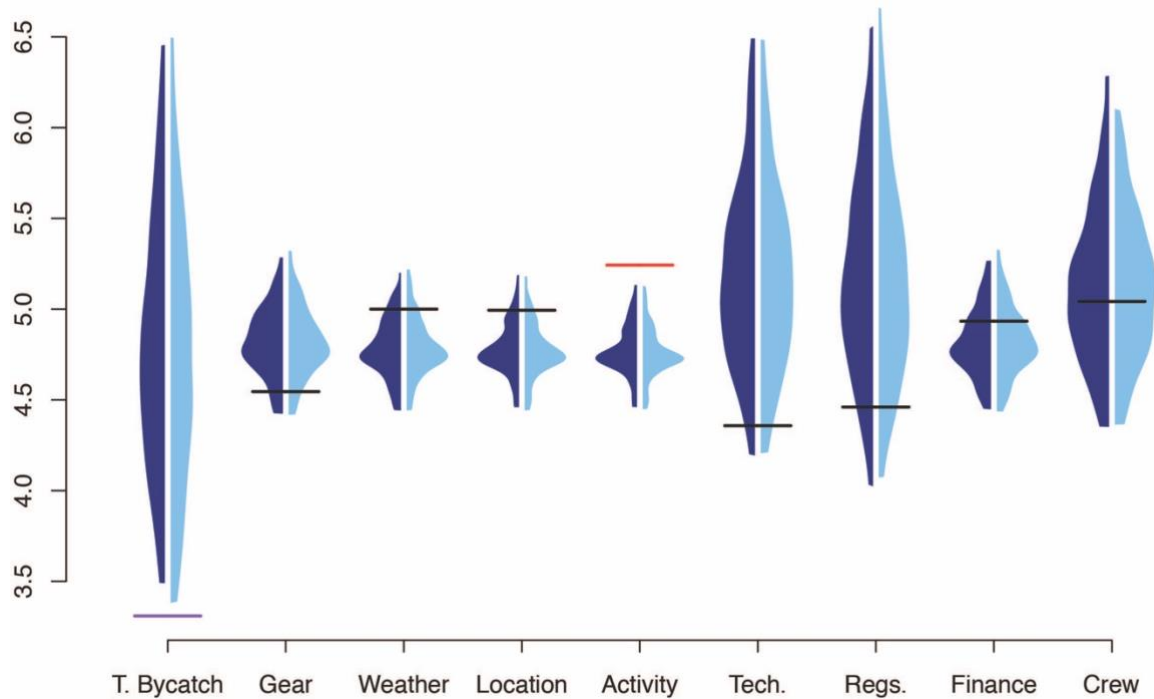


Figure A3.1.6. The observed mean node eccentricity in comparison to the null distributions (generated from the context permutations) for the different information-sharing networks. Horizontal lines show the observed values from the actual networks (red = observed values are above the permutations, black = observed values are within the range of the permutations, purple = observed values are below the permutations). Polygon distributions show those generated by permutations (dark blue = context swap that maintains the no. of nominations each individual makes and also the number of times each individual was nominated, but swaps the context these were made within whilst maintain the number of times each context was nominated as overall, light blue = conservative context swap that is the same as dark blue, but also maintains the number of contexts each dyad nominated each other for – but changes those contexts (same as a gbi permutation but on the dyad-by-context edges). For details on information contexts refer to Table 1 in the main text.

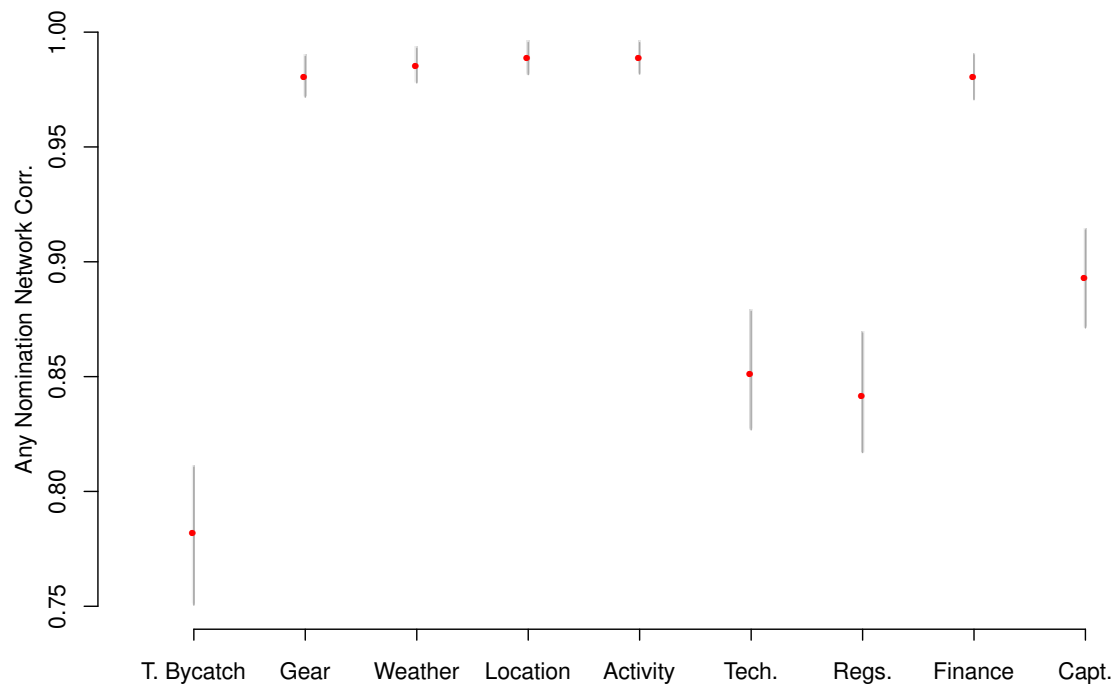


Figure A3.1.7. Context differences to the ‘any’ nomination network. Differences seen between different contexts in how predictive/correlated they are to the ‘any’ nomination network (lines show bootstrap). For details on information contexts see main text Methods – Experimental Design –Table 1.

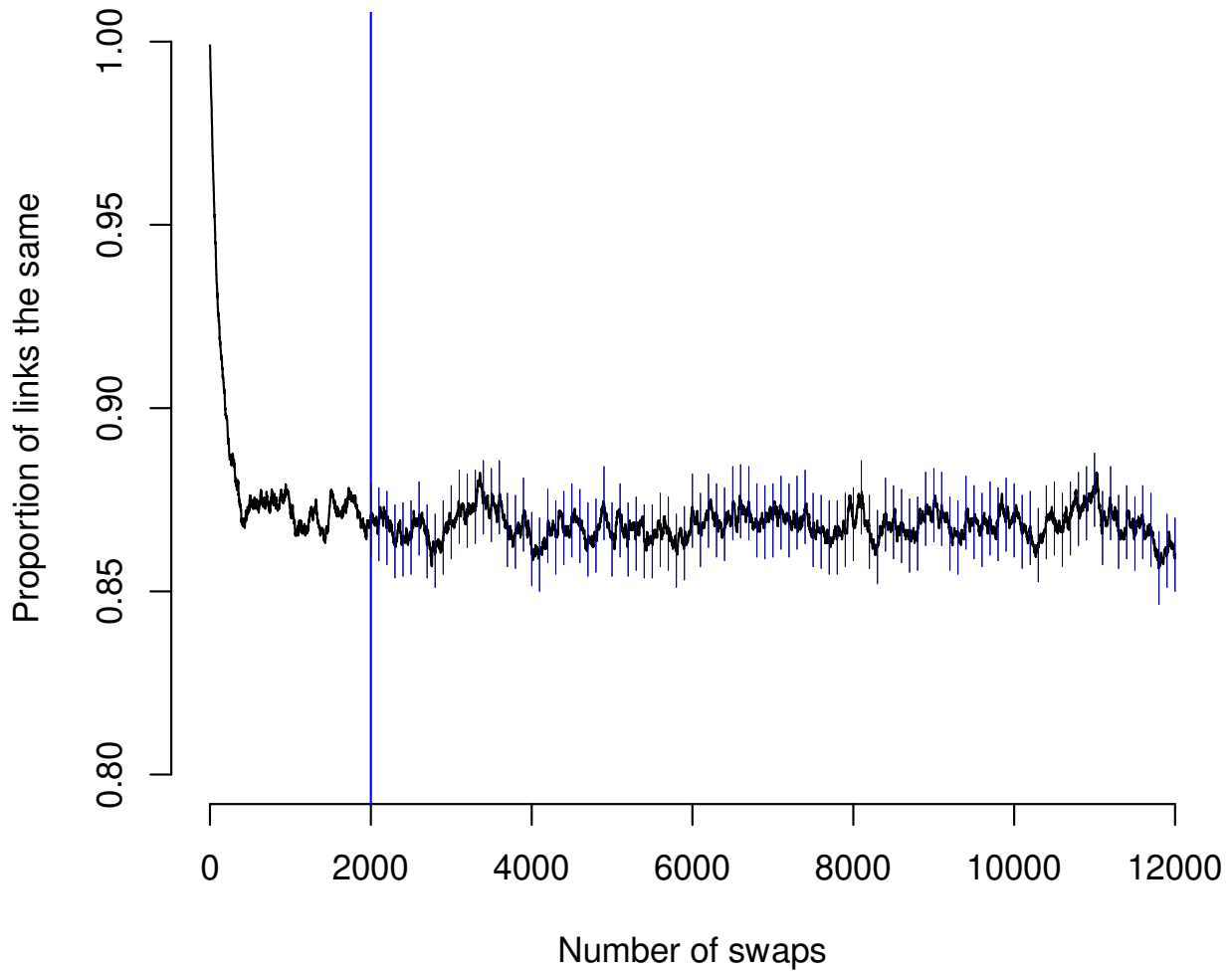


Figure A3.1.8. Output for evaluation of context null model 2. This permutation procedure required sequential swaps of the contexts in which nominations occurred between dyads (see main text Methods – Statistical Analysis – Section *Bii*) to generate the null networks. The y-axis illustrates the number of nominations between individual-to-individual dyads that are in the same context as those in the observed data, and the x-axis shows the number of swaps that took place during the permutation procedure. The long vertical blue line indicates the burn-in period for the randomisation swaps (2000 swaps before a null network was stored) and the short vertical blue lines show the points at which the following 999 null networks were stored (i.e., every 100 swaps).

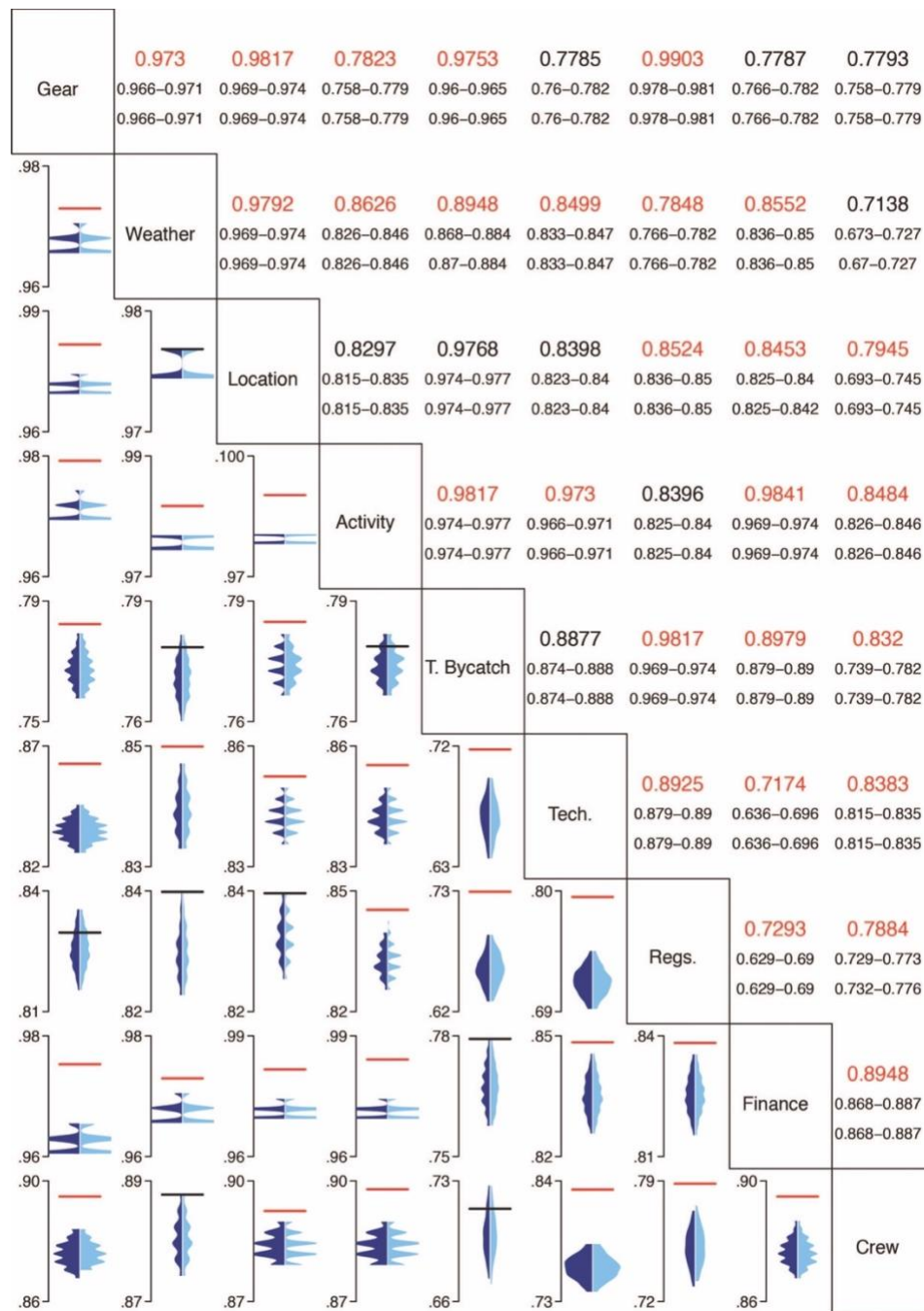


Figure A3.1.9. Observed correlation (and the correlation expected from the context permutations) between all of the information-sharing networks. Horizontal lines show the observed values from the actual networks (red = observed values are above the permutations, black = observed values are within the range of the permutations, purple = observed values are below the permutations). Polygon distributions show those generated by permutations (dark blue = context swap that maintains the no. of nominations each individual makes and also the number of times each individual was nominated, but swaps the context these were made within whilst maintain the number of times each context was nominated as overall, light blue = conservative context swap that is the same as dark blue, but also maintains the number of contexts each dyad nominated each other for – but changes those contexts (same as a gbi permutation but on the dyad-by-context edges). For details on information contexts refer to Table 1 in the main text.

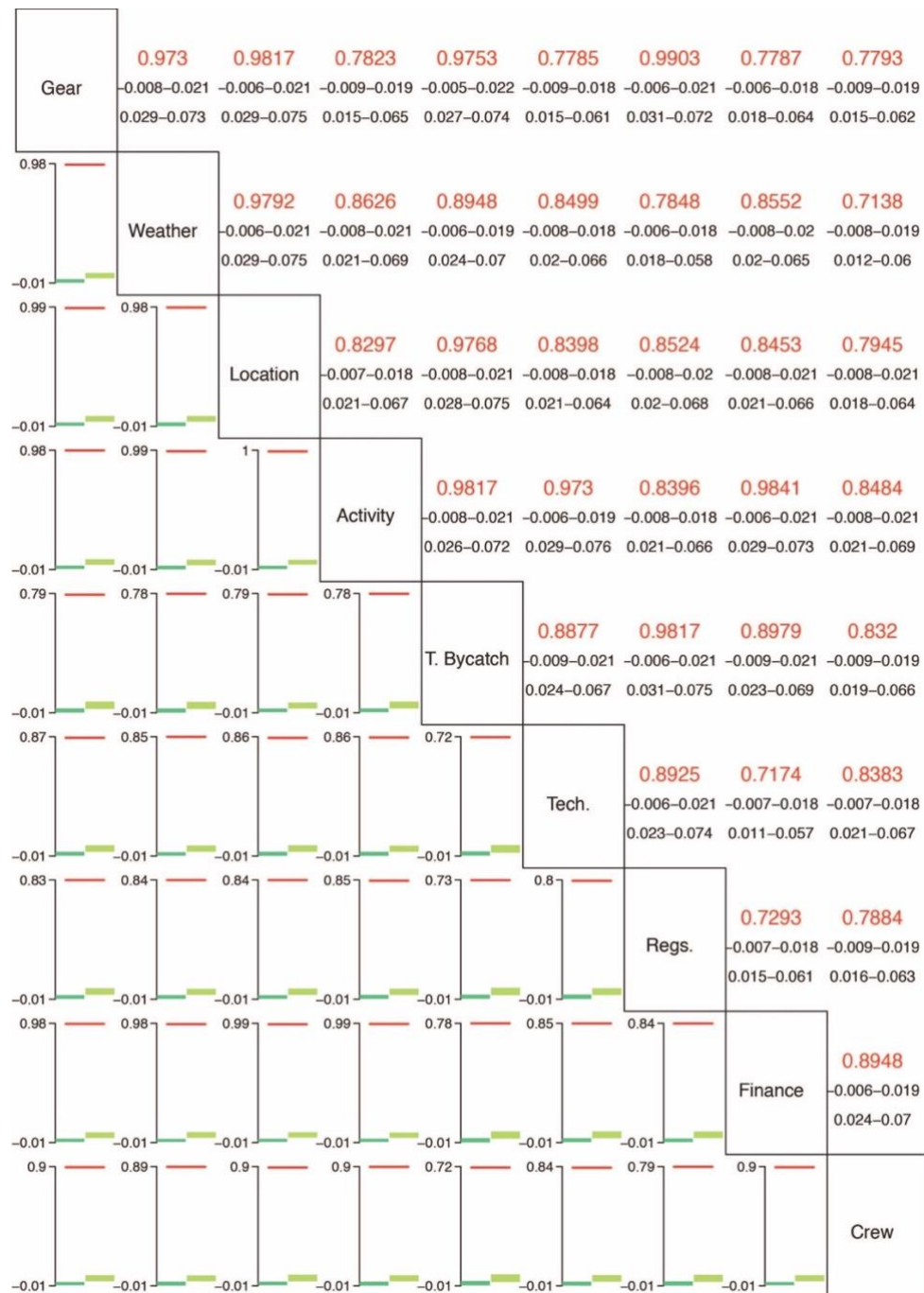


Figure A3.1.10. Observed correlation (and the correlation expected from the edge permutations) between all of the information-sharing networks. Horizontal lines show the observed values from the actual networks (red = observed values are above the permutations, black = observed values are within the range of the permutations, purple = observed values are below the permutations). Polygon distributions show those generated by permutations (dark green = outgoing edge permutation that maintains the no. of nominations each individual makes, light green = edge swap that maintains the no. of nominations each individual makes and also the number of times each individual was nominated). For details on information contexts refer to Table 1 in the main text.

Table A3.1.1. Measures of network structure with statistics describing in-assortment (in assort) and variance eccentricity (var eccent). Table includes the observed statistic and the statistic from the permutations as the mean, sd, 95% range from 2.5% (lq) to 97.5% (uq), and the p value (when compared to the observed stat).

stat	context	obs1	mean.sd.1	lq.uq.1	p1	mean.sd.2	lq.uq.2	p2
in assort	T.bycatch	0.0377	-0.0049 (0.0593)	-0.1228 to 0.1047	0.512	-0.0107 (0.0593)	-0.1316 to 0.1041	0.39
in assort	Gear	0.105	-0.0034 (0.0481)	-0.0921 to 0.0963	0.034	0.0052 (0.0486)	-0.088 to 0.1028	0.04
in assort	Weather	0.0932	-0.0089 (0.0494)	-0.1113 to 0.0882	0.044	0.0032 (0.0485)	-0.0859 to 0.0983	0.068
in assort	Location	0.1069	-0.005 (0.0471)	-0.095 to 0.0858	0.016	0.0057 (0.0467)	-0.0813 to 0.096	0.022
in assort	Activity	0.1038	-0.0048 (0.0452)	-0.0944 to 0.0822	0.022	0.004 (0.047)	-0.0885 to 0.1024	0.042
in assort	Tech	0.1143	-0.0091 (0.0547)	-0.1108 to 0.1041	0.032	0.0087 (0.0565)	-0.0982 to 0.1189	0.064
in assort	Regs	0.1246	-0.0051 (0.0566)	-0.113 to 0.1077	0.02	0.0038 (0.0507)	-0.0953 to 0.1038	0.026
in assort	Finance	0.1002	-0.0056 (0.0481)	-0.0995 to 0.0915	0.036	0.0049 (0.0473)	-0.0882 to 0.0976	0.048
in assort	Crew	0.1891	-0.0084 (0.0528)	-0.109 to 0.0956	0	0.0198 (0.0525)	-0.0821 to 0.1247	0
var eccent	T.bycatch	14.71	41 (13.5)	22.41 to 73.73	0.006	22.66 (5.335)	15.58 to 36.53	0.02
var eccent	Gear	16.24	8.819 (2.326)	5.209 to 14.11	0.016	12.84 (1.592)	10.67 to 16.65	0.066
var eccent	Weather	19.28	8.717 (2.206)	5.101 to 13.63	0.004	12.76 (1.563)	10.77 to 16.56	0.012
var eccent	Location	18.96	8.366 (2.17)	4.778 to 13.41	0.002	12.39 (1.397)	10.45 to 15.72	0.008
var eccent	Activity	19.6	8.595 (2.251)	5.068 to 13.99	0.004	12.48 (1.362)	10.52 to 15.63	0.002
var eccent	Tech	19.15	27.14 (7.831)	16.93 to 46.5	0.202	20.19 (3.709)	14.95 to 29.94	0.894
var eccent	Regs	22.85	30.48 (9.694)	18.21 to 57.39	0.392	19.87 (3.483)	15.04 to 28.13	0.342
var eccent	Finance	17	8.884 (2.249)	5.369 to 14.09	0.014	12.74 (1.405)	10.65 to 16.2	0.03
var eccent	Crew	24.46	19.43 (5.77)	11.87 to 34.63	0.264	16.87 (2.629)	13.3 to 23.2	0.022

Table A3.1.2. Measures of network structure with statistics describing assortativity coefficient for outgoing links (out assort), mean node eccentricity (mean eccent) and variance in node betweenness (var between). Table includes the observed statistic and the statistic from the permutations as the mean, sd, 95% range from 2.5% (lq) to 97.5% (uq), and the p value (when compared to the observed stat).

stat	context	obs1	mean.sd.1	lq.uq.1	p1	mean.sd.2	lq.uq.2	p2
out assort	turtle	-0.0563	-0.008 (0.0614)	-0.1223 to 0.1124	0.424	-0.0208 (0.0576)	-0.1297 to 0.0968	0.534
out assort	gear	0.0205	-0.004 (0.049)	-0.0981 to 0.0937	0.584	-0.0055 (0.0471)	-0.0983 to 0.0882	0.544
out assort	weather	0.0517	-0.0078 (0.0475)	-0.1025 to 0.0854	0.212	-0.0044 (0.0471)	-0.1021 to 0.0911	0.226
out assort	loc	0.0129	-0.0078 (0.05)	-0.1028 to 0.0914	0.65	-0.0091 (0.0465)	-0.1004 to 0.0794	0.626
out assort	activ	0.0152	-0.0039 (0.0506)	-0.1043 to 0.1003	0.662	-0.0082 (0.046)	-0.0926 to 0.0818	0.628
out assort	tech	0.0425	-0.0095 (0.055)	-0.1145 to 0.1023	0.36	-0.004 (0.0523)	-0.1058 to 0.0985	0.384
out assort	regs	0.0129	-0.0049 (0.0592)	-0.119 to 0.1104	0.766	-0.01 (0.0533)	-0.1103 to 0.0955	0.648
out assort	financ	0.0232	-0.0053 (0.0481)	-0.0982 to 0.0863	0.58	-0.0044 (0.0479)	-0.1062 to 0.0856	0.544
out assort	capt	0.0735	-0.0057 (0.0529)	-0.1101 to 0.1018	0.13	-0.0025 (0.0493)	-0.0998 to 0.0976	0.136
mean eccent	turtle	3.309	8.754 (1.498)	5.988 to 11.82	0	4.63 (0.5432)	3.776 to 5.867	0
mean eccent	gear	4.546	8.259 (0.6509)	7.242 to 9.782	0	4.701 (0.2994)	4.194 to 5.37	0.612
mean eccent	weather	5	8.28 (0.6363)	7.285 to 9.813	0	4.708 (0.2935)	4.254 to 5.352	0.26
mean eccent	loc	4.994	8.216 (0.674)	7.157 to 9.808	0	4.735 (0.2826)	4.261 to 5.376	0.336
mean eccent	activ	5.242	8.266 (0.6777)	7.218 to 9.813	0	4.792 (0.2752)	4.333 to 5.449	0.13
mean eccent	tech	4.358	9.346 (1.164)	7.472 to 12.15	0	5.188 (0.4979)	4.357 to 6.297	0.052
mean eccent	regs	4.461	9.576 (1.21)	7.652 to 12.44	0	5.233 (0.4729)	4.46 to 6.249	0.056
mean eccent	financ	4.933	8.259 (0.6506)	7.242 to 9.891	0	4.809 (0.2827)	4.291 to 5.431	0.572
mean eccent	capt	5.042	9.198 (0.9606)	7.606 to 11.47	0	5.18 (0.4175)	4.473 to 6.116	0.79
var between	turtle	55170	321700 (147600)	61560 to 633200	0.042	147500 (38430)	86450 to 234900	0
var between	gear	159300	285100 (39540)	218300 to 375500	0	178100 (21950)	140500 to 226800	0.376
var between	weather	214700	290400 (40420)	223900 to 381500	0.016	182700 (20020)	147900 to 224800	0.136
var between	loc	197800	280400 (42240)	213200 to 372600	0.01	180200 (19870)	143900 to 224500	0.344
var between	activ	239300	284800 (42540)	215400 to 377400	0.264	184700 (20600)	148500 to 228600	0.02
var between	tech	158600	428400 (108200)	272400 to 700500	0	206500 (40180)	143000 to 293500	0.18
var between	regs	129000	454000 (125800)	275500 to 768900	0	214900 (38420)	149300 to 305400	0.004
var between	financ	234000	286700 (40670)	222700 to 378600	0.136	182200 (20930)	144800 to 226000	0.034
var between	capt	174800	405800 (84840)	282100 to 606900	0	215600 (32690)	161400 to 287600	0.176

Table A3.1.3. Measures of cross-contextual comparisons with statistics describing variance in node eccentricity (var eccentric) and mean node eccentricity (mean eccentric). Table includes the observed statistic and the statistic from the permutations as the mean, sd, 95% range from 2.5% (lq) to 97.5% (uq), and the p value (when compared to the observed stat).

stat	context	obs1	mean.sd.1	lq.uq.1	p1	mean.sd.2	lq.uq.2	p2
var eccentric	turtle	14.71	31.63 (10.14)	16.98 to 56.19	0.02	30.91 (10.73)	15.05 to 59.44	0.038
var eccentric	gear	16.24	16.74 (1.665)	13.95 to 20.67	0.874	16.72 (1.661)	14.04 to 20.62	0.926
var eccentric	weather	19.28	16.26 (1.372)	13.96 to 19.57	0.058	16.33 (1.449)	13.94 to 19.76	0.088
var eccentric	loc	18.96	16.13 (1.282)	13.94 to 19.36	0.06	16.09 (1.308)	13.89 to 19.38	0.074
var eccentric	activ	19.6	16.07 (1.207)	13.92 to 18.98	0.034	15.98 (1.247)	13.9 to 19.2	0.042
var eccentric	tech	19.15	28.23 (6.785)	18.65 to 45.21	0.07	28.42 (7.11)	18.88 to 47.12	0.068
var eccentric	regs	22.85	29.16 (7.625)	17.81 to 47.81	0.338	29.18 (8.034)	18.08 to 51.99	0.382
var eccentric	financ	17	16.66 (1.606)	14.07 to 20.22	0.75	16.76 (1.651)	13.98 to 20.8	0.8
var eccentric	capt	24.46	24.1 (4.986)	17.49 to 35.94	0.774	24.17 (4.963)	17.3 to 34.21	0.786
mean eccentric	turtle	3.309	4.831 (0.7783)	3.491 to 6.455	0.022	4.771 (0.824)	3.381 to 6.498	0.04
mean eccentric	gear	4.546	4.824 (0.2212)	4.424 to 5.285	0.216	4.824 (0.221)	4.418 to 5.321	0.182
mean eccentric	weather	5	4.786 (0.1861)	4.442 to 5.2	0.238	4.796 (0.1884)	4.442 to 5.218	0.262
mean eccentric	loc	4.994	4.786 (0.1698)	4.46 to 5.188	0.202	4.779 (0.177)	4.442 to 5.182	0.208
mean eccentric	activ	5.242	4.776 (0.1616)	4.46 to 5.134	0.03	4.765 (0.1657)	4.448 to 5.127	0.03
mean eccentric	tech	4.358	5.207 (0.5926)	4.194 to 6.492	0.102	5.208 (0.5861)	4.206 to 6.485	0.102
mean eccentric	regs	4.461	5.194 (0.6411)	4.024 to 6.558	0.212	5.184 (0.6511)	4.073 to 6.661	0.228
mean eccentric	financ	4.933	4.817 (0.2121)	4.448 to 5.267	0.522	4.831 (0.216)	4.436 to 5.328	0.548
mean eccentric	capt	5.042	5.152 (0.4853)	4.351 to 6.285	0.904	5.159 (0.4691)	4.364 to 6.103	0.852

A3.2. Social Network Analysis structured questionnaire (English)

Section A: Individual socio-demographic information

First, I'm going to ask you a few questions about yourself. Note that your individual responses to this survey will remain confidential and we will only use the data collected in aggregate form.

Survey ID

Date

Full name

Nickname

Gender

☐ Male

☐ Female

Fisher / decision maker status: ☐ Skipper ☐ Vessel owner ☐ Skipper AND Owner

Plate number

Name of boat

Q1) What is your age? _____

Q2) Do you live in San José. Y _____ yrs., N, where do you live?
_____ region / city

Q3) If < 5 years, where did you live before and why did you move here?

Q4) What generation of gillnet fisherman in San Jose are you? _____

For boat owners that are not skippers:

Q5) Were you formally a gillnet captain?

☐ No

☐ Yes (please specify when you stopped fishing) _____

Q6) Which best describes your situation:

- ☐ My family fish with my boat as we divide the profits evenly. Or some other percentage_____
- ☐ I hire my boat to non-family members and receive a percentage of the catch profit: _____

For skippers and skippers AND boat owners

Q7) How many years have you been fishing?

Q8) Do you launch or land at any other ports?

- ☐ No
- ☐ Yes (please specify) _____

Q9) During which months did you not fish last year?

Q10) What is the principal net that you use? Trammel, Lineal, Other:

- ☐ Surface / driftnet
- ☐ Mid-water net
- ☐ Bottom net
- ☐ Other net type (please specify) _____

Q11) Do you ever switch net types from your main net type?

- ☐ No
- ☐ Yes (please explain to what, and under what circumstances)

Q12) What are you three main target species?

1. _____

2. _____

3. _____

For everyone:

Q13) Which of the following best describes you?

- ☐ President of a gremio / social group (which) _____
- ☐ Board member of a gremio / social group (which) _____
- ☐ Member of gremio / social group (which) _____
- ☐ I'm not a member of any gremio / social group (Individual owner operator)

Q14) What is your highest level of education?

- ☐ No formal education
- ☐ Primary school, please specify if completed _____
- ☐ Secondary school, please specify if completed _____
- ☐ Trade or technical certificate / fishing course, please specify if completed _____
- ☐ University degree, please specify if completed _____

[Personal income]

Q15) Is fishing your primary occupation/source of income?

- ☐ Yes
- ☐ No (please specify what is) _____

Q16) How much do you spend on fishing trips per month (on average)? Summer_____ Winter_____

Q17) How many days a month (in average) do you spend on fishing trip? Summer_____ Winter_____

Q18) What is your take-home monthly income (in soles) after all expenses in:

Summer: Max: _____

Winter: Max: _____

Average: _____

Average: _____

Min: _____

Min: _____

[Household income]

Q19) Which of the following household descriptions best fits you?

- ☐ Couple with children – with some children still living at home
- ☐ Couple with children – with all children having left home
- ☐ Couple without children
- ☐ Single with children
- ☐ Single without children

Q20) Are you the main wage earner in your household?

- ☐ No
- ☐ Yes

Q21) How many people are currently living in your household?

Q22) Of these, how many are fishermen? _____

Q23) Are there any other wage earners in your household that are not fishermen?

- ☐ No
- ☐ Yes (what jobs do they do?)

Q24) What percentage of your household income (including all wage earners) comes from fishing?

- | | | | | | | |
|---------------------------------|----------------------------------|----------------------------------|----------------------------------|-----------------------------------|---------------------------|--|
| ☐ 0-
20% | ☐ 21-
40% | ☐ 41-
60% | ☐ 61-
80% | ☐ 81-
100% | ☐ All | ☐ Don't know / rather not
say |
|---------------------------------|----------------------------------|----------------------------------|----------------------------------|-----------------------------------|---------------------------|--|

Section B: Social Network Analysis structured questionnaire

We need you to think about the people from San Jose that you share useful information about fisheries with; consider those you think may influence your fishing success. Remember that the shared information and names will remain anonymous and will not be revealed. This will help us understand how the information flows between fishermen.

Please consider relationships that you have had with other vessel owners, captains, owner/captains (owners who also captain their vessel), other fishery leaders, fishery management officials, members of the scientific or NGO community, boat launching / landing support, fish transport associations, fish sellers/market operators, your family and friends, and any other people you have fished with, or shared information with about fishing over the last 5 years.

Table A3.2.1. Please identify up to 10 individuals (providing first and last names, and known nicknames) that you *exchange useful information* with about fishing that you consider *valuable to your fishing success*.

Full name		Nickname	Rel	Crew	Meet	tMeet	Often	Topic of conversation									Value
								I	II	III	IV	V	VI	VII	VIII	IX	
1																	
2																	
3																	
4																	
5																	
6																	
7																	
8																	
9																	
10																	

Rel = Relation: A) Professional acquaintance, B) Friend, C) Family

Crew = Crew member: Y / N

Meet = How did you meet: A) family member, B) through a friend, C) through fishing, D) from a family member, E) Other: _____

tMeet = How long have you known this person: A) <1 yr, B) 1-5 yrs, C) >5 yrs

Often = How often do you share useful information about aspects of fishing with this person? A) 1-3 times/yr, B) 1-3 times/month, C) 1-3 times/week or more

I: Gear type (i.e. Changes, technology, maintenance)

II: Weather conditions

III: Fish location / catch sites

IV: Fishing activity (How many people fishing, who is fishing, who caught what, etc.)

V: Turtle bycatch

VI: Vessel technology / maintenance

VII: Fishery regulations (laws, rules)

VIII: Fishing finances (market prices, loans, fines, penalties)

IX: Hiring new crew / captain

Value: In general, how valuable do you feel the information that you exchange with this individual is to your fishing success? A) Very valuable, B) somewhat valuable, C) a little valuable.

To finish up with the network analysis, I have four more questions on bycatch and new gear uptake

Q26) Which of the people you've identified is the most influential to you when you are considering making changes to your fishing gear?

Q27) Which of the people you've identified is the most influential to you in (potentially) deciding about changing the way you fish (e.g. changing your behaviour such as shorter soak time)?

Q28) What do you think about taking on new technologies to reduce bycatch of turtles and dolphins? (-1 Negative, 0 Neutral, +1 Positive)

Q29) Are you aware of the work that the NGO ProDelphinus is undertaking with a few fishermen here in San Jose to help reduce the number of turtles and dolphins that are captured in nets? Do you know about the technologies that they are using?

Q30) Do you think the Orca underwater acoustic alarm used to deter dolphins attract sea lions to your nets?

- ☐ No
- ☐ Yes
- ☐ I don't know

Q31) Do you think lights on your nets to deter turtles attract sea lions to your nets?

- ☐ No
- ☐ Yes
- ☐ I don't know

If you have any comments on this survey or about information sharing between fishermen within the San José community, please tell us or write them below.

A large, empty rectangular box with a black border, intended for the respondent to write their comments or feedback.

Thank you very much for your time and help in this survey

Sección A: Información socio-demográfica individual

ID de la encuesta	Fecha
Nombre completo	Apodo

Número de matrícula	Nombre de la embarcación

Q3) Si < 5 años, ¿dónde vivías antes y por qué te mudaste aquí?

Q4) ¿Qué generación de pescador de redes de enmalle de San José eres tú?

Para dueños de embarcaciones que no son PATRONES.

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Q6) ¿Cuál describe mejor tu situación?:

☐ Mi familia pesca con mi bote, dividimos las ganancias igual. Otro porcentaje?

☐ Rento mi bote a un ajeno y recibo un porcentaje de la ganancia, cuanto?

Solo para PATRONES y PATRONES que son ARMADORES

Q7) ¿Cuántos años llevas pescando? _____

Q8) ¿Embarcas o desembarcas de otros puertos?

☐ No

☐ Sí (por favor especifica) _____

Q9) ¿En que meses descansaste el año pasado?

Q10) ¿Cuál es el tipo principal de red de enmalle que usas? Trasmallo, Lineal

Otro: _____

☐ Red de superficie / red de deriva

☐ Red de mediagua

☐ Red de fondo

☐ Otro tipo de red (por favor especifica) _____

Q11) ¿Cambias tu tipo de red principal por otros?

☐ No

☐ Sí (por favor especifica a qué, y debido a qué) _____

Q12) ¿Cuáles son tus 3 objetivos principales de pesca?

1. _____

2. _____

3. _____

Para todos

Q13) ¿Cuál de los siguientes te describe mejor?

- ☐ Presidente de un gremio / grupo social (cuál) _____
- ☐ Miembro de consejo de gremio / grupo social (cuál) _____
- ☐ Miembro de gremio / grupo social (cuál) _____
- ☐ No soy agremiado / no pertenezco a grupos sociales (Dueño operador individual)

Q14) ¿Cuál es tu nivel educativo?

- ☐ Sin educación formal
- ☐ Primaria (por favor especificar si completó) _____
- ☐ Secundaria (por favor especificar si completó) _____
- ☐ Técnico / capacitado en pesca (por favor especificar si completó) _____
- ☐ Universitario (por favor especificar si completó) _____

[Ingresos personales]

Q15) ¿Es la pesca tu principal ocupación / fuente de ingresos?

- ☐ Sí
- ☐ No (por favor especifica cuál es) _____

Q16) Cuánto es el gasto promedio mensual en viajes en: Verano _____,
Invierno _____

Q17) Cuántos días (promedio) te embarcas al mes en: Verano _____,
Invierno _____ .

Q18) ¿Cuál es el ingreso mensual promedio (después de costos) que obtienes en:

Verano: Bueno: _____	Invierno: Bueno: _____
Medio: _____	Medio: _____
Bajo: _____	Bajo: _____

[Ingresos familiares]

Q19) ¿Cuál de las siguientes descripciones familiares se aplica a ti?

- ☐ Pareja con hijos – con algunos de los hijos viviendo en el hogar
- ☐ Pareja con hijos – con todos los hijos fuera del hogar
- ☐ Pareja sin hijos
- ☐ Soltero sin hijos
- ☐ Soltero con hijos

Q20) ¿Eres el sustento económico principal de tu hogar?

- ☐ No
- ☐ Sí

Q21) ¿Cuántas personas viven actualmente en tu hogar? _____

Q22) De ellos, ¿cuántos son pescadores? _____

Q23) ¿Existen otros proveedores de sustento económico en tu hogar que no sean pescadores?

- ☐ No
- ☐ Sí (¿qué trabajos realizan?)

Q24) ¿Qué porcentaje del ingreso de tu hogar (incluyendo a todos los que proven) proviene de la pesca?

- ☐ 0-20%
- ☐ 21-40%
- ☐ 41-60%
- ☐ 61-80%
- ☐ 81-100%
- ☐ Todos
- ☐ No se / Preferiría no decirlo

Sección B: Cuestionario estructurado de Análisis de Red Social

Piensa con quienes intercambias INFORMACION UTIL de pesca en San Jose y que sientes que PODRIA INFLUENCIAR en que te vaya bien en la pesca. Los nombres y la informacion que des se mantendran en anonimato y no sera revelada. Esto servira para saber como fluye la informacion entre pescadores.

Recuerda a: otros dueños de embarcaciones, capitanes, otros líderes pesqueros, oficiales de manejo pesquero, científicos o ONGs, embarcadores/ayudantes de embarque y desembarque, asociaciones de chalaneros, vendedores de pescado/operadores de mercado, tu familia y amigos, y todas las otras personas con las que hayas pescado o compartido información de pesca en los últimos 5 años.

Table A3.3.1. Social Network Analysis questionnaire (Spanish). Por favor identifica hasta 10 individuos (nombres y apellidos, no solo apodos) con los que *intercambias información útil* acerca de la pesca que consideres *valioso para tu éxito pesquero*.

Nombre completo		Apodo	Rel	Crew	Meet	tMeet	Often	Tema de conversación									Valor
								I	II	III	IV	V	VI	VII	VIII	IX	
1																	
2																	
3																	
4																	
5																	
6																	
7																	
8																	
9																	
10																	

Rel = Relacion: A) Profesional conocido, B) Amigo, C) Familiar

Crew = Colega-tripulante, Y / N

Meet = Como lo conociste: A) familiar, B) por un amigo, C) a traves de la pesca, D) por un familiar, E) OTRO: _____

tMeet = Cuanto tiempo lo conoces: A) <1 año, B) 1-5 años, C) >5 años

Often = Que tan seguido comparten info: A) 1-3 veces/año, B) 1-3 veces/mes, C) 1-3 veces/semana o más

I: tipo de arte (i.e. cambios, tecnologia, mantenimiento)

II: condiciones climaticas

III: ubicacion de los peces y sitios de captura

IV: actividad pesquera (cuanto, quienes estan pescando, que estan pescando, quien cogio que, etc.)

V: Captura incidental de tortuga

VI: tecnologia y mantenimiento de la nave

VII: regulaciones pesqueras (leyes, reglas)

VIII: finanza pesquera (precios del Mercado, prestamos, multas, penalidades)

IX: Contratacion de tripulantes o capitan

Value: Que tan valiosa es la informacion que intercambias: A) muy valiosa, B) algo valiosa, C) un poco valiosa

Solo para terminar el análisis de red social, tengo cuatro preguntas más acerca de pesca incidental y aceptación de nuevos artes de pesca.

Q26) ¿Cuál de las personas que has identificado es la más influyente para ti cuando se trata de hacer cambios en los artes de pesca?

Q27) ¿Cuál de las personas que has identificado es la más influyente para ti en (potencialmente) decidir cambiar la forma en la que pescas (e.g. cambiar el momento y duración que pones la red)?

Q28) ¿Qué opinas de adoptar nuevas tecnologías para reducir la captura incidental de tortugas y delfines? (-1 , 0 , +1)

Q29) ¿Estás al tanto del trabajo que la ONG ProDelphinus viene llevando a cabo con un pequeño grupo de pescadores aquí en San José para ayudar a reducir el número de tortugas y delfines que son capturados en las redes? Conoces las tecnologías que usan?

Si tienes comentarios acerca de esta encuesta por favor dinos o escríbelos en el cuadro.

Muchas gracias por tu tiempo y colaboración con esta encuesta