Reducing fishing impacts on species of conservation concern at multiple scales

Leslie Amlwch Roberson

Master of Science, Marine Science

Bachelor of Arts, Environmental Studies

ORCID: 0000-0002-7490-0795

A thesis submitted for the degree of Doctor of Philosophy at The University of Queensland in 2021

School of Earth and Environmental Sciences

Abstract

Numerous human activities directly and indirectly threaten marine biodiversity and ecosystems, but fishing is a primary threat directly driving the decline of many marine species. Advancing fishing technologies have enabled humans to exploit nearly every corner of the ocean and expand into increasingly deep, remote, and previously unexplored areas. Fishing disrupts the integrity of marine ecosystems in many different ways, including damage to benthic habitats by fishing gear, alteration of fish community structure, changes to species' behavior, selection for less genetically advantageous traits, and disruption of trophic webs. But perhaps the most obvious impact of fishing is simply that it removes vast amounts of biodiversity from the ocean, whether species are targeted or caught incidentally. Protecting species from fishing impacts is a monumental task. To prevent marine biodiversity loss and ensure the future viability of marine ecosystems and the billions of people that rely on them, marine conservation efforts must work in tandem with dedicated fisheries management.

Policies for mitigating fishing impacts exist across multiple scales. The legal foundation for fisheries management both on the High Seas and within national waters stems from the UN Convention on the Law of the Sea and its various implementing agreements, such as UN Fish Stocks Agreement. At regional scales, many countries join fisheries management organizations, which mandate monitoring and management. The onus to meet these requirements falls on federal or state management bodies within each country. They interpret the mandates and, in turn, enforce specific rules—such as limiting how, when, and where fishing can occur for each "fishery," which is defined by some combination of a geographical area, fishing method, and target species. In addition to fisheries legislation, more general conservation legislation can also force changes in fishing practices. For example, the Convention on Biological Diversity's Aichi Target 11, which aims to protect at least 10% of the ocean through marine protected areas and other effective area-based conservation measures, has resulted in no-take areas as well as other restrictions on fishing effort, such as prohibited gear types and regulations on catch and trade of particular species.

Despite increased efforts to protect marine biodiversity and manage fishing, serious issues and gaps exist across all levels of fisheries management. One-third of all assessed commercial fish stocks globally are considered to be overexploited, and this represents only a small portion of global fishing

effort and the species impacted by fisheries. There are some successful efforts to reduce fishing impacts on threatened fish species and charismatic megafauna in certain contexts, but overall, fishing remains a key driver of decline for many targeted and incidentally caught elasmobranchs, sea turtles, seabirds, and marine mammals. Lack of adequate enforcement of fishing and seafood trade regulations is a pervasive problem. A more insidious issue is the many layers of disconnect between management frameworks and the reality of how fishing activities are carried out. One common example of management mismatches is the limited list of species that are actively managed, compared to how many species are caught. Another example is where the scale of regulation overlooks the importance of particular gear types, geographic areas, or fishing vessels within a fishing sector with regards to its cumulative impact on threatened species.

The perverse impacts of fisheries on marine species is a vast topic, and there are numerous research gaps that, if addressed, would help deliver effective fisheries management and conservation solutions. Through this thesis, in eight chapters, I explore and help address gaps in our understanding of how to manage overfishing impacts on biodiversity at different geographic and regulatory scales. First, I map the political distributions of marine biodiversity, including many fished species, and find that marine biodiversity is far more transboundary than terrestrial biodiversity, with the vast majority (over 90%) of species' distributions spanning an international border and over 50% of species occurring in more than ten jurisdictions (**Chapter 2**, Roberson et al. [*in review*]). Second, I provide a baseline assessment of the conservation status of widely exploited seafood species and find that 92 threatened fish and invertebrate species are reported in global catch records, with many wealthy nations driving both catch and international trade of threatened seafood (**Chapter 3**, Roberson et al. 2020). Third, in **Chapters 4-6**, I focus on tuna fisheries in an important and understudied region, the Indian Ocean. I show how an outdated categorization of fishing sectors allows the industrial-scale gillnet fisheries to operate essentially without monitoring or regulation (**Chapter 4**, Roberson et al. 2019). I present a case study of cetaceans' susceptibility to capture in tuna gillnet fisheries, and demonstrate a method that provides more mathematically robust estimates of risk using expert judgment in data-poor contexts (**Chapter 5**, Roberson, Hobday and Wilcox [*in prep*]). I then use this new method to provide the first spatially-explicit risk assessment of catch susceptibility of cetaceans, sea turtles, and elasmobranchs in Indian Ocean tuna fisheries, and find that—as anecdotes and reports suggest—gillnets likely pose a serious threat to many threatened megafauna species, and all three gear types likely interact with a much wider range of species than available records show (**Chapter 6**, Roberson et al. [*in prep*]). Finally, I explore fishing impacts at the level of individual vessels, and show that there are significant variations in threatened species bycatch among skippers within five Commonwealth fisheries, which suggests that an alternative framing of management questions could improve the environmental performance of fisheries (**Chapter 7**, Roberson and Wilcox [*in prep*]).

Context-appropriate innovations in fisheries management are instrumental in reducing overfishing impacts on marine biodiversity. Considerable barriers remain to actually implementing effective management solutions, but this work provides baseline information and tools for management in different contexts. If we are serious about protecting the ocean and our fisheries, we need a portfolio of management actions at many different scales, from high-level national and international policies all the way down to changes in the behavior of the fishers themselves.

Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

I have clearly stated the contribution of others to my thesis as a whole, including statistical assistance, survey design, data analysis, significant technical procedures, professional editorial advice, financial support and any other original research work used or reported in my thesis. The content of my thesis is the result of work I have carried out since the commencement of my higher degree by research candidature and does not include a substantial part of work that has been submitted to qualify for the award of any other degree or diploma in any university or other tertiary institution. I have clearly stated which parts of my thesis, if any, have been submitted to qualify for another award.

I acknowledge that an electronic copy of my thesis must be lodged with the University Library and, subject to the policy and procedures of The University of Queensland, the thesis be made available for research and study in accordance with the Copyright Act 1968 unless a period of embargo has been approved by the Dean of the Graduate School.

I acknowledge that copyright of all material contained in my thesis resides with the copyright holder(s) of that material. Where appropriate I have obtained copyright permission from the copyright holder to reproduce material in this thesis and have sought permission from co-authors for any jointly authored works included in the thesis.

Publications during candidature

Peer-reviewed papers

Roberson, L.A., Watson, R.A. and Klein, C.J. (2020) Over 90 endangered fish and invertebrates are caught in industrial fisheries. *Nature Communications* 11, 4764.

Roberson, L.A., Kiszka, J.J. and Watson, J.E.M. (2019) Need to address gaps in global fisheries observation. *Conservation Biology* 0, cobi.13265.

Tulloch, A.I.T., Auerbach, N., Avery-Gomm, S., Bayraktarov, E., Butt, N., Dickman, C.R., Ehmke, G., Fisher, D.O., Grantham, H., Holden, M.H., Lavery, T.H., Leseberg, N.P., Nicholls, M., O'Connor, J., **Roberson, L.A.,** Smyth, A.K., Stone, Z., Tulloch, V., Turak, E., Wardle, G.M., Watson, J,E.M. (2019) Reply to 'Consider species specialism when publishing datasets' and 'Decision trees for data publishing may exacerbate conservation conflict.' Nature Ecology and Evolution 3, 320–321.

Tulloch, A.I.T., Auerbach, N., Avery-Gomm, S., Bayraktarov, E., Butt, N., Dickman, C.R., Ehmke, G., Fisher, D.O., Grantham, H., Holden, M.H., Lavery, T.H., Leseberg, N.P., Nicholls, M., O'Connor, J., **Roberson, L.A.,** Smyth, A.K., Stone, Z., Tulloch, V., Turak, E., Wardle, G.M., Watson, J,E.M. (2018) A decision tree for assessing the risks and benefits of publishing biodiversity data. *Nature Ecology & Evolution* **2**, 1209–1217

Other publications

Davey, M. and **Roberson, L.A**. (2020) Submission to the Independent Review of the Environment Protection and Biodiversity Conservation Act 1999. Canberra, Australia. DOI: 10.13140/RG.2.2.13380.24960

Publications and manuscripts included in this thesis

I have incorporated three publications and three manuscripts into my thesis as per UQ policy (PPL 4.60.07 Alternative Thesis Format Options). This section provides details for each publication, including where it appears in the thesis and others' contributions to the authorship as per the requirements in section 5.1 of the UQ Authorship Policy (PPL 4.20.04 Authorship).

Publications

Incorporated as Chapter 3:

Roberson, L.A., Watson, R.A. and Klein, C.J. (2020) Over 90 endangered fish and invertebrates are caught in industrial fisheries. *Nature Communications* 11, 4764.

Contributor	Statement of contribution	%
Leslie A. Roberson	drafting and production analysis and interpretation preparation of figures conception and design	65 95 100 60
Reg A. Watson	analysis and interpretation drafting and production supervision, guidance	5 5 10
Carissa J. Klein	Drafting and production supervision, guidance Conception and design	30 90 40

Incorporated as Chapter 4:

Roberson, L.A., Kiszka, J.J. and Watson, J.E.M. (2019) Need to address gaps in global fisheries observation. *Conservation Biology* 0, cobi.13265.

Contributor	Statement of contribution	%
Leslie A. Roberson	writing of text initial concept	80 90
James E.M. Watson	writing of text supervision, guidance initial concept	10 50 5

Jeremy J. Kiszka	writing of text supervision, guidance initial concept	10 50 5

Manuscripts

Incorporated as Chapter 2:

Roberson, L.A., O'Hara, C., Watson, J.E., Halpern, B., Klein, C.J., Dunn, D., Frazier, M., Beyer, H., Keumpel, C., Williams, B., Grantham, H., Montgomery, J., Kark, S., and Runting, R. (*In Review)* Multinational Coordination Required for Conservation of Over 90% of Marine Species. *Global Change Biology*.

Contributor	Statement of contribution	%
Leslie A. Roberson	writing of text theoretical derivations numerical calculations preparation of figures initial concept	75 20 80 90 10
Casey O'Hara	theoretical derivations drafting and editing numerical calculations	5 5 10
James EM Watson	writing of text drafting and editing supervision, guidance theoretical derivations initial concept	5 15 25 20 25
Benjamin S Halpern	writing of text drafting and editing supervision, guidance theoretical derivations initial concept	5 5 10 10 10
Carissa Klein	drafting and editing supervision, guidance theoretical derivations initial concept	20 20 5 5
Daniel Dunn	writing of text drafting and editing	5 10

	supervision, guidance theoretical derivations	5 10
Hawthorne Beyer	drafting and editing supervision, guidance theoretical derivations initial concept	10 5 10 5
Melanie Frazier	preparation of figures numerical calculations initial concept preparation of figures	5 10 5 5
Caitlin Keumpel	writing of text drafting and editing initial concept	5 5 5
Brooke Williams	initial concept drafting and editing preparation of figures	5 5 5
Hedley Grantham	initial concept drafting and editing	5 5
Jamie Montgomery	initial concept drafting and editing	5 5
Salit Kark	initial concept drafting and editing	5 5
Rebecca Runting	writing of text drafting and editing supervision, guidance theoretical derivations initial concept	5 15 30 20 15

Contributions by others to the thesis

Chapters 2-7 are based on manuscripts that are published or prepared for submission to peer review journals in collaboration with other authors. **Chapter 2** is under review in *Global Change Biology*, **Chapter 3** is published in *Nature Communications*, **Chapter 4** is published in *Conservation Biology,* Chapter 5 is in preparation for submission to *Fish and Fisheries* as a "Ghoti" forum article, **Chapter 6** is in preparation for submission to *Fish and Fisheries,* and **Chapter 7** is in preparation for submission to *Nature Sustainability.* For these chapters, I have retained the text consistent with the journal's format, including the order of the sections (e.g., Methods at the end of the article for some journals and different section heading layout for the Ghoti forum article). I also maintain the use of the plural first-person pronoun "we," whereas I use the singular pronoun "I" for the Introduction (**Chapter 1**) and Conclusion (**Chapter 8**).

Chapter 1 (Introduction): This chapter was written by the Candidate, with editorial input from Carissa Klein, James Watson, and Salit Kark. The schematic illustration was conceived by the Candidate and drawn by Dan Vallentyne.

Chapter 2: James Watson conceived the original idea for the manuscript based off earlier work with Rebecca Runting. All authors attended a workshop where they contributed to the initial concept, and the Candidate further developed the idea to its current state. The Candidate, Casey O'Hara, and Melanie Frazier collated the species distributions data. The Candidate constructed and performed the analysis with input from Casey O'Hara. Rebecca Runting provided input on the GIS components of the analysis. The Candidate interpreted the results and wrote the manuscript. All authors provided editorial input on the manuscript.

Chapter 3: The Candidate and Carissa Klein conceived the idea for the manuscript. Reg Watson built the model of global seafood trade (Watson *et al.* 2016). The Candidate constructed and performed the analysis. The Candidate wrote the manuscript with editorial input from Carissa Klein.

Chapter 4: The Candidate conceived the original idea and James Watson helped develop the concept of the paper. The Candidate wrote the manuscript with editorial input from James Watson and Jeremy Kiszka.

Chapter 5: Chris Wilcox and the Candidate conceived the project, and Alistair Hobday contributed to the development of the idea. The Candidate performed the analysis and wrote the manuscript, with editorial input from Chris Wilcox and Alistair Hobday.

Chapter 6: The Candidate conceived the project with guidance from Jeremy Kiszka. The Candidate constructed the analysis with input from Chris Wilcox. Yannick Rousseau created the model of fishing effort (Rousseau *et al.* 2019) and Kristin Kaschner built the AquaMaps model of species distributions (with others) (Kaschner *et al.* 2016). The Candidate designed the species database and populated it with help from Emma Dugan, Kristofer Gonzalez, and Madeline Green. The Candidate processed the data and Germain Boussarie spatially smoothed the fishing effort. The Candidate performed the remainder of the analysis and wrote the manuscript.

Chapter 7: Chris Wilcox conceived the original idea for the project and the Candidate further developed the concept to its current state. The Candidate performed the analysis and interpreted the results with input from Chris Wilcox. The Candidate wrote the manuscript with editorial contributions from Chris Wilcox.

Chapter 8 (Conclusion): This chapter was written by the Candidate, with editorial input from Carissa Klein, James Watson, and Salit Kark.

Appendix 1: This appendix provides supplementary information for Chapter 2. All additional materials and analyses were prepared by the Candidate.

Appendix 2: This appendix provides supplementary information for Chapter 3. All additional materials and analyses were prepared by the Candidate.

Appendix 3: This appendix provides supplementary information for Chapter 5. All additional materials and analyses were prepared by the Candidate.

Appendix 4: This appendix was written by the Candidate and provides supplementary information for Chapter 6. All additional materials and analyses were prepared by the Candidate.

Appendix 5: This appendix provides supplementary information for Chapter 7. All additional materials and analyses were prepared by the Candidate.

Statement of parts of the thesis submitted to qualify for the award of another degree

No works submitted towards another degree have been included in this thesis.

Research Involving Human or Animal Subjects

No animal or human subjects were involved in this research.

Acknowledgements

This is a long list of acknowledgements, and it's far from complete. It turns out that moving to a country where you don't know a single person and completing a PhD (and really enjoying that PhD) definitely cannot be done alone.

First, to my supervisors: Carissa, Salit, James, and Chris. I was advised that having four supervisors is a bad idea—and maybe it is—but in my case it turned out really well and I am extremely grateful for your mentorship. You all brought a different suite of skills and expertise to my PhD and I learned so many different things from you these past four years. You all truly care about your students, as scientists and as people, and it shows. Thank you for being there for me one thousand percent when COVID and health crises hit my family at home. This past year was certainly not an easy time to wrap up a thesis.

To Salit, thank you for getting me here! I would never have ended up at UQ if I hadn't connected with you. Thank you for giving so much time and energy to your students no matter what time zone you're in, and for your incredibly prompt responses to anything your students need. I have no idea how you keep up with your email.

To Carissa, thank you for making this monstrous undertaking so enjoyable and rewarding. I had so much fun working (and not working) on seafood projects with you. Thank you for all the opportunities and help you've given me that went far beyond what I'd expect from a supervisor, and that made this experience extra valuable. I'm so glad I got to be your first primary supervisee - you are a fantastic mentor and there will be many more lucky students coming after me.

To James, your energy and your passion about your work is a huge inspiration to me. You set an extremely high bar for conservation science. Thank you for letting me bring some fishiness into the bird club, it's been an absolute pleasure. I appreciate your generosity and everything that you have done for me and for the lab group. You are doing really important work and I'm excited to see where the next phase takes you. I hope one day I'll be able to come up with an idea, create a narrative, write a story, and present to an audience with as much flair as you can. And while I know I may never earn

an invite on the real twitcher missions, I do hope you'll notice that (sea)birds make a notable appearance in my seventh chapter.

To Chris, I don't know where to start listing all the things I've learned from you. Thanks for giving me the opportunity to work with CSIRO, for teaching me about fisheries, project management, real world applications of science, how my van works, statistics... whether you were stuck in a traffic jam, running across a parking lot, driving a child to the emergency room, or supervising a large crane operation. It has been really, really fun, and I am certainly a much better scientist thanks to your mentorship.

It's quite amazing to reflect on the amazing mentorship I've had these past four years. I feel that I'm a very lucky PhD student. Beyond my supervisors there are so many other peers and mentors, academics and otherwise, who provided support and assistance that were integral to completing this thesis. To my past mentors, Colin and Mandy, who inspired me to come to Australia and cheered for me from afar this entire time, you continue to remind me why I want to work in marine conservation. I look forward to my next sabbatical at U. Groenvlei. To everyone at the UQ Centre for Biodiversity and Conservation Science, you are honestly the kindest and friendliest group of people to work with, eat Tim Tams with, drink beers with, and share stuffy offices with. You've helped me with everything from bugs in my R code, massive GIS analyses that crashed the SEES server, battling with colour schemes in Illustrator, and finding the links to the ten different UQ forms I need to fill out. I think all of those things might be equally valuable. I learned more from you all than from stackoverflow.com, and that's not a trivial thing to say.

Thanks to Ruben and Lucia for your generosity and humour, your advice and guidance, and for literally putting a roof over my head when I moved to Australia in the middle of cyclonic rain. To the Greenfire Science Crew, thanks for the great lab retreats, genuinely interesting lab meetings, and thoughtful feedback. To Brooke and Michelle, thanks for answering my panicked Facebook messages at all hours, staying calm when we got lost down dark dirt roads in rural Queensland, and picking leeches off my leg (while I remained completely composed) in the Malaysian rainforest. I'm so impressed with and proud of you both and can't wait to see what you do next. Thanks to Matt, Scott, Felipe, Andrew, and Caitie for your willingness to share your mad skills with all of us. Thanks to Awais and Stephane for going above and beyond in assisting with my complicated Illustrator and

R problems. To Georgina and the UQ Brand Team, thanks for showing me how media outreach is supposed to be done. To Dr. Rosy, I think half of the student body would have succumbed to some exotic sickness if you weren't there to care for us. Thanks to the many staff members who are always calm and helpful when frantic PhD students call them at 4:59pm to help them sort out some administrative mess that's entirely their fault. By "them," I obviously mean "me." Thanks to Aaron for the IT support that always came with good company and lots of jokes, to the entire SEES finance staff for dealing with the things I know absolutely nothing about, and to Erich for your tireless support for all the students. I apologize that after four years I still email the wrong SEES hdr support address at least half the time.

To the fantastic Monitoring Control and Surveillance Analytics team at CSIRO, thank you for such an exciting "real job" experience. Working on these big projects with you all has been challenging and rewarding and has given me invaluable insight into where I want to take my career next. To Madi, thanks for all your encouragement - I'm looking forward to taking on the Indian Ocean with you. To Denise, you are definitely one of the most awesome and inspiring women I know in STEM, and outside of STEM to. We need more of you in science!

Everyone needs a (non-binary, gender inclusive) PhD wife and Maddie, fate and the Beardo gave me the best possible one. I don't know how to thank you for all the good times. You are simply the best raccoon and the best mate.

I owe so much to my friends (who doesn't?) and that became extra clear when it got to that final period of long days and too much coffee trying to push my thesis over the finish line. To Cindy, you are the woman that can do all things and I am so glad I met you. I can't wait to see your new crocs. To Ryan, for your eternally muddy optimism. To Midjimberry Road and the Point Lookout Crew, thanks for the good times, for always asking me how my PhD was going and then instantly reminding me of the more important things in life - you all never let me forget the sun was shining and the waves were cooking. Thanks to Bruce and Jocelyn, for your generosity and the best writing retreat a PhD student could dream of. To Madi, Rhy, Jessie, Paula, Emmie, and Sophie, thanks for the surfs, emergency muffins, sunset sessions and wrap pizzas. It would have been a lot less fun without you. Thanks to Dan Socks Vallentyne for your dance moves, artistic inspiration, and bringing back the joy in basic taxonomy. To my loved ones overseas—in Cape Town, Seattle, Oregon, California, and many scattered cities in between—I'm so lucky to know you have my back (and have for years), and I miss you all a lot. To Julia, for being my proof reader and always giving me more credit and encouragement than I deserve, ever since that day in Kindergarten.

And finally, thank you to my family for being absurdly supportive of this massive endeavour and everything that led up to it.

Financial support

This research was supported by an Australian Government Research Training Program Scholarship.

Keywords

fisheries management, overfishing, threatened species, bycatch, non-target species, biodiversity conservation, sustainable seafood, megafauna, ecological risk assessment, environmental behaviours

Australian and New Zealand Standard Research Classifications (ANZSRC)

ANZSRC code: 050202, Conservation and Biodiversity, 50%
ANZSRC code: 050209, Natural Resource Management, 30%
ANZSRC code: 050211, Wildlife and Habitat Management, 20%

Fields of Research (FoR) Classification

FoR code: 0502, Environmental Science and Management, 100%

Dedications

This thesis is, of course, dedicated to my Mom - who has threatened to read every word, and I think she just might do it.

Table of Contents

List of Figures

List of Tables

List of Abbreviations used in the thesis

ABNJ - Areas Beyond National Jurisdiction
AIS - automatic identification system
CBD - Convention on Biological Diversity
CITES - Convention on International Trade in Endangered Species of Wild Fauna and Flora
CMS - Convention on the Conservation of Migratory Species of Wild Animals
EEZ - Exclusive Economic Zone
ERA - Ecological Risk Assessment
GDP - Gross Domestic Product
GFW - Global Fishing Watch
IOTC - Indian Ocean Tuna Commission
IUCN - International Union for Conservation of Nature
IUU - Illegal, unreported and unregulated
PSA - Productivity Susceptibility Analysis
RFMO - regional fisheries management organization
SAU - Sea Around Us
SESSF - Southern and Southern and Eastern Scalefish and Shark Fishery
UN - United Nations
VMS - vessel monitoring system
WMA - weighted moving average

1 Introduction

1.1 Challenges for effective marine conservation

Marine species and ecosystems are declining at an unprecedented rate, despite increased efforts to curtail these declines (IPBES 2019). The highly connected nature of the marine environment presents a monumental challenge for conservation as threats in one area (e.g., acoustic pollution, habitat destruction, overfishing) can affect species or ecosystems thousands of kilometres away, or—in the case of climate change—even further (Slabbekoorn *et al.* 2010; O'Leary and Roberts 2018; Ramesh *et al.* 2019; Brito-Morales *et al.* 2020). The ocean is under immense pressure from human activity and the rate of change may outpace the rate of conservation action, especially in remote and poorly described habitats like the deep sea and open ocean (Costello 2015; García Molinos *et al.* 2015). Conserving and managing marine systems is further complicated by the fact that almost 95% of the ocean's volume lies beyond national jurisdictions (the High Seas) and is largely ungoverned (FAO 2016). Thus, protection of marine species and ecosystems requires conservation efforts at multiple levels and these efforts, in turn, must match the scale of threat to the impacted biodiversity (Duarte *et al.* 2020). This includes localised actions aimed at key habitat areas, species, or point-source threats, all the way up to broad international policy instruments.

In addressing the many threats synergistically driving the declines in marine biodiversity, there is widespread consensus that improving fisheries management is imperative if we hope to maintain any semblance of functioning ocean ecosystems (Jackson 2001; Costello *et al.* 2010). Fishing affects marine ecosystems in many ways beyond the direct destruction of biodiversity. Some fishing impacts, such as changes to species' behaviour or the genetic makeup of fish populations, are insidious and hard to see or quantify (Jennings and Kaiser 1998). Other impacts, such as destructive fishing methods and overfishing, are more visible. Broadly, overfishing occurs when fisheries deplete a species or population faster than it can replenish itself; that fishing method may be purposeful (target species) or incidental (bycatch) (Froese 2004; Worm *et al.* 2009). Overfishing is an umbrella term used to describe both biological and socioeconomic phenomena. Biological overfishing includes recruitment or growth overfishing of a particular species or population, where the population is too depleted for individuals to find each other and spawn or when there are not enough mature animals to produce a sustainable biomass. As a result, the population declines towards extinction (Hilborn and Hilborn 2012). A related concept that has recently gained more traction is ecosystem overfishing, which considers how excessive removal of fished species can disrupt trophic relationships and ecosystem function potentially to the point of alternative stable states (Murawski 2000; Link and Watson 2019). Economic or Malthusian overfishing occurs when too much fishing drives each fisher's profits below what they should be—the classic story of "too many fishers chasing too few fish" (Hilborn and Hilborn 2012). The Malthusian narrative also touches on the socioeconomic effects of overfishing such as the marginalization of small-scale fisheries in areas characterized by high population growth (Steneck 2009; Finkbeiner *et al.* 2017). Often, multiple types of overfishing occur simultaneously; for example, if a target stock is fished beyond maximum sustainable yield, the trophic web is disrupted as the species' predators and prey are affected, the fishery is no longer profitable, and access to the resources is socially inequitable as small-scale fishers are forced out by subsidized industrial sectors (Murawski 2000; Link and Watson 2019).

Overfishing is a pervasive problem across all types of fisheries, from subsistence and artisanal sectors operating in nearshore territorial waters to the largest industrial fleets operating on the High Seas (Mills *et al.* 2011; Basurto and Nenadovic 2012; Rousseau *et al.* 2019). It is not a recent phenomenon; overfishing precedes all other major human disturbances to marine systems such as acoustic pollution, eutrophication, and anthropogenic climate change (Jackson 2001; Swartz *et al.* 2010). Fishing has been fundamentally altering coastal ecosystems for thousands of years, and many large marine predators were driven to local extinction long before European colonization and the Industrial Revolution (Jackson 2001; McCauley *et al.* 2015; Pauly 2017). But it was in the second half of the twentieth century, when a war-inspired surge in technology and fossil fuel availability catalysed fishers' ability to exploit the ocean, that we began to see collapses of species that had long been considered inexhaustible (Worm *et al.* 2009; Pinsky *et al.* 2011; Hilborn and Hilborn 2012; Zeller and Pauly 2019). Recent assessments by global fisheries report that one third of all commercial fish stocks measured are now considered to be overexploited (FAO 2018) with shared, migratory, and High Seas stocks faring even worse than fisheries in national waters (Cullis-Suzuki and Pauly 2010; Scholtens and Bavinck 2014).

Overexploitation of fishery resources has also led to grave social inequities. Pressure on fishery resources is increasing as demand for seafood continues to rise, both from poorer nations with rapidly growing populations and from wealthier ones with rising middle classes. Seafood is the world's most traded food commodity and is a primary source of protein for billions of people (FAO 2018), but access to this resource is certainly not equitable (Kittinger *et al.* 2017; Teh *et al.* 2019). Before the industrialization of fishing fleets, a given coastal community or country would have had much greater control over the management of their fishery resources. Now, due to the wider footprint of human activities and globalization of fishing, the fisheries supply chain is plagued by glaring inequities in resource access and human rights abuses (Le Manach *et al.* 2013; Tickler *et al.* 2018b,a; Belhabib *et al.* 2019). It is increasingly apparent that the global fisheries supply chain—from recreational to industrial sectors—has been engineered to serve wealthy consumers in food-rich countries at the expense of poorer people in food-scarce countries (Kittinger *et al.* 2017; Ye and Gutierrez 2017; McCauley *et al.* 2018).

1.2 Understanding fishing impacts biodiversity

Fishing is the most common threat listed for marine animals that have been assessed by the Conservation of Nature's Red List of Threatened Species ("Red List") (Kappel 2005; Reynolds *et al.* 2005a; IUCN 2020). A relatively small number of species account for most of the reported catch volume in fisheries globally (Hilborn and Ovando 2014). And yet, many other species interact with fishing gear. Although we do not know the exact number, available fisheries data and conservation assessments suggest that those indirectly targeted account for a substantial proportion of fish species and certain invertebrate taxa (e.g., molluscs and crustaceans) (Reynolds *et al.* 2005b; OBIS 2020; FAO Fisheries and Aquaculture 2021). Many of the species that are regularly caught in fisheries are not actively monitored by fisheries management—especially if they are not primary "targets"—and only a small proportion of the world's marine species have been assessed on the IUCN Red List (Ricard *et al.* 2012; Costello 2015; Hilborn *et al.* 2020).

In both conservation and fisheries management contexts, fisheries catch is often categorized as purposeful (target catch) or incidental (bycatch). While this delineation may be appropriate in some fisheries, in most cases there is a spectrum of targeting, and binary categories quickly fall apart (Davies *et al.* 2009). Bycatch might be sold ("byproduct"), or for a variety of reasons it may be thrown back to sea unutilised ("discards"). In many subsistence or small-scale fisheries, there is some value for almost every animal landed, and very few catches are discarded (Jacquet and Pauly 2008; Zeller *et al.* 2017). Non-industrial fisheries are usually subject to less oversight than industrial sectors and—although their impact can be substantial—catch documentation is usually patchy, and robust stock assessments are rare (Costello *et al.* 2012). Most industrial or actively managed fisheries, in contrast, have a list of species defined as targets that are monitored and managed (Ricard *et al.* 2012). However, market forces, seafood preferences, and changing resource availability can shift targeting dynamics at a pace that exceeds the management framework designed to protect species from overexploitation (Oliver *et al.* 2015). Often there is no requirement to report catch or interactions with non-target species, and sometimes that bycatch is not reported even if retained (Davies *et al.* 2009; Gray and Kennelly 2018). In addition to discarded and retained catch, bycatch also extends to unobserved mortalities. This includes animals that are released or escape the gear but later die, and mortality from ghost-fishing in which animals are caught in lost or discarded gear (Matsuoka *et al.* 2005; Crowder and Murawski 2017).

Regardless of the intention of the fisher and the circumstance of the fishing, bycatch is one of the most pressing issues affecting fisheries management today (Hall *et al.* 2000; Komoroske and Lewison 2015; Gray and Kennelly 2018). Discarded bycatch is a striking waste of biodiversity, can damage equipment, reduces the efficiency of fishing activities, and has deleterious effects on the ecosystem (e.g., changing foraging behaviour of other species) (Hall and Mainprize 2005; Kelleher 2005). Ghost-fishing is perhaps an even more harmful form of waste, causing substantial habitat damage in addition to species mortality (Gilman 2015). For many mega-vertebrates such as sea turtles, marine mammals, seabirds, and elasmobranchs, bycatch mortality is a primary threat driving population declines globally (Lewison *et al.* 2004, 2014; Read *et al.* 2006; Dulvy *et al.* 2017; Hall *et al.* 2017).

Bycatch of iconic species such as dolphins and sea turtles has generated considerable attention and has become a focus of many conservation initiatives, with good reason. Marine megafauna are crucial to maintaining ecosystem function, and have become increasingly valuable for livelihoods and tourism (Estes *et al.* 2016; Grose *et al.* 2020). For certain highly threatened species and populations, even the mortality of a small number of adults could pose a serious threat to the species' viability (Lewison *et al.* 2004). However, although important, these large and relatively visible species represent only a small fraction of the species that interact with fishing gear. Fisheries monitoring and stock assessments will understandably focus on the most valuable or primary target species, which overlooks huge numbers of poorly recorded bony fish, elasmobranchs, and invertebrates that are caught in fishing gear as byproducts or secondary targets (Collier *et al.* 2016; Crespo *et al.* 2019). Less iconic or lesser-known bycatch species also tend to receive very little conservation attention. Many of these species have not been assessed by the Red List, and are usually not closely monitored or managed by fisheries agencies (Ricard *et al.* 2012; Zeller *et al.* 2017).

1.3 Conservation and fisheries management

The development of a network of global fisheries governance has been a fragmented scramble to respond to the latest and loudest crisis, whether a battle over valuable fish stocks or increasing extinction risk of iconic species (Moore *et al.* 2009; Blanchard 2017). Fisheries management has been playing catch up ever since the boom in global fishing capacity. As fishing vessels were able to catch more fish and venture further from their national coastline, conflicts between countries grew increasingly common and more violent (Spijkers *et al.* 2019). Several decades after the end of World War II, the glaring need for international governance of the ocean gave rise to the Law of the Seas (UNCLOS). The UNCLOS addresses many issues besides fishing, but provides the legal baseline that generates nearly all other fisheries governance (Warner 2014; Anderson *et al.* 2019).

From UNCLOS flows a cascade of management organizations and structures at various scales. Bilateral and multilateral organizations, such as regional fisheries management organizations (RFMOs), are extremely important because the bulk of our most valuable fisheries resources are connected across political boundaries, including the High Seas (White and Costello 2014; Palacios-Abrantes *et al.* 2020). However, a lack of real power to enforce regulations and international commitments is a consistent limitation; there is no international agency that polices fisheries on the High Seas and often there is minimal leverage to influence actions within a country's waters (Friedman 2019). Although regional organizations have some clout, the onus to make specific rules to achieve these commitments—and to enforce those rules—typically falls on national environment and fisheries agencies (Barkin and DeSombre 2013; Haas *et al.* 2020). Within a country, fisheries are managed at the level of fishing fleets or sectors based on geographic areas, gear types, and the stocks or populations they are meant to be catching. Fisheries are also categorized by scale (e.g., industrial, recreational, or subsistence), which can be problematic for management (Cooke and Cowx 2006; Arlinghaus *et al.* 2019; Rousseau *et al.* 2019). These overlapping, and often conflicting, delineations and layers of regulation can result in management gaps and patchwork governance (Smith and Basurto 2019).

In addition to fishery-specific legislation, there are many relevant international and national conservation commitments and legislation that interact with fisheries management. The international biodiversity conservation legislation that has arguably had the most measurable effect on fisheries is the Convention on Biological Diversity (CBD) Aichi Target 11, which aims to protect at least 10% of the ocean (by surface area) through marine protected areas (MPAs) and other effective area-based conservation measures, and the "oceans" Sustainable Development Goal (Goal 14), which specifically addresses sustainable management of fisheries (Campbell and Gray 2019). Although these global goals are not limited to MPAs, they have proved to be much easier to conceptualize, implement, and report compared to other tools, and many countries have declared MPAs in their national waters (Laffoley *et al.* 2017). MPAs restrict fishing activities but relatively few of them completely exclude exploitation (Costello and Ballantine 2015); even still, they are generally met with fierce resistance from fishing sectors (Agardy *et al.* 2003).

While MPAs have been the primary conservation tool used to protect biodiversity from fishing, other conservation legislation has influenced fisheries management in important ways. Both the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) and the Convention on Migratory Species (CMS) address trade of marine species threatened by fisheries. Some countries have passed national legislation relevant to fisheries both within and beyond their national waters. For example, because the US is one of the world's major fishing and seafood consuming countries, its Marine Mammal Protection Act, the Endangered Species Act, and the Seafood Import Monitoring Program have effected changes in both domestic fisheries and the international fisheries supply chain particularly for threatened species bycatch (Hall and Mainprize 2005; Williams *et al.* 2016; He 2018). Ultimately, most marine species are not adequately protected by a protected area or by international or national conservation legislation (Gaines *et al.* 2010; Klein *et al.* 2015; Jones *et al.* 2018b; Crespo *et al.* 2019).

Although some national and international legislation and area-based conservation measures do have the capacity to change fishing practices, these tools must be accompanied by effective fisheries management in order to realize long-term gains for biodiversity (Hilborn *et al.* 2004; Hall and Mainprize 2005; Rassweiler *et al.* 2012; Selig *et al.* 2017; Campbell and Gray 2019). However, the current patchwork fisheries governance network does not meet the Convention on Biological Diversity's Sustainable Development Goals to effectively manage fisheries resources and protect marine biodiversity, nor is it sufficient to sustain fisheries to meet the protein and livelihood needs of the global population (Diz *et al.* 2017; Friedman *et al.* 2018; Anderson *et al.* 2019; Crespo *et al.* 2019). Much work has been done to document and detail the many ways fisheries management fails marine biodiversity and the people who depend on it including poor data, lack of compliance with regulations, poor economic performance, reduced catch per unit of fishing effort, failure to manage fishing impacts on ecosystems and habitats, and bycatch of non-target species (Hilborn 2007b; Beddington *et al.* 2007; Salomon *et al.* 2011). These issues arise in all types of fisheries, from recreational or subsistence sectors to large industrial fleets, and in both poor and wealthy countries (Cooke and Cowx 2004; Pitcher *et al.* 2009; Arlinghaus *et al.* 2019). In fact, no country is emblematic of effective management across all their fishing sectors (Mora *et al.* 2009; Melnychuk *et al.* 2017; Juan-Jordá *et al.* 2018; Pons *et al.* 2018). Some certainly perform better than others, but the challenge of improving fisheries sustainability and reducing impacts on marine biodiversity is relevant to all countries globally.

1.4 Solutions to address overfishing

Ultimately, we must reduce catch of species that cannot withstand the current fishing pressure—including less valuable seafood species and non-charismatic species that receive less conservation attention. To do this we need a portfolio of changes to the technical, regulatory, and socioeconomic systems in order to reduce impacts of overfishing on biodiversity, including bycatch of threatened species (Hall and Mainprize 2005).

The technical system refers to how the fishing is conducted. This includes modifications to the fishing gear itself and controls on the fishing effort (e.g., how much, where, when, for what species). Gear modifications aim to make fishing less damaging to the habitat and more selective for target species; this includes use of acoustic deterrent devices, lights to illuminate nets, bird scare lines, and turtle-excluder devices, or changing net material and mesh size, hook shape and size, and type of bait (Senko *et al.* 2014; Squires and Garcia 2018). To control fishing effort, there is a suite of tools that includes individual quotas or total limits for target catch or bycatch, temporal or spatial closures, restrictions on certain gears or fishing methods (e.g., setting purse seines near cetaceans, fishing with lights, or using trawls in certain areas), and safe release protocols for bycatch that is landed alive (Anderson *et al.* 2019; Suuronen and Gilman 2020).

Widespread uptake of these technical measures is only possible if backed by appropriate regulation. Fishers, like all humans, have a tendency to resist change without sufficient incentive (Sutinen and Kuperan 1999). Both fisheries-specific legislation and more general legislation are important (Hall and Mainprize 2005). For example, conservation legislation (e.g., the US Marine Mammal Protection Act, CITES) has the power to impose changes in fishing practices. Regulation of seafood trade, such as import restrictions or food labelling laws, can also influence important changes at sea (Williams *et al.* 2016; Young 2016; Telesetsky 2017; He 2018).

The social system is the third pillar for improved fisheries sustainability. Without a shift in the attitudes and values of fishers to drive changes in behaviour, the impact of changes to the technical and legislative systems will be limited (Hall and Mainprize 2005). It has long been understood that, "most fisheries problems arise from a failure to understand and manage fishermen, and that the study of fishermen should be a major part of fisheries research" (Hilborn 1985). However, it is also difficult to incorporate individual humans—and their complex behaviours—into management frameworks (Fulton *et al.* 2011). Most fisheries management systems are structured so that different levels of the management ladder operate in isolation—with scientists and managers deciding on rules and catch limits without consistent, direct engagement with the fishers themselves—despite frequent acknowledgement of this problem (Granek *et al.* 2008; Freed *et al.* 2016; Stephenson *et al.* 2016; Kincaid *et al.* 2017). Given the global structure of the seafood market, the need for behaviour change extends across the entire supply chain including the fishers, enforcement officers, seafood company executives, distributors, and the consumers themselves.

1.5 Thesis overview

The scale of overfishing in the ocean is vast, and the solutions are not straightforward. However, fisheries management presents an important opportunity for conservation gains because, compared to more disperse and nuanced pressures such as ocean acidification or warming, the source of the threat is identifiable and can be directly managed. There are examples of successful fisheries management, and an understanding of the key components of their successes (Beddington *et al.* 2007; Worm *et al.* 2009; Basurto and Nenadovic 2012; Selig *et al.* 2017). There is also general agreement that biodiversity conservation and fisheries management must work in tandem, which so far has been the exception and not the norm (Hilborn 2007a; Salomon *et al.* 2011; Pauly 2013). Ultimately, both silos want our global ocean to be in a state where species are not driven towards extinction, marine ecosystems are functioning and providing key services, and fisheries are both economically and socially sustainable (Hilborn 2007b; Salomon *et al.* 2011; Davies and Baum 2012). It is not clear how we will achieve this state, but we will certainly need an ensemble of interventions aimed at specific problems across all levels of the fisheries supply chain (Hilborn 2007b; Young *et al.* 2018).

AIM: The aim of this thesis is to address specific problems in marine biodiversity conservation and fisheries management to help improve the status of marine species globally. Although there are numerous problems that require attention, I choose to focus on a selection of problems at four different scales: 1) multinational conservation commitments (**Chapter 2**); 2) global fishing and seafood trade (**Chapter 3**); 3) regional fisheries management (**Chapters 4-6**); and 4) individual fishing fleets (**Chapter 7**). Each chapter focuses on a problem that is policy-relevant and inclusive of marine species of conservation concern.

In **Chapter 2**, I combine the two largest available databases of marine species distributions to map the political jurisdiction of known biodiversity to help facilitate greater multinational collaboration and, hopefully, better outcomes for a wider range of biodiversity. This includes many poorly described species that could be impacted by activities such as fishing, but are left out of existing management mechanisms that are primarily aimed at commercially valuable or charismatic species. Despite knowing that marine species do not observe maritime boundaries, we lack a comprehensive global assessment of which marine species span national jurisdictions, and where they occur. This baseline information on the political distribution of marine biodiversity is particularly timely as 2021 begins the United Nations Decade of Ocean Science for Sustainable Development and is a critical year for ocean conservation. Several major negotiations are currently underway, including the Convention for Biological Diversity's post 2020 global biodiversity framework, and the final intergovernmental convention to draft a new treaty for the conservation of biodiversity in the High Seas.

In **Chapter 3**, I investigate the conservation status of species caught and traded in large-scale fisheries globally to raise awareness of a major seafood sustainability issue. There is increasing public awareness of and demand for more sustainable seafood; for example, a recent survey by the World Economic Forum found that 77% of people who buy seafood regularly support a ban on fishing of endangered species. However, little is known about which fished species are endangered, and who fishes and consumes them. A few of these species (mostly large elasmobranchs) have recently gained attention from the public, and there is increasing pressure on fisheries management to better monitor and manage them. For example, there were thousands of public submissions to the Australian government in support of new legislation to ban shark finning at sea, which passed in 2020.

Chapters 4-6 focus on a case study of bycatch in tuna fisheries in the Indian Ocean, which was conceptualised in consultation with the Secretariat of the Indian Ocean Tuna Commission (IOTC). These chapters focus on a particular gear type (drift gillnets) and suite of species (elasmobranchs, sea turtles, and cetaceans) that the IOTC has identified as priority data gaps. **Chapter 4** is a short commentary on tuna drift gillnets in the Indian Ocean. I argue that this sector is a major data gap and oversight in the global surveillance network, and an example of widespread inconsistencies in fisheries surveillance where there is often little correlation between a fisheries' potential biodiversity and ecosystem impacts and the regulatory attention it receives. In **Chapter 5**, I develop an adaptation of a widely used ecological risk assessment method to make more mathematically robust conclusions from limited biodiversity and threats data, which is a common problem for fisheries managers. I use the example of the susceptibility to capture of cetaceans in Indian Ocean gillnet fisheries to compare results from the new method and the original method (described in Hobday *et al.* 2007). In **Chapter 6**, I use the new ecological risk assessment method to present the first spatially explicit assessment of risk for megafauna in the IOTC management area, including a comparison across the three major tuna gears (longlines, purse seines, and gillnets) and megafauna taxa (sea turtles, cetaceans, and elasmobranchs).

Finally, in **Chapter 7,** I investigate how variations at the level of individual fishers may be an important, and overlooked, opportunity to design management interventions that better address threatened species bycatch. Threatened species bycatch is usually managed with command-and-control approaches (such as technology standards) aimed at the level of a fishing fleet. I use detailed data from five Commonwealth fishery sectors to explore variation in bycatch-to-target ratios among individual operators across different gear types, geographic areas, and bycatch species. If bycatch problems are to some extent driven by particular low performance operators, then incentive-based management approaches aimed at individuals may be more effective than traditional fleet-wide regulations.

Each of these six chapters can be viewed as stand-alone case studies, but all are linked by their relevance to policies related to different scales of fisheries management (Figure 1.1). Together these chapters aim to make incremental but important progress towards addressing gaps in our knowledge of fishing impacts on marine biodiversity and our understanding of how best to move forward.

Figure 1.1: Schematic illustration of the key themes of this thesis. The central focus is management of overfishing as a priority threat to marine biodiversity. All the chapters explore data and management gaps, specifically relating to international conservation targets and the political geography of species (Chapter 2); conservation of non-charismatic biodiversity (Chapter 2), targeted seafood species (Chapter 3), and bycatch (Chapters 5,6,7); gaps in monitoring and surveillance of fishing fleets (Chapter 4), risk assessment methods for data-poor fisheries (Chapters 5,6); and mismatches in the scale of management actions and behaviors of individual actors in the system (Chapter 7).

2 Multinational coordination required for conservation of over 90% of marine species

2.1 Abstract

Marine species are declining at an unprecedented rate, catalysing many nations to adopt conservation and management targets within their jurisdictions. However, marine species and the biophysical processes that sustain them are naive to international borders. An understanding of the prevalence of cross-border species distributions is important for informing high-level conservation strategies, such as bilateral or regional agreements. Here, we examined 28,252 distribution maps to determine the number and locations of transboundary marine plants and animals. Over 90% of species have ranges spanning at least two jurisdictions, with 58% covering more than ten jurisdictions. All jurisdictions have at least one transboundary species, with the highest concentrations of transboundary species in the USA, Australia, Indonesia, and the Areas Beyond National Jurisdiction. Distributions of mapped biodiversity indicate that overcoming the challenges of multinational governance is critical for a much wider suite of species than migratory megavertebrates and commercially exploited fish stocks—the groups that have received the vast majority of multinational management attention. To effectively protect marine biodiversity, international governance mechanisms (particularly those related to the Convention on Biological Diversity, the Convention on Migratory Species, and Regional Seas Organizations) must be expanded to promote multinational conservation planning, and complimented by a holistic governance framework for biodiversity beyond national jurisdiction.

2.2 Introduction

Political jurisdictions have significant economic and cultural implications for humans and can also have a strong influence on regulation and management regimes that affect many marine species. However, species ranges and movements cross administrative boundaries, especially in the marine environment where boundaries are permeable and connectivity is high. For example, larvae can disperse hundreds of kilometres (Ramesh *et al.* 2019) and many marine mammals, sea turtles, seabirds and fish annually migrate across hemispheres.

Yet, global initiatives aimed at promoting the conservation and sustainable use of marine biodiversity, such as the Sustainable Development Goals and the Aichi Biodiversity Targets under the United Nations Convention on Biological Diversity, are implemented by individual countries within their borders with no explicit requirements for international coordination (CBD 2011). Environmental policy built around administrative jurisdictions and structures risks perverse or ineffective outcomes for species because effective management within one jurisdiction may be undermined by inadequate management in other jurisdictions. Examples include protection of only a fraction of a species' life cycle or migration route (Studds *et al.* 2017; Dunn *et al.* 2019), intense harvesting pressure of particular species along arbitrarily located management boundaries (Song *et al.* 2017), and relaxation of conservation policy in neighbouring jurisdictions (Gjerde 2012). To guard against these unintended outcomes, future policy mechanisms must more explicitly address transboundary management. The fundamental disconnect between geopolitical jurisdictions and ecological domains constitutes a major threat to effective long-term conservation, a problem which is exacerbated by projected shifts in species ranges resulting from climate change (Hobday *et al.* 2015; Burden and Fujita 2019).

The legal foundation for transboundary management stems directly from the UN Convention on the Law of the Sea (UNCLOS). However, management mechanisms and governance structures have arisen both through implementing agreements to UNCLOS (e.g., for high sea fisheries through the Fish Stocks Agreement and for deep-sea mining through the establishment of the International Seabed Authority) as well as through the proliferation of biodiversity conventions and organizations (such as the Convention on Biological Diversity, Convention on Migratory Species, UN Food and Agriculture Organization, and Regional Seas Organizations under the UN Environment Programme) (Cullis-Suzuki and Pauly 2010; Ardron *et al.* 2014; Warner 2014). So far, these mechanisms have focused on particular threats to the marine environment or small subsets of marine biodiversity. For example, Regional Seas Programmes offer a regional approach to transboundary management of marine biodiversity (Regional seas programmes 2020), but have been largely focused on pollution and management within jurisdictions (Gjerde 2012). Most other initiatives focus on highly migratory or mobile species (e.g., instruments under the Convention on Migratory Species), charismatic megafauna (e.g., the International Whaling Convention), or commercially valuable species (e.g., the five regional fisheries organizations that manage tuna). Many charismatic megafauna and commercially valuable species are also highly migratory, and the need for multi-national management of these species is clear (Harrison *et al.* 2018). However, only a small fraction of marine biodiversity falls into these categories. Migrations are not the only way in which species are connected across their distributions; even sessile or non-migratory species can be impacted by threats such as overexploitation, noise, debris, or coastal runoff that occur in another part of their distribution (Gregory 2009; Slabbekoorn *et al.* 2010; Ramesh *et al.* 2019).

The need for more holistic and coordinated governance of marine biodiversity is at the core of the negotiations over a new international legally binding instrument on the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction (BBNJ) (United Nations 2020). The solutions being offered in the draft BBNJ agreement start to address the gaps described above, and reflect both the need for a global understanding of marine biodiversity (e.g., through a central scientific body) as well as understanding of regional contexts (implementation through regional bodies, and the central role of capacity development and technology transfer) (Vierros and Harden-Davies 2020). Thus, while there is consensus that effective management of many marine species requires new conservation goals that foster multinational coordination (Gjerde 2012; Kark *et al.* 2015; Crespo *et al.* 2019; Dunn *et al.* 2019), little is known about the magnitude and extent of transboundary marine biodiversity. Using species distribution data on 28,252 marine species to determine how marine biodiversity is distributed across ocean jurisdictions, we identify priorities for coordinating better protection of marine species.

2.3 Materials and Methods

2.3.1 Species maps

We combined maps from the IUCN and AquaMaps, which host the two largest global databases of marine species range maps. The IUCN has published range maps for over 31,000 terrestrial, aquatic, and marine species (IUCN 2019). Experts review the maps and outline the spatial boundaries of each species' distribution, based on observation records and expert knowledge of occurrence and habitat preferences. This analysis focuses on predominantly marine species, although we recognize that the marine and terrestrial categories are ill-suited to many coastal species that occur in mangroves, estuaries, and intertidal zones and depend heavily on terrestrial, fresh and saltwater ecosystems.

We used a series of automated and manual filtering processes to select 9,916 predominantly marine species from the IUCN database. The IUCN classifies species by the broad "system" they occur in (e.g. marine, freshwater, freshwater and marine) and then by finer habitat categories within those systems (e.g. Marine Neritic – Subtidal rock and rocky reefs). First, we used the systems and habitat information to select marine species. We removed all amphibians listed as "marine" (e.g. cane toad, *Rhinella marina*), which can adapt to saline environments but primarily inhabit and depend on freshwater ecosystems (Hopkins and Brodie 2015). We then used two additional filters for taxon groups that are particularly difficult to categorize based on ecosystem and habitat: for birds, we used the expert-reviewed list of seabirds compiled by BirdLife International, and for reptiles, we combined two peer-reviewed lists of marine reptiles (Rasmussen *et al.* 2011; Elfes *et al.* 2013). We considered only species' global distributions, removing 57 maps of subpopulations from the data (most of which are sea turtles or mammals), and then selected cells where each species is extant (presence = 1).

AquaMaps has 22,938 marine species distribution maps in a global 0.5° grid with a relative probability of occurrence for each species in each grid cell. A small proportion (12%) of the maps have been reviewed by experts. We excluded chromists, protists, and bacteria because there were only 47 species maps available for these three kingdoms combined, indicating they were far from comprehensive. For the plant and animal species, we selected cells with at least 50% probability of occurrence and did not repeat the analysis with different probability of occurrence thresholds, as results of previous studies have shown that global scale results are robust to these thresholds (Tittensor *et al.* 2010; Selig *et al.* 2014; O'Hara *et al.* 2017).

To combine the AquaMaps and IUCN databases, we first created a lookup table of species present in both databases by performing several iterations of matching. We began with exact matches of scientific names, then compared the databases using lists of previous names or synonyms. Spelling is not always consistent even for the same name, so we compared the remaining species by genus name and manually checked similar names in online species databases (marinespecies.org, sealifebase.org, fishbase.org). In total, the two datasets provide range maps for 28,252 unique plant and animal species, with 4,033 occurring in both datasets. For these species, we elected to use the IUCN maps because they are expert reviewed and have a conservation status for each species (although many are listed as Data Deficient). Both mapping approaches make assumptions and will introduce errors of commission and omission, especially for poorly studied species where empirical data is lacking. For instance, IUCN maps tend to overpredict coral presence in deep waters and the AquaMaps model tends to extrapolate ranges beyond known occurrences to a greater extent than the expert-reviewed IUCN maps (O'Hara *et al.* 2017). However, overall there is strong agreement between IUCN and AquaMaps range maps especially for well-studied species (e.g. mammals) (O'Hara *et al.* 2017).

2.3.2 Ocean jurisdictions

To analyse the distribution of species across jurisdictions, we analysed the AquaMaps and IUCN datasets separately at their respective resolutions, before rasterizing both spatial grids and reprojecting the 0.5° AquaMaps grid to the higher resolution IUCN raster using nearest neighbour assignment to preserve cell values. Next, we overlaid the combined species map onto a map of maritime jurisdictions from marineregions.org, which we adjusted by combining all Antarctic EEZs into one jurisdiction, and all High Seas regions into the Areas Beyond National Jurisdiction (ABNJ). A number of EEZ boundaries are disputed; we identified the 13 contiguous disputed areas and labelled them as separate jurisdictions with the claiming sovereignties (except for the "Disputed South China Sea," which is claimed by 11 nations).

We calculated the number of jurisdictions in which each species occurs, and compared patterns across broad taxonomic groupings (vertebrates, invertebrates, plants) and IUCN threat statuses. For a species to occur in a jurisdiction, we used a cut-off of 10 cells (1,000km^2) or at least 10% of a species' total range falling in that jurisdiction. We conducted two sensitivity analyses for thresholds for species occurrence in a jurisdiction: one with no cut-off, and a second using a cut-off of five percent of a species' total range or 10 cells in a jurisdiction. Results for the proportion of species that are transboundary differed by less than 1% between the five percent and 10 cell scenarios. We chose the latter for the final analysis because many marine species have extremely large ranges, thus, five percent of their range could encompass an entire jurisdiction, if not multiple jurisdictions. The 10 cell cut-off was the most conservative threshold for determining if a species was transboundary, but compared to the no-cutoff scenario, the proportion of species considered to be transboundary only decreased by 2.1% for AquaMaps species and 1.5% for IUCN species. Ten coastal or semi-aquatic species with small or medium-sized distributions did not meet either criteria (10 cells or 10% of their range in a jurisdiction); for these species, we included all jurisdictions overlapping their ranges. We then calculated the number of single-jurisdiction (n=1) and transboundary (n>1) species occurring in each jurisdiction. To map the distributions of transboundary species globally, we calculated the number of species occurring in each grid cell.

2.3.3 Country governance scores

Effectively managing large numbers of transboundary marine species is a major governance challenge. We used information on six governance indicators from the World Bank to explore correlations between countries' governance capacity and transboundary species richness in their marine estates. We used the "WDI" and "wbstats" packages in R (version 3.6.0) to pull the six governance indicators for each country and year (1996-2018). We then filled missing scores with the closest year available, calculated the average score for each country in 2018, and scaled the composite score from 0-1. For overseas territories that do not have individual governance scores, we substituted the sovereign country's score, recognizing this score often does not accurately reflect the actual governance capacity of the territory (e.g., the many French territories in the Indian Ocean). Seventeen jurisdictions do not have governance scores: Antarctica, the ABNJ, Ascension, Western Sahara, and the 13 disputed jurisdictions. We used Pearson's correlation tests and found no significant correlation between governance score and number of transboundary species for the 209 jurisdictions with WGI scores (r = -0.0479, p = 0.488, 95% CI [-0.1819, 0.0877]), or for the 161 sovereign nations with overseas territories excluded (r =0.0011, p=0.988, 95% CI [-0.1526, 0.1548]).

2.4 Results

We combined the two largest databases of marine species maps, which represent approximately one-fifth of the marine species listed in the Ocean Biogeographic Information System (OBIS) database (OBIS 2020). Large vertebrates have the best representation, with range maps available for close to 100% of chondrichthyans, vascular plants (mangroves and seagrasses), and mammals (Figure 2.1). Coverage is also essentially complete for reptiles and seabirds (Table S1.1). the discrepancies between numbers of species listed in the OBIS database and the maps we included in our analysis arise from different definitions of whether a species is marine (particularly for shorebirds and snakes). Compared to vertebrates and vascular plants, coverage is much poorer for invertebrate chordates (jawless fish, lancelets, and tunicates), invertebrates, and red and green algae (Table S1.1).

Many of the invertebrate groupings are polyphyletic (for example, worms and microscopic animals includes approximately 16 phyla). The polyphyletic groups encompass a wide variety of species that are genetically disparate compared to the well-studied classes of vertebrates. Many of these group classifications are under debate even at high taxonomic levels, such as a phylum.

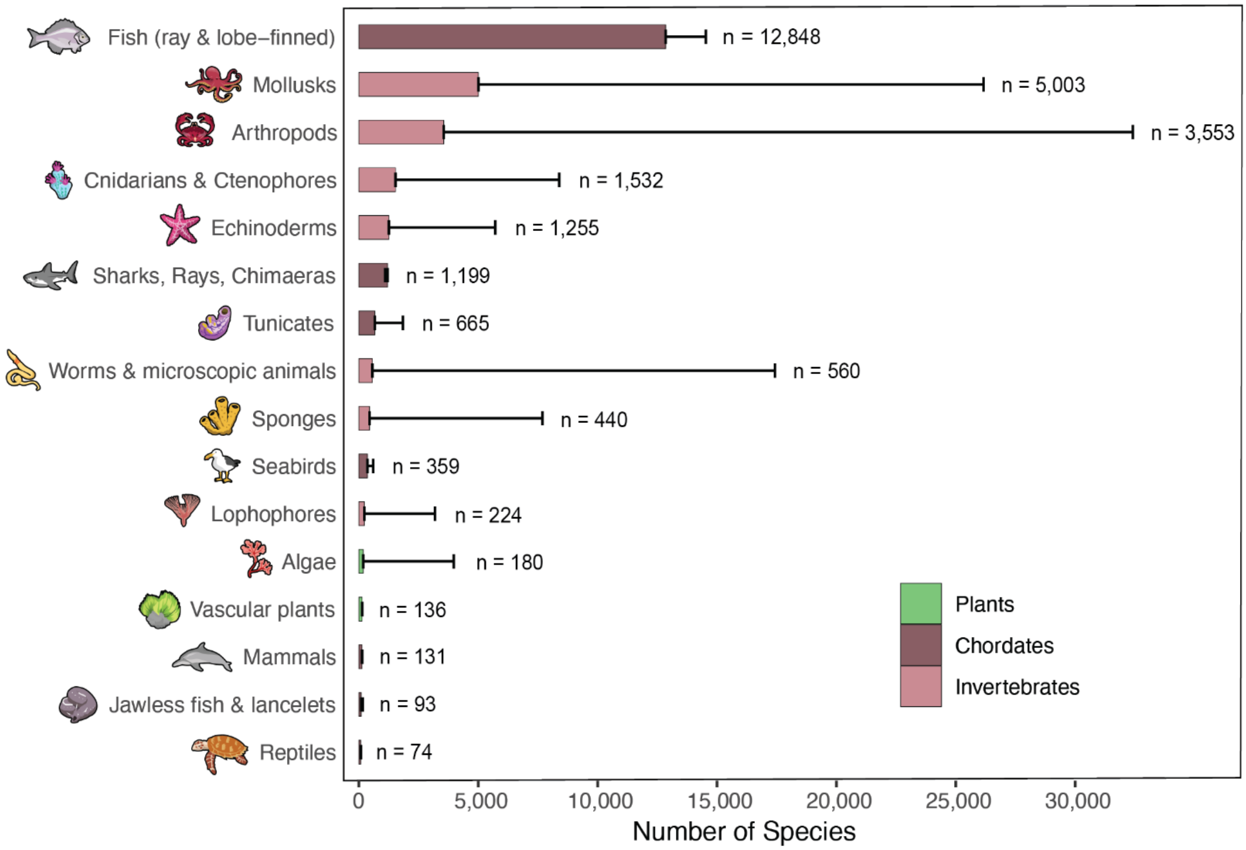

Figure 2.1: Number of species range maps in the combined IUCN and AquaMaps databases (colored bars) compared to the number of confirmed species listed in the OBIS database (black lines). Color indicates whether species are plants, chordates, or invertebrates. Bars are labeled with the number of range maps included in the analysis. Species groups are ordered by descending proportion of recorded species that have range maps.

Only 10% of all mapped marine species assessed occupied a single jurisdiction (i.e. endemics, Figure 2.2), but half of the 228 jurisdictions have endemic species, with Australia (n=706), the USA (n=231), and Mexico (n=174) hosting 41% of the 2,691 endemics (Figure 2.3). Jurisdictions that host species solely within their marine territories are the primary stewards of those species and thus hold sole responsibility for implementing effective conservation actions to ensure their persistence. The other 90% of species (n=25,561) considered in this analysis are found in multiple jurisdictions. Six percent of species occur in exactly two jurisdictions; the country pairs that share the most dual-jurisdiction species are the USA and Mexico (n=240), the USA and Canada (n=224), and Australia and New Zealand (n=193). These countries present important opportunities for conservation partnerships. However, the majority (84%) of transboundary species occupy more than two jurisdictions: 58% occupy more than ten jurisdictions and 15% occupy more than 50 jurisdictions. This presents a significant governance challenge as it requires coordination among approximately a quarter of the nations on Earth to manage these species effectively.

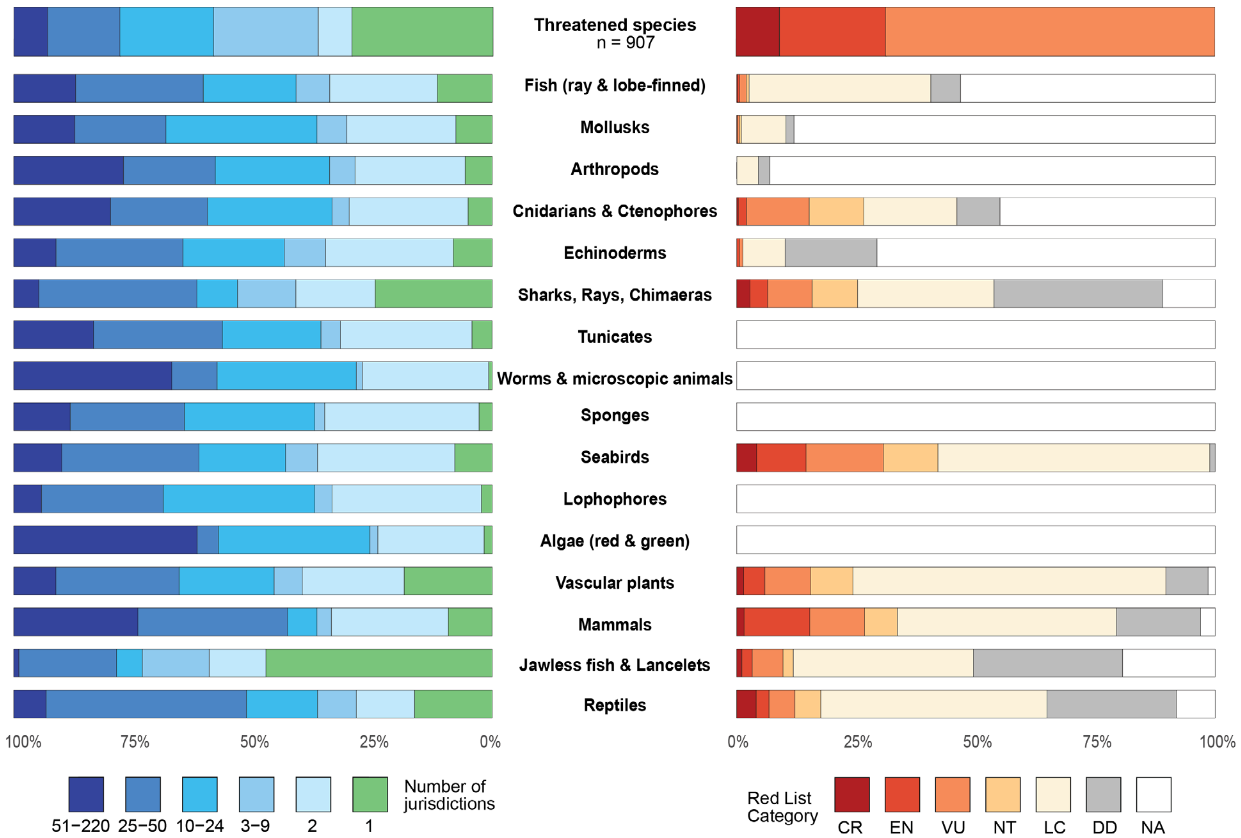

Figure 2.2: Species' conservation statuses and number of jurisdictions overlapping their distributions. Colored bars show the proportions of each taxonomic group in each IUCN threat category (CR = Critically Endangered, EN = Endangered, VU = Vulnerable, NT = Near Threatened, LC = Least Concern, DD = Data Deficient, NA = not assessed) and range of jurisdictions. Taxonomic groups are ordered by descending number of mapped species. Threatened (CR, EN, VU) species are shown at the top.

The taxonomic groups with the highest proportions of transboundary species represent poorly studied phyla of worms and microscopic animals, algae (red and green), lophophores (small sessile filter feeders), and sponges (Figure 2.2). Most of the species with distributions spanning the highest number of jurisdictions are charismatic vertebrates (e.g., cetaceans, sea turtles) and commercially valuable fish (e.g., tunas and billfish, pelagic sharks) (Table S1.2). Orca whales (*Orcinus orca*) occur in the most jurisdictions (n=220), followed by minke whales (*Balaenoptera acutorostrata,* n=211) and common bottlenose dolphins (*Tursiops truncatus,* n=211). However, several species of deep-water fish and cephalopods are also found in hundreds of jurisdictions; for example, short-rod anglerfish (*Microlophichthys microlophus,* n = 200) and jewel enope squid (*Pyroteuthis margaritifera*), which occurs in the largest number of jurisdictions (n=199) of any invertebrate.

Over one-third (35%) of the marine species included have been assigned a threat status by the IUCN, but most (78%) assessed species are vertebrates and 7% are listed as Data Deficient. Consistent with the expected pattern of greater extinction risk for species with smaller ranges (Purvis *et al.* 2000; Reynolds *et al.* 2005a), we find that 71% of species listed as threatened (i.e. classified as Critically Endangered, Endangered, Vulnerable) on the IUCN Red List (n=907) occur in only one jurisdiction compared to 10% of non-threatened species. This provides more opportunities for individual nations with threatened endemics (e.g., Australia, Ecuador, Mexico) to abate the marine extinction crisis.

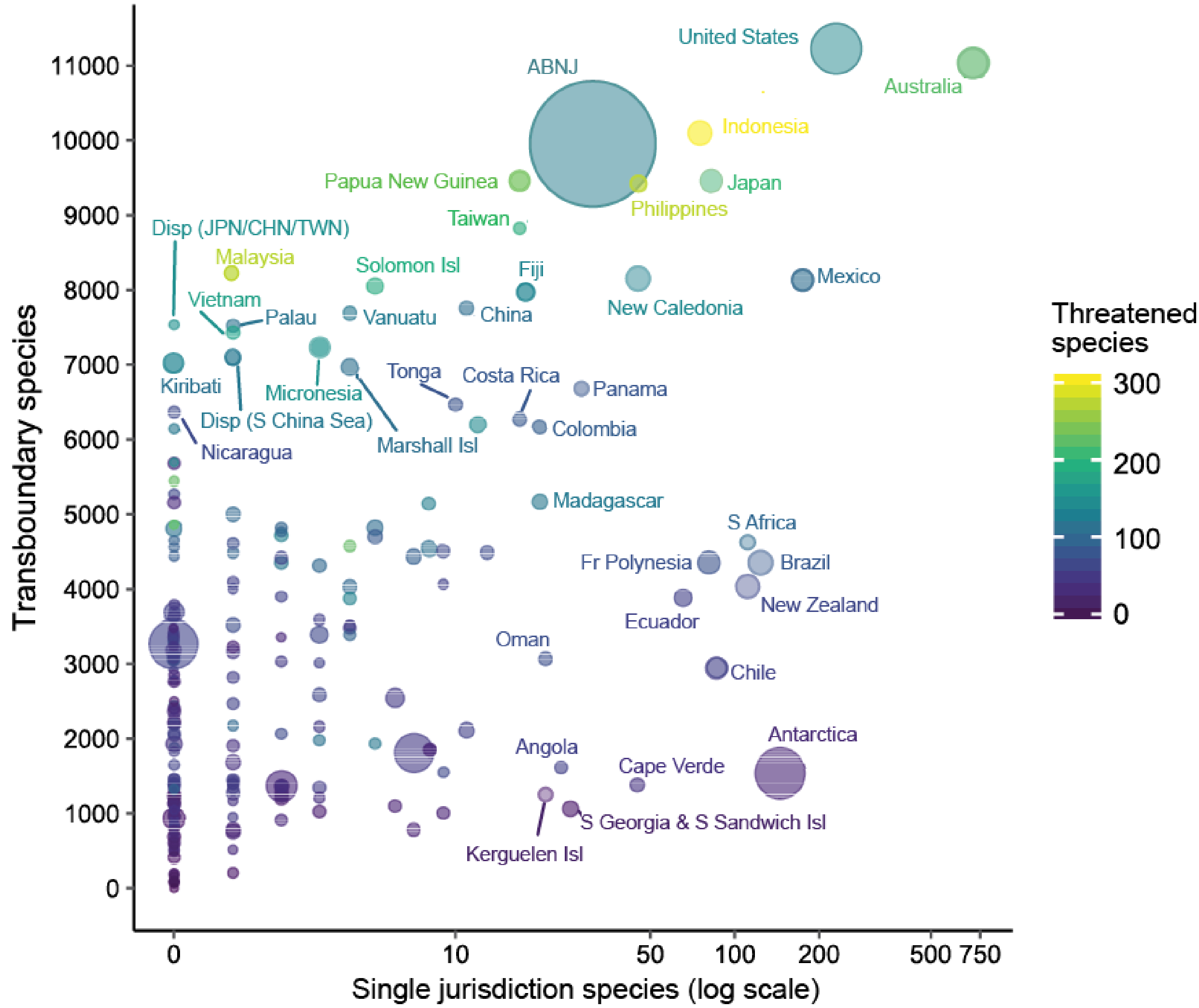

Figure 2.3: Number of species per jurisdiction. Color corresponds to the number of threatened (Critically Endangered, Endangered, or Vulnerable) transboundary species and size corresponds to jurisdiction area (larger dots represent larger areas). All 228 jurisdictions are shown, with labels for jurisdictions ranking in the top 25 for number of transboundary or single jurisdiction species

Transboundary species are concentrated in three biodiversity hotspots in the tropics that have high densities of small island states: East Asia and Oceania, Central America and the Caribbean, and the Western Indian Ocean (Figures 3, 4). As the vast majority of mapped marine species are distributed across multiple jurisdictions, patterns of transboundary species richness are similar to previous species richness maps with smaller subsets of species (e.g., Selig et al. 2014, Tittensor et al. 2010, O'Hara et al. 2017). Our results indicate that transboundary species richness is more closely correlated with latitude than with area; large jurisdictions in temperate latitudes have fewer species than many small tropical jurisdictions (Figure S1.1), although uneven research effort across countries and regions biases our knowledge of marine biodiversity. Sampling bias affects what species are recorded in databases such as OBIS (due to uneven research effort across geographic or taxonomic domains), and what species have enough observations to build a range map. Interestingly, the Mediterranean does not include any of the highest ranking countries for transboundary species richness, even though it has many jurisdictions in a fairly small area, is relatively well-studied, and is considered a hotspot of marine biodiversity (Bianchi and Morri 2000). Our approach likely reduces this sampling bias towards areas such as the Mediterranean because it is most pronounced for large vertebrates (Donaldson *et al.* 2016), whereas we include all mapped plants and animals. Regions such as the Mediterranean and parts of the Arctic are notable for other aspects of biodiversity, for instance, species' range rarity, but are less prominent areas for known species richness (Selig *et al.* 2014).

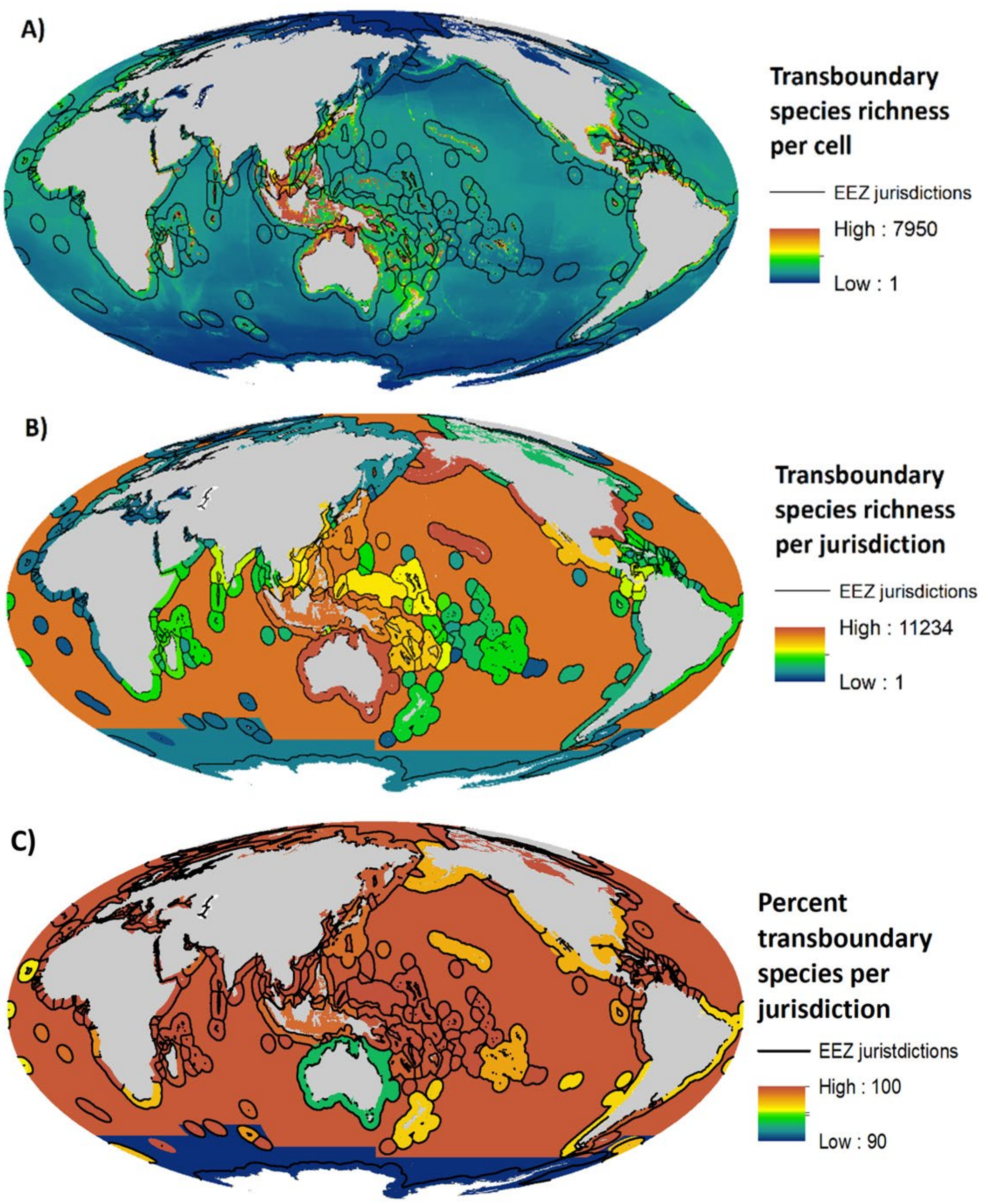

Figure 2.4: Transboundary species richness. Maps of the number of transboundary species (A) per grid cell, (B) per jurisdiction, and (C) as a proportion of the total number of mapped species in each jurisdiction

The jurisdictions with the most transboundary species are the USA, Australia, Indonesia, and Areas Beyond National Jurisdiction (ABNJ) (Figure 2.3, Figure 2.4), with Australia and Indonesia harbouring the greatest richness of threatened transboundary species. Half (114) of the 228 jurisdictions share 100% of their mapped species with at least one other jurisdiction, and all jurisdictions have more than 97% transboundary species except for Antarctica (91.3%), Australia (94.0%), and Cabo Verde (96.8%) (Figure 2.4, Panel C). The country pairs that share the most species are Australia and Papua New Guinea, Australia and Indonesia, and Australia and the Philippines. Countries with large numbers of transboundary species all share many species with ABNJ, especially the USA, Australia, and Japan, which all have more than 5,000 species that also occur in ABNJ. Pearson's correlation tests showed no significant correlation between governance score and number of transboundary species for the 209 jurisdictions with WGI scores (r = -0.0479, p = 0.488, 95% CI [-0.1819, 0.0877]), or for the 161 sovereign nations with overseas territories excluded (r =0.0011, p=0.988, 95% CI [-0.1526, 0.1548]). However, it is notable that many of the tropical countries with large numbers of transboundary species are island states with large ocean territories to govern, and limited capacity to manage or report on marine biodiversity (e.g., New Caledonia, Indonesia; See Figure S1.1, Table S1.3) (Failler *et al.* 2019).

2.5 Discussion

This work establishes that the majority of marine biodiversity is extremely transboundary. The frequency of transboundary distributions is similar among a broad range of taxonomic groups, and many marine species are distributed among large numbers of jurisdictions (more than 50 and up to 220). We find that small, sessile, or non-migratory species have similar transboundary patterns to larger and better-known vertebrates, such as commercially exploited fish stocks (Maureaud et al. 2020). Although there is sampling bias across countries and regions, overall, both understudied and well-studied countries share the vast majority of their marine biodiversity with other jurisdictions.

The transboundary nature of virtually all marine biodiversity exacerbates the complexity of marine conservation. Whereas most land belongs to a single country, over 60% of the ocean's surface—and nearly 95% of its volume—lies beyond national jurisdictions. In the ABNJ, persistent geographic and taxonomic governance gaps have resulted in greater cumulative impacts on species and ecosystems compared to EEZs (O'Hara et al. 2019). ABNJ present a significant governance challenge because there are few avenues for recourse if agreements are not honoured (Friedman 2019), no set rules regarding how to assess transboundary impacts from activities in ABNJ, and no global mechanism to allow the implementation of protected areas in ABNJ. Another key challenge for transboundary marine species conservation is that many biodiversity-rich countries lack governance capacity—a pattern that is also true on land (Mason et al. 2020)—but face additional obstacles when they are small-island nations with vast EEZs to govern (and are often surrounded by the ABNJ). This geography makes effective implementation and enforcement for typical marine conservation strategies, such as marine protected areas, even more difficult (Marinesque et al. 2012; Failler et al. 2019).

Best practice for transboundary conservation considers each country's geographic and cultural context, and includes collaboration, cost-sharing, and resource transfer at multiple scales. This includes both intraregional (e.g., among countries in South East Asia) and interregional (e.g., between Northern European and South East Asian regional management organizations) scales, as well as between individual nations (e.g., Australia and Papua New Guinea). Better outcomes can be achieved by redistributing the burden of conservation, which currently falls disproportionately on countries with lower management capacity (Marinesque et al. 2012; Hanich et al. 2015). International conservation initiatives could encourage countries with greater capacity but fewer species (e.g. Northern European countries) to set higher targets for marine biodiversity in their waters, as well as create avenues to transfer resources to higher biodiversity but lower capacity countries.

An example of coordinated regional management of transboundary species is the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR), the governing body for fisheries and biodiversity in Antarctica and the Southern Ocean. Although focused on commercially exploited biodiversity, CCAMLR has effectively facilitated collaboration among individual States to govern a large and remote area with considerable success (Maguire et al. 2006; Pons et al. 2018). In contrast to terrestrial species (at least terrestrial vertebrates)—of which almost half occur within the borders of individual countries (Mason et al. 2020)—the highly transboundary distribution of marine biodiversity means that complex management contexts such as CCAMLR and the need for countries to engage with governance of ABNJ are the norm, not the exception.

We collated maps for roughly one-fifth of recorded marine species (OBIS 2020). While this analysis is the first attempt to show the geopolitical distribution of marine biodiversity across international boundaries, substantial taxonomic and geographic knowledge gaps remain, especially for invertebrates and algae and for offshore and deep-sea habitats. In particular, large and remote areas such as ABNJ and Antarctica likely harbor many more species than indicated by this analysis. However, we also know surprisingly little about some large, visible species. We limited this study to the plant and animal kingdoms, omitting the chromists (which include kelp) because of very poor representation in the databases. Giant kelps are keystone species that provide critical habitats, but only recently have comprehensive mapping efforts begun (Mora-Soto et al. 2020). Collaboration around research and monitoring—including data sharing—is a crucial element of transboundary conservation (Maureaud et al. 2020), as even research institutions in wealthy nations lack the resources required to explore and document marine biodiversity across a typical EEZ.

Thus, holistic assessment of transboundary marine biodiversity requires integrating data across sectors and engagement beyond traditional academic sources of biodiversity data. If we are to provide reasonable baselines to enable meaningful environmental impact assessment and guide sustainable use of the ocean, then military, industry and traditional sources of knowledge must be fused with scientific research data streams and fed into open-access ocean observing frameworks (e.g., those provided by the Global Ocean Observing System). This requires increased structural support for the Global Ocean Observing System and for its Regional Alliances through increased and targeted support for the Intergovernmental Oceanographic Commission of UNESCO. The opportunity to develop these partnerships and implement these structural changes is now, as part of the strategy for delivering on the goals of the UN Decade of Ocean Science for Sustainable Development. While fisheries biodiversity data remain very difficult to access, other industries have been more open to release of such information. After years of work, the International Seabed Authority has developed an MoU with the Intergovernmental Oceanographic Commission and released its database of contractor biodiversity data, which includes surveys of some of the deepest and most remote areas of the ocean floor. If we are to confront the global marine defaunation crisis and more effectively protect species across borders, incentives for engagement in ocean observation from sectors that typically do not participate in biodiversity conservation are critical.

Global maps of the political distribution of marine biodiversity help inform the need for better and broader reporting and governance of the more than 25,000 mapped transboundary marine species. There are examples of successful conservation or management of transboundary biodiversity for some charismatic migratory species; for example, humpback whales (Bejder et al. 2016), some sea turtle populations (Mazaris et al. 2017), and a few fish stocks, notably Pacific halibut and some

Northeast Pacific salmon stocks (Dankel et al. 2008). However, transboundary management of megavertebrates remains a central obstacle to their conservation with virtually all albatross and migratory sharks listed as threatened or near threatened, along with the majority of sea turtle populations (Dunn et al. 2019). Transboundary fish stocks may be the most egregious example, with shared and highly migratory stocks experiencing twice the level of overfishing and declining more quickly than those within a single jurisdiction (FAO 2014; Palacios-Abrantes et al. 2020).

The need for conservation policy to address transboundary distributions will only become greater as climate change phenomena such as warming, acidification, and sea-level rise alter species ranges, shifting ranges into (and out of) different countries, complicating existing conservation mechanisms for both transboundary and single-country species (Hobday et al. 2015; Burden and Fujita 2019; Kapsenberg and Cyronak 2019; Spijkers et al. 2019). Climate change effects on marine biodiversity also extend beyond shifting species ranges; for example, altering the location of key habitat areas and biological processes (e.g., migration routes, spawning, nesting, and feeding grounds), species' interactions (e.g. invasive species), and ecosystem function (e.g., primary productivity, nutrient processing and exchange) (Doney et al. 2012; Hewitt et al. 2016). Therefore, we urgently need to create flexible and cooperative transboundary management frameworks so that conservation can keep pace with rapid changes in marine biodiversity (Maureaud et al. 2020). We need to conceptualize the biodiversity crisis in the same way we understand climate change, as a truly global problem that requires coordinated global solutions at many different scales (Gattuso et al. 2018).

All countries—even if they are landlocked—are linked to the ocean via the provision of protein, raw materials, and climate regulation, and thus have an interest in protecting marine biodiversity. While persistent political tensions between countries (e.g. South China Sea, Persian Gulf, Baltic Sea) continue to impede ocean conservation efforts, cooperation on biodiversity protection can also serve as a peace-building tool (Mackelworth 2012; Roulin et al. 2017). Given the rapid declines of many marine species, conservation mechanisms must transcend political conflicts so they are robust to transient political fads. Although international cooperation is foundational to the Convention on Biological Diversity (as it is core to the founding Rio Principles), nations remain primarily focused on implementing conservation actions within their own borders without coordinating actions with their adjacent or regional neighbours. Our analysis shows it is imperative that the Strategic Plan for the UN Decade of Ocean Science, the new BBNJ treaty, and the next phase of global biodiversity commitments under the Post-2020 Global Biodiversity Framework incorporate effective mechanisms for transboundary cooperation to improve monitoring, reporting on, protection and governance of marine biodiversity.

3 Over 90 endangered fish and invertebrates are caught in industrial fisheries

3.1 Abstract

Industrial-scale harvest of species at risk of extinction is controversial and usually highly regulated on land and for charismatic marine animals (e.g. whales). In contrast, threatened marine fish species can be legally caught in industrial fisheries. To determine the magnitude and extent of this problem, we analyse global fisheries catch and import data and find reported catch records of 91 globally threatened species. Thirteen of the species are traded internationally and predominantly consumed in European nations. Targeted industrial fishing for 73 of the threatened species accounts for nearly all (99%) of the threatened species catch volume and value. Our results are a conservative estimate of threatened species catch and trade because we only consider species-level data, excluding group records such as 'sharks and rays.' Given the development of new fisheries monitoring technologies and the current push for stronger international mechanisms for biodiversity management, industrial fishing of threatened fish and invertebrates should no longer be neglected in conservation and sustainability commitments.

3.2 Introduction

Seafood is an important source of protein for billions of people globally, with over 80 million tonnes of marine animals taken from the ocean annually for consumption (FAO 2020). Fishing, either targeted or incidental, is the primary driver directly causing declines in marine biodiversity (IPBES 2019). Numerous global and regional-scale initiatives address fishing pressure on marine species, including regional fisheries management bodies, the United Nations Convention on the Law of the Sea and its subsequent agreements, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), and the Convention on Migratory Species (CMS). Yet, one-third of fished stocks are exploited at biologically unstainable levels (FAO 2018) and 1 in 16 marine fish species are listed as threatened with extinction by the International Union for the Conservation of Nature's Red List of Threatened Species (Red List) (IUCN 2019).

A great deal of conservation and fisheries management resources have been invested in reducing the impact of fishing on threatened charismatic species, such as dolphins, turtles, and seabirds (McClenachan *et al.* 2012). While certain populations of threatened fish and invertebrates are closely monitored with fisheries stock assessments, they are treated differently to other wild animals and are, in many cases, permitted to be caught in industrial fisheries regardless of the species' global conservation status. This is unique to marine fish and invertebrates as industrial-scale exploitation of imperilled terrestrial or charismatic marine species is unacceptable from a conservation perspective, even when some populations are considered stable (McClenachan *et al.* 2016; Ripple *et al.* 2019). For example, although highly contested, hunting of African elephants (*Loxodonta africana*)—listed as Vulnerable on the Red List—is allowed for trophies but not for commercial-scale food provision, even where elephants are locally abundant (Di Minin *et al.* 2016; Nelson *et al.* 2016; Batavia *et al.* 2019; IUCN 2019). Similarly, hunting whales for food is highly controversial, even for species or populations that could likely sustain regulated exploitation (Aron *et al.* 2000). In contrast, the International Game Fishing Association grants licences to target many threatened fish and sharks, including species that are Critically Endangered, which receives relatively little attention (Shiffman *et al.* 2014).

While we have yet to fish a widely abundant marine fish or invertebrate species to extinction, we have fished populations or stocks to local or functional extinctions, such as totoaba in Mexico, sturgeons in Europe, and white abalone in California (Dulvy *et al.* 2003). Many stock collapses have been small, short-lived species, proving that slow-growing and long-lived animals are not the only ones at risk (Pinsky *et al.* 2011). Collapses of individual populations do not necessarily precursor species extinction, primarily because there are economic constraints to exploitation of distant or dwindling stocks. However, widespread government subsidies to enhance fishing capacity allow many sectors to operate at eco- nomic loss, further threatening declining fish and invertebrate populations (Vincent *et al.* 2014; Sumaila *et al.* 2019a). Species that span international borders are highly migratory, or exist in areas beyond national jurisdiction where restrictions on fishing are largely voluntary, are at increased risk of extinction even if certain stocks are well managed (Crespo *et al.* 2019). Even for distinct stocks of closely monitored commercial species, there is risk of mismatch between management units and biological units that could mask population declines (Reiss *et al.* 2009; Collette *et al.* 2011). Populations reduced to severely low abundances can take much longer to recover than predicted, and former levels of abundance can become ecologically infeasible (Hutchings 2000; Neubauer *et al.* 2013). Climate change impacts will exacerbate pressures on threatened fish and invertebrates through warming waters, acidification, and loss of critical habitat and prey availability (Doney *et al.* 2012).

Several key fishing and seafood importing nations—notably USA and some European countries—have taken important steps to curb overfishing, actively rebuild overfished stocks, and reduce incidental catch of charismatic species (Williams *et al.* 2016; Ye and Gutierrez 2017). However, the global conservation status of commercially targeted fish and invertebrate species is largely overlooked in fisheries management frame-works, which operate at the level of individual stocks or populations (Watson and Pauly 2001). At a global scale, we lack understanding of the magnitude and extent of exploitation of imperilled species, and which fishing and consuming nations are most important for improving monitoring and management of threatened fish and invertebrates. Here, we use Red List assessment information to (1) determine which globally threatened species appear in industrial catch and import records, (2) determine the volume and value of catch and imports of these species, and (3) identify the countries driving catch and imports of imperilled seafood species.

3.3 Results

Analyses of catch and imports data

We found 92 globally threatened species (50 teleosts, 39 chondrichthyans, and three invertebrates) in industrial fisheries catch records between 2006 and 2014. One of these species, Atlantic cod (*Gadus morhua*), has a controversial Red List status and was omitted from the remainder of our analysis (Hutchings 2000; Powles *et al.* 2000). The remaining 91 species comprise 1.6% of the total catch volume and 2.5% of the value, estimated from ex-vessel price data (the price fishers receive for their landed catch).

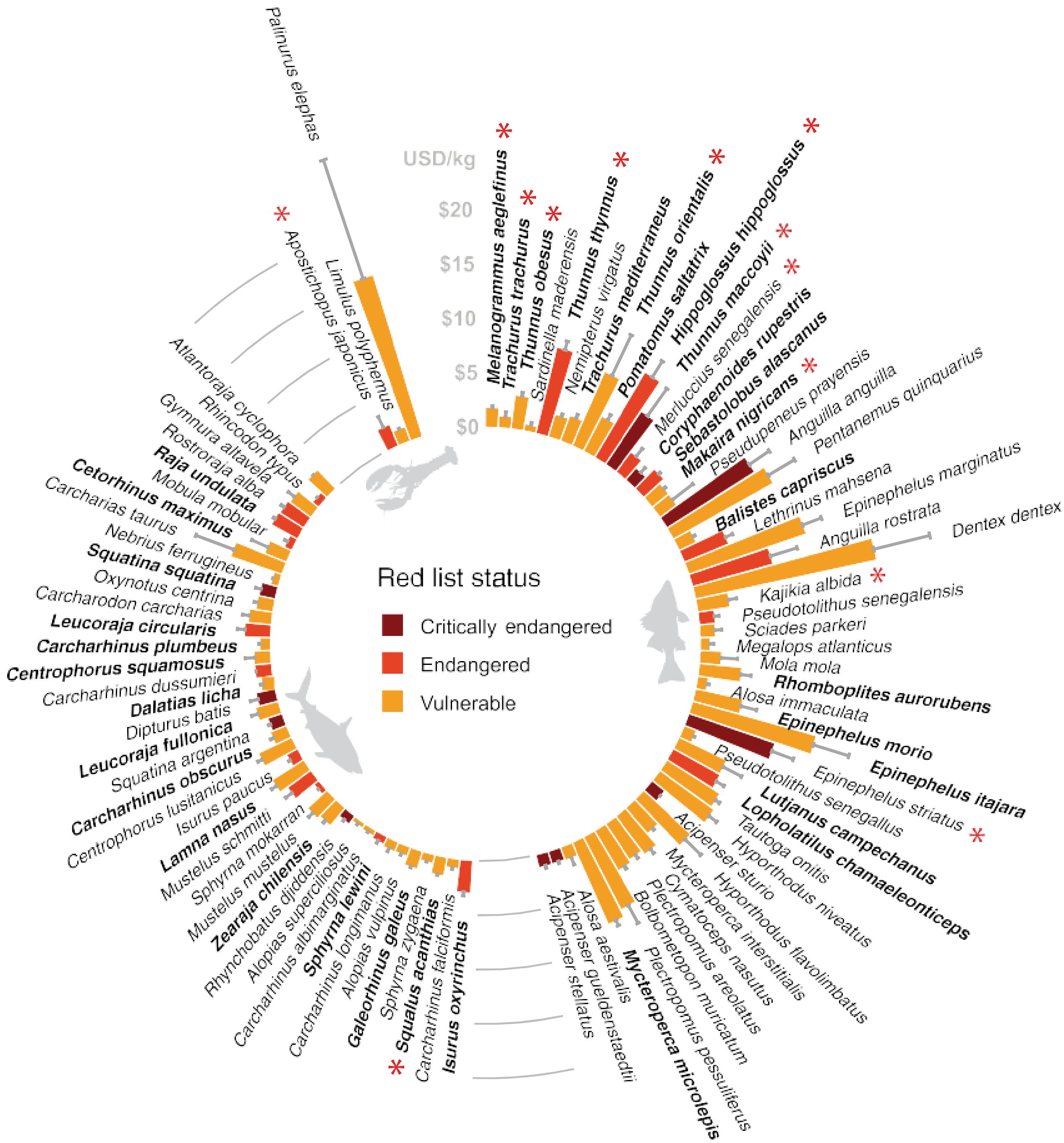

Fig. 3.1: Average ex-vessel price and Red List status for 91 threatened catch species from 2006 to 2014. Prices are global averages for 2010. Error bars show max price for 2010. Species are ordered clockwise by descending catch volume for each taxonomic group (teleosts, chondrichthyans, and invertebrates). The 13 species with red asterisks are found in global import records from 2006 to 2015. The 34 species in bold have commercially exploited populations listed in the RAM Legacy Stock Assessment database. The animal silhouettes are from Freepik.com.

The 60 Vulnerable, 20 Endangered, and 11 Critically Endangered species (Fig. 3.1) have a wide range of body sizes and life history traits, from small and fast growing to large bodied and slow growing. Three wide-ranging teleosts—haddock (*Melanogrammus aeglefinus*), Atlantic horse mackerel (*Trachurus trachurus*), and bigeye tuna (*Thunnus obesus*)—account for 76% of threatened species catch volume and 64% of catch value. Compared to chondrichthyans, teleost species generally fetch higher ex-vessel prices per kg (Fig. 3.1, Table S2.1). However, mean price is less meaningful for chondrichthyans because they are often disaggregated with the liver, skin, gills, and especially the fins sold separately at a higher price per kg than the meat (Dent and Clarke 2015).

We explored the threats data from the Red List assessments and found that fishing is listed as an ongoing threat for 87 (96%) of the threatened species, and is the only ongoing threat listed for the majority of species (Table S2.2, Table S2.2). Large-scale, targeted fishing is specifically listed as a threat for 65 (71%) species and is the only ongoing threat listed for seven species: rock grenadier (*Coryphaenoides rupestris*), sky emperor (*Lethrinus mahsena*), golden threadfin bream (*Nemipterus virgatus*), common spiny lobster (*Palinurus elephas*), and the Southern, Pacific, and Atlantic bluefin tunas (*Thunnus maccoyii, T. orientalis, T. thynnus*). The global population trend is decreasing for 80 (88%) of these species and the remainder have unknown population trends.

Industrial catch of threatened species can be targeted or incidental (bycatch) (Davies *et al.* 2009; Oliver *et al.* 2015). To indicate which threatened species are targeted in industrial fisheries, we used the RAM Stock Legacy Database, which compiles stock assessment results for commercially exploited marine fish and invertebrates around the world (https://www.ramlegacy.org/). We found 34 (37%) of the threatened species listed in the RAM database (Fig. 3.1). These commercially targeted species account for 88% of the threatened species catch volume. Industrial targeting of additional species not listed in the RAM database is indicated by records of international imports in the trade database (four species), and by the IUCN threats data (35 additional species with targeted large- scale fishing listed as a threat). Together, the 73 species account for 99% of threatened species catch volume.

To estimate the final destination of the seafood, we used a global seafood database that uses FAO FishStat Exports and UN ComTrade data to build a virtual marketplace that links fisheries catch to importers and re-exporters (Watson *et al.* 2016). We found species-level import records for 13 of the

91 species (11 teleosts, 1 chondrichthyan, and 1 invertebrate, Fig. 3.1), comprising 2.1% of global import volume and 2.5% of import value (based on ex-vessel prices) from 2006 to 2015. The top three species bycatch volume (Atlantic horse mackerel, haddock, and bigeye tuna) comprise 92% of the total threatened species import volume.

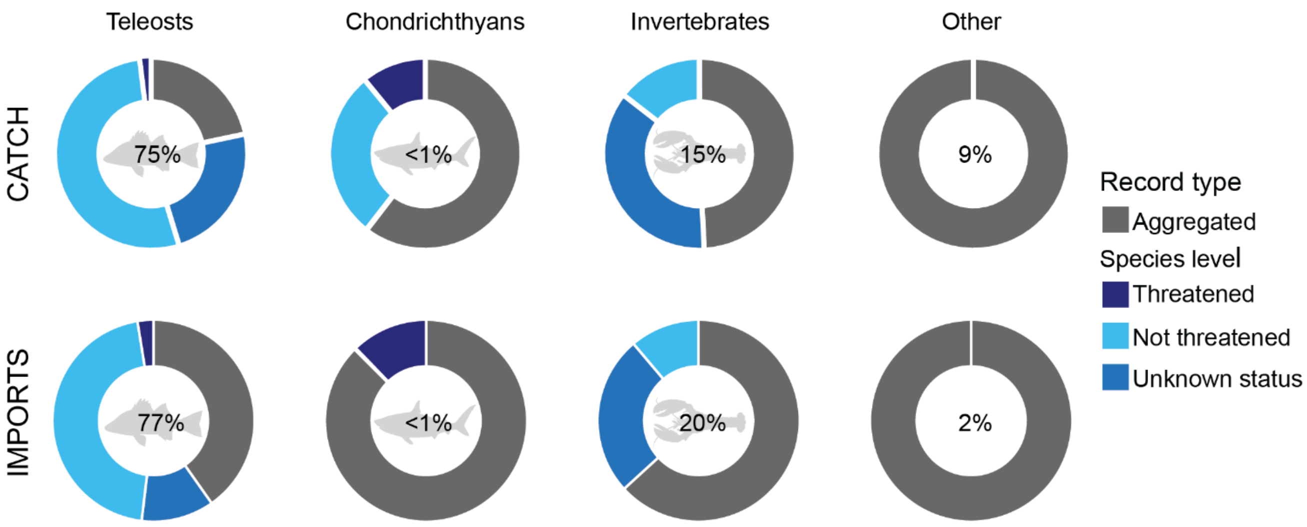

Figure 3.2: Taxonomic resolution of catch and import records. Proportions of catch and imports volumes recorded at species level are shown in blue and aggregated records are shown in grey for teleosts, chondrichthyans, invertebrates, and other commodities (e.g. "marine animals"). The number indicates the proportion of total catch or import volume in each taxonomic group over the time period (2006–2014 for catch and 2006–2015 for imports). Threatened: Critically Endangered, Endangered, or Vulnerable, Not Threatened: Least Concern or Near Threatened, Unknown status: Data Deficient or has not been assessed, Aggregated: not a species-level record. The animal silhouettes are from Freepik.com

Resolution of seafood data

We make a conservative estimate of the volume and value of threatened species catch and imports by limiting our analysis to species-level records. We gauge the extent of our underestimate by comparing species-level to aggregated records (Figure 3.2). One-third (33%) of the reported industrial catch volume from 2006 to 2014 consists of aggregated records such as "Marine pelagic fishes". Almost one-quarter (23%) of the catch volume is comprised of species that are Data Deficient or have not been evaluated on the Red List. Resolution of catch and import records is much better for teleosts and invertebrates than for chondrichthyans, which have more complete Red List coverage but the largest proportion of aggregated records (Figure 3.2). As expected, import records were lower resolution than catch records, with almost half (46%) the total import volume recorded in aggregated commodity groups.

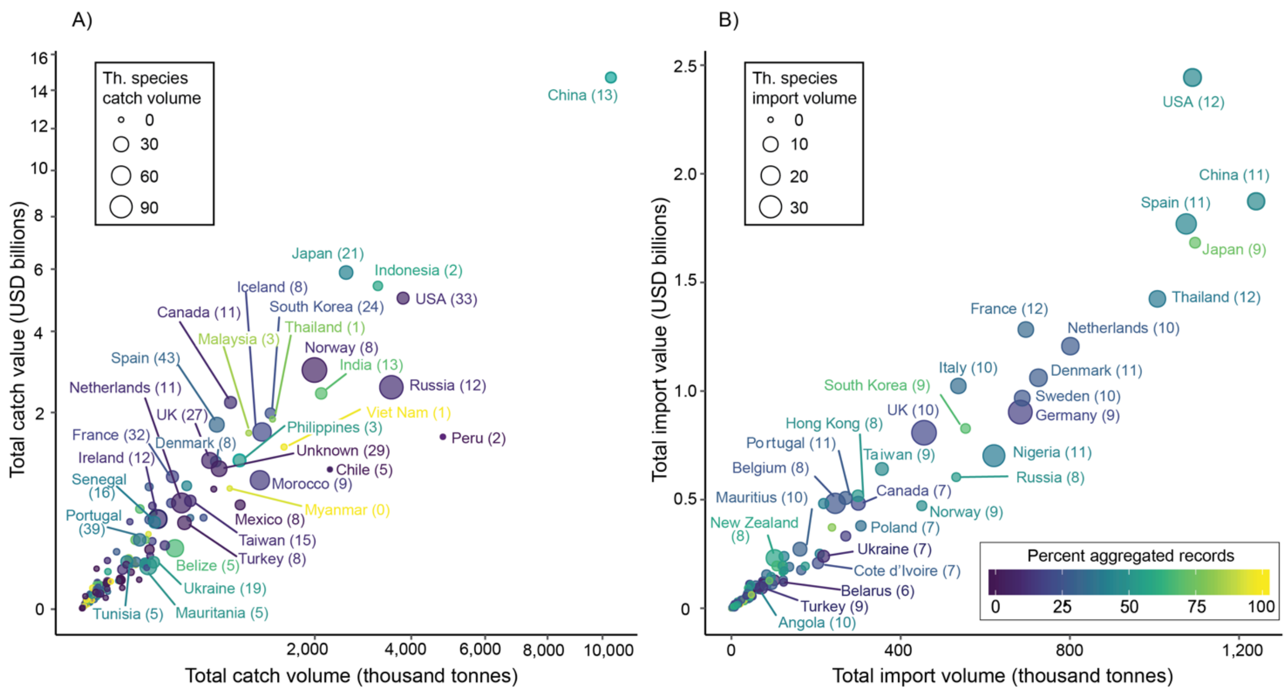

Figure 3.3: Threatened species catch and import volumes and values compared to country totals. Catch volume and estimated value for 163 fishing countries are shown on a log transformed scale (a) and import volume and estimated value for 204 importing countries are shown on a continuous scale (b). Bubble size corresponds to volume of threatened species catch or imports (thousand tons). Number of threatened species each country catches or imports is in parentheses. Color shows the percent of each country's catch or import volume that is aggregated (i.e. yellow indicates catch and import volumes mostly recorded in aggregated groups and purple indicates catch and import volumes mostly recorded to the species level). Volumes and values are weighted moving averages for 2014 for catch and 2015 for imports.

Country level patterns in catch and imports

We found records of the 91 threatened species in catch data from 138 of the 163 fishing countries between 2006 and 2014. On average, these countries catch seven threatened species with Spain, Portugal, and USA catching the highest number (43, 39, and 33 species, respectively). The world's major fishers in terms of catch volume and value were not necessarily the countries catching the largest volumes of threatened species (Figure 3.3). Six of the ten countries with the highest volume and value of threatened species catch are European (e.g. Norway, Russia) (Figure 3.3, Table S2.3). However, several countries known to catch threatened species, especially chondrichthyans, have no records of threatened species in the catch database (e.g. Oman, Hong Kong) (Jabado *et al.* 2018). Also absent were countries severely lacking fisheries management capacity (e.g. Eritrea, Yemen) (Jabado *et al.* 2018) or transparency (e.g. Myanmar, North Korea) (Anticamara *et al.* 2011).

Over the decade, 204 countries reported imports of 13 globally threatened species (Figure 3.3). On average, countries importing threatened species imported six of the 13 species. European countries (e.g. Germany, UK, Spain) and USA comprise most of the top importers of threatened species by volume and value, with Nigeria, Thailand, and China also ranking among the top ten (Figure 3.3, Table S2.4). Countries with few species-specific records compared to aggregated records likely catch or import more threatened species than appear in the data (e.g., Myanmar, Malaysia, Philippines, Japan, and South Korea, Figure 3.3). We used linear models to test whether large volumes of threatened species catch or imports were artefacts of good recordkeeping (more species-level records) or were simply the countries with the largest volumes of catch and imports. Large volumes of threatened species catch were negatively correlated with larger volumes of aggregated records and positively correlated with larger total catch volumes and with higher per capita GDP, which could indicate greater capacity for catch documentation ($df = 139$, adj. $R2 = 0.21$, $p = 0.0015$, $p = 7.4e{-}6$, and $p = 0.0017$, respectively) (Table S2.5). Volume of threatened species imports showed strong positive correlation with total import volume and strong negative correlation with volume of aggregated import records ($df = 206$, adj. $R2 = 0.66$, $p < 2e{-}16$), but not with GDP (Table S2.6). The model explained more of the variation in volume of threatened species imports compared to the model of catch volumes, which is not surprising given the much greater variability in catch volumes and record quality between fishing countries compared to importing countries (Figure 3.3). Many fishing countries deviate from the pattern of more catch and better records corresponding to larger volumes of threatened species; for example, Peru and Chile, which catch large volumes of least concern anchovy and sardine species in relatively selective fishing gears (Figure 3.3A). In contrast, there are fewer records of threatened species imports and poorer record quality overall, thus seafood importers tend to have threatened species imports that are more proportional to their total import volumes (Figure 3.3B). Composite governance score was not a significant predictor variable for catch or imports, likely because fishing threatened species is not illegal and there is no binding international requirement to report catch or imports of fish or invertebrate species in high taxonomic detail.

3.4 Discussion

The 2019 Global Assessment by the Intergovernmental Platform on Biodiversity and Ecosystem Services emphasizes that exploitation is the primary direct driver of marine biodiversity declines (IPBES 2019). The prevalence of fishing—and targeted industrial fishing specifically—in the Red List data further indicates the importance of controlling large-scale exploitation to ensure the future viability of these species. For the first time, we analyse industrial fishing data to determine how much and which type of threatened species are reported in catch records and by whom; information critical for focusing conservation and management action towards threatened marine fish and invertebrates.

We present the most conservative estimate of catch volumes of threatened seafood species by excluding unreported catch, records from non-industrial sectors (which are often not reported to the FAO), or catch reported in aggregated commodity groups. Stock assessment and Red List data suggest that most of these threatened species are targeted to some extent in industrial fisheries. Other threatened fish and invertebrate species were undoubtedly caught in industrial fisheries but were not recorded to the species level. For example, many species of sea cucumbers are fished commercially and listed as threatened on the Red List (Anderson *et al.* 2011b), but the Endangered Japanese spiky sea cucumber (*Apostichopus japonicus*) was the only species that appeared in our global catch data. In addition, there were 444 species in the catch records that were Data Deficient or unassessed on the Red List. Models of extinction risk suggest that up to one-quarter of these unassessed marine species may be threatened (Dulvy *et al.* 2014; Webb and Mindel 2015). The number of Data Deficient or unassessed invertebrate species is particularly concerning because invertebrate fisheries are rapidly expanding as market demand grows and many fish stocks decline (Anderson *et al.* 2011a).

Global catch and import records for industrial fishing indicate that European countries play a central role in driving exploitation of threatened fish and invertebrates. However, developed countries with greater monitoring and management capacity (e.g., UK, Norway, Netherlands) tend to have higher resolution catch and import records, which likely results in more records of threatened species compared to countries with few species-level records (e.g. Myanmar, Thailand, Malaysia). We also identify countries that have poor catch and import documentation despite having the financial means for better monitoring (e.g., China, Spain, Japan). Compared to catch, it is more difficult to identify the countries driving threatened species imports because of the overall lower taxonomic resolution of global seafood trade records. For example, USA has very little industrial reported catch that is not recorded at species level, but almost half of its imported commodities are aggregated records because, like many wealthy nations, it imports seafood from countries with less stringent regulations or management capacity (Willette and Cheng 2018). We likely underestimate the value of imports for wealthy countries and overestimate those of poorer countries because we use ex-vessel prices to compare the value of seafood imports. In general, wealthier countries import more expensive commodities, so the actual value of their imports will be higher compared to lower-income countries importing the same species or commodity group (Watson *et al.* 2016).

Ideally, consumers should be able to purchase seafood that is from a well-managed stock that is secure on a global scale, consistent with World Trade Organization measures relating to the conservation of exhaustible natural resources, international fisheries agreements such as the UN Fish Stocks Agreement, and global targets for biodiversity such as the UN Sustainable Development Goal 15 (Watson and Pauly 2001; Simpfendorfer and Dulvy 2017). Some distinct populations of globally threatened species may be fished sustainably, but the current structure of the seafood supply chain makes it difficult for consumers to make informed, sustainable purchases (Simpfendorfer and Dulvy 2017; Hobbs *et al.* 2019). A crucial first step to better management of fishing pressure on threatened marine species is better taxonomic resolution of catch and trade data, so that we can more accurately understand what species we are catching and consuming and their conservation statuses. Better catch records will also facilitate more accurate Red List assessments (Porszt *et al.* 2012; D'Eon-Eggertson *et al.* 2015) and help identify marine species that merit consideration of CITES or CMS listings, which aim to better monitor and manage international trade. Although a large proportion of teleost species are listed as Least Concern of extinction, many species have only been recorded a handful of times, especially those inhabiting international waters where fisheries are least restricted (Crespo *et al.* 2019).

Governments and fisheries management organizations have made considerable progress in managing fishing and trade of charismatic marine species such as whales and sea turtles (McClenachan *et al.*

2012) but we maintain a cognitive dissonance with threatened fish and invertebrates that we eat. Some fishing sectors have national catch restrictions for certain endangered species, usually for large chondrichthyans caught primarily as bycatch (e.g. basking shark *Cetorhinus maximus*) (Simpfendorfer and Dulvy 2017; IUCN 2019). However, the US Endangered Species Act is the only national legislation that effectively extends beyond direct exploitation of species within domestic borders to address imported species (Foley *et al.* 2017). Threatened seafood species also receive limited international protection from agreements such as the CMS or CITES, which address but do not always restrict international trade, do not restrict catch, and only apply to voluntary signatory countries. None of the 13 internationally imported threatened species from our data are listed on these two conventions (Table S2.1), although many meet the criteria as endangered or migratory species. Atlantic bluefin tuna (Endangered) was denied CITES listing in 2010 after fierce resistance from Japan and other wealthy countries with tuna fleets; the Vulnerable piked dogfish (*Squalus acanthias*) was also denied listing, and the Critically Endangered Southern bluefin tuna has never been nominated (Sky 2010; UNEP-WCMC 2019). Ultimately, voluntary international agreements such as CITES will offer limited protection to imperilled species, unless the signatories shift their focus from purely economic interests to the long-term viability of marine species. Expanding the scope and power of international agreements, such as the recent negotiation of a legally binding instrument for biodiversity beyond national jurisdiction, could potentially be a major gain for threatened fish and invertebrates (Crespo *et al.* 2019).

Despite the challenges of improving traceability of species across the seafood supply chain, it is increasingly possible and cost effective to identify an animal and trace it to the consumer using emerging technologies such as electronic monitoring, DNA testing, code tags, blockchain, data mining, and artificial intelligence (Lewis and Boyle 2017; Kamilaris *et al.* 2019; Probst 2019). For example, OpenSc—one of several new digital platforms for tracing food—has been successful in pilot projects for tuna and Patagonian toothfish (Boulais 2019; WWF 2019). Greater and more coordinated efforts from governments, seafood companies, and NGOs are necessary to implement catch documentation schemes, align processes across supply chains, and develop better incentives to improve traceability (Hosch and Blaha 2017; Lewis and Boyle 2017).

A few glaring regulatory loopholes remain that impede traceability of threatened species, and seafood in general. One major problem is lack of mandatory reporting of species not listed as targets, as many species are caught intentionally and incidentally in different contexts (Oliver *et al.* 2015). Fisheries management often lags behind evolving patterns of targeting as changing resource availability shifts species from bycatch to targets (Davies *et al.* 2009). A second example is the common practice of transshipment—where catch is transferred from a fishing vessel to a cargo vessel (reefer) at sea—often beyond national jurisdiction and enforcement systems (Miller *et al.* 2018a). A third key problem is flags of convenience—vessels registered under flags of countries not affiliated with the owner—which typically have lax regulation or enforcement (Miller *et al.* 2018a). For example, Russia and Belize both have very high reported catch volumes of the 91 threatened species in our databases, but are well-known flags of convenience for both fishing and reefer vessels, so much of that catch is probably taken and traded by foreign-owned ships (Miller *et al.* 2018a).

Major fishers and seafood consumers such as China, Japan, USA, and European nations have power and responsibility to improve traceability and sustainability of seafood globally (Miller *et al.* 2014), and are also important for reducing industrial fishing impacts on threatened species. Our analysis also highlights several countries that are not among the world's top fishers or seafood consumers but are particularly important for threatened species. These countries either have large recorded catch or imports of threa- tened species (e.g. Morocco, Germany) or very low-resolution records (e.g. Myanmar, Malaysia), which may mask high incidence of threatened species. Importantly, the global catch and imports data is recorded at the country level, but a relatively small number of transnational corporations actually do the fishing, processing, and trading (Osterblom *et al.* 2015). The countries that license these companies to fish in their waters or consume their seafood products can pressure seafood companies to improve production practices. Regional fisheries management and non-governmental organizations both play important roles in persuading and incentivizing countries—and the seafood companies they authorize—to perform better.

Here, we provide the most conservative inventory of global catch and imports of threatened fish and invertebrates as a basis to prioritise research and policy development at the international level. Greater awareness of the global conservation status of seafood species from seafood consumers, fisheries management institutions, and conservation organizations would help expand these initiatives to commercially exploited species of conservation concern. Efforts to preserve marine biodiversity and maintain viable ecosystems will fail if we focus only on charismatic species or individual stocks. We need to treat fish and invertebrates as wild marine animals as well as seafood commodities, better align conservation assessments and fisheries management frameworks, and reduce fishing pressure that is pushing species towards extinction.

3.5 Methods

IUCN Red List

We explored the IUCN Red List conservation statuses of all seafood commodities in two global catch and trade databases. We used the Red List because it is the most commonly used global dataset for identifying the types of threat and levels of extinction risk to marine species, it incorporates fishery stock assessment information where available, and typically aligns with fishery management statuses where populations listed as threatened are usually below target fisheries reference points for stock biomass or target catch (Dulvy *et al.* 2005; Collette *et al.* 2011; Davies and Baum 2012; Hornborg *et al.* 2013; D'Eon-Eggertson *et al.* 2015; Jabado *et al.* 2018). However, we acknowledge two issues with Red List assessments of some commercially targeted species. First, the global status does not capture the heterogeneity of distinct populations, which is substantial for some species (e.g. Atlantic cod). Second, the Red List's population reduction thresholds were originally designed for terrestrial species, and may overestimate the extinction risk of abundant and fecund species such as tuna and sardines (Hutchings 2000; Powles *et al.* 2000; Mace *et al.* 2008).

We selected all marine invertebrates, teleosts, and chondrichthyan species from the Red List version 2019.2 and matched to the commodity list using species names. We included synonyms and defunct names provided by IUCN. We considered only the global Red List assessments—excluding regional assessments— for three main reasons: (1) regional assessments are disproportionately available for Europe and North America, (2) there is often uncertainty about the congruence between biological populations and management units, and (3) for many species it is not possible to accurately determine which population the catch originates from the global catch data (Reiss *et al.* 2009). We made an exception for Atlantic cod, where we used the 2013 European assessment (Least Concern, population trend is increasing) because the 1996 assessment of Atlantic cod as globally Vulnerable was highly controversial (Hutchings 2000; Powles *et al.* 2000). Stocks in North America remain depleted after a dramatic crash in the 1980s and the vast majority of the global catch of Atlantic cod now comes from Europe, although there remains some concern about population declines and potential overexploitation of the European cod stocks (Fernandes *et al.* 2017).

We explored the Red List information on threats to the 91 threatened species recorded in the catch and imports data, excluding threats not listed as "Ongoing”. We divided the threats into six categories based on the IUCN threats classification scheme, recognizing that the scale of the fishing (e.g. industrial versus small scale) is difficult to define: (1) targeted industrial fishing, (2) incidental industrial fishing, (3) targeted non-industrial fishing, (4) incidental non-industrial fishing, and (5) unspecified fishing. Any threat other than fishing (e.g., pollution, climate change, intrinsic characteristics) we categorized as (6) other (Table S2.2).

Global catch and imports data

We linked the Red List information to species-level records in global catch and trade databases to estimate the volume and value of reported threatened species catch and imports from industrial fishing, relative to total catch and imports. We used the Sea Around Us (SAU) global catch database (Pauly *et al.* 2020) to calculate the total and average annual catch volumes for each wild-caught marine seafood commodity and fishing country or flag state (referred to as countries). The SAU database builds from FAO global catch data using a bottom up, country and sector-specific approach that draws on grey literature and other sources to reconstruct catch patterns in each country. We limit our analysis to reported catch from industrial sectors, which are major suppliers of internationally traded seafood and tend to have more taxonomically detailed catch documentation. We repeated the analysis using a second global catch database also built from FAO catch data (Watson and Tidd 2018) (Table S2.7, Figure S2.1). We excluded one species, *Coregonus lavaretus*, because it exclusively inhabits freshwater ecosystems. There were more species-level catch records in the SAU database, but overall the patterns of threatened species catch and fishing countries were similar, with the exception of China. China’s total reported catch in the SAU database is more than double any other fishing country, but the 2014 volume is likely an overestimate because it is derived from reconstructed catch estimates during a period of enormous expansion enabled by massive subsidies (Pauly and Zeller 2019; FAO 2020).

We then used a global seafood trade database to estimate the volume of international imports of each seafood commodity across importing countries, our best estimate of where the species is consumed (Watson *et al.* 2016). The seafood trade database builds a virtual marketplace that links FAO FishStat Exports data to the fisheries catch. Country catches are matched to FAO FishStat exports records using the best approximations of taxa to commodity descriptions and data on bilateral trade partners from the United Nation's International Trade Statistics Database (UN ComTrade) (Watson *et al.* 2016). The virtual marketplace identifies the source of the export (domestic catch, domestic aquaculture, foreign fishing, or re-exported product), and categorizes all non-matching exports or problematic import records as a re-export. Internationally traded seafood is difficult to trace through complex loops of importation, processing, and re-exportation as a different product, especially by major processors such as China30. We considered each country's catch and imports, excluding re-exported trade and aquaculture records.

Species biomass and fishing effort fluctuate considerably across years, so we selected the most recent decade in the databases (2006–2014 for catch and 2006–2015 for imports) to understand broad trends in fishing and seafood trade. To compare trends across threatened species and fishing or importing countries, we calculated weighted moving averages (WMAs) with 8- and 9-year windows for the most recent year (2014 and 2015, respectively). The WMA gives greater weight to more recent years by multiplying each value by a weighting factor. It is a common metric for forecasting data because it better represents trends compared to a simple average or total values.

Catch and imports are recorded as tonnes, underrepresenting the importance of small-bodied or rare species. We used ex-vessel price data from SAU to compare the economic value of threatened fish and invertebrates to industrial fisheries and to better represent low volume but higher value species. The SAU database uses available price records to derive average ex-vessel prices (the price the fishers receive when they sell their landed catch), adjusted to USD, for all species-specific and non-species-specific commodities in the global catch database for each fishing country and year from 1950 to 2010 (Tai *et al.* 2017). Catch value is the product of volume and ex-vessel price for each commodity, country, and year. The price paid at the dock is often far less than the price of a highly processed commodity (e.g. breaded fillets) at its final import destination, but we use ex-vessel price to compare import value as well as catch value because it provides a data-driven metric of relative value for each species and commodity at a global scale.

Statistical tests

We posed two hypotheses about the key countries driving catch and trade of threatened species in industrial fisheries: (1) the world's major fishers and importers of all seafood commodities are the same countries that catch and import the largest volumes of threatened species, and (2) countries with better taxonomic resolution in their catch and import records will have larger volumes of threatened species recorded. To explore these questions, we used multiple linear regression models of threatened species catch and import volumes compared to the total volumes, and to the volumes of other record types (e.g. aggregated records). We tested per capita GDP and composite governance score as predictor variables using World Bank data accessed via the WDI and wbstats packages in R.

4 Need to address gaps in global fisheries observation

4.1 Introduction

Military technologies accelerated the ability to navigate and find fish, leading to widespread overfishing and some rapid stock declines (Pauly et al. 2002). These technologies evolved into radar-based systems that enable near real-time observation of fishing vessels. Harvest rates increased dramatically with these technologies, but lack of basic monitoring and surveillance remains a major problem for global fisheries management (Beddington et al. 2007; Anticamara et al. 2011). Much knowledge of global fishing effort is still derived from handwritten logbooks. Vessels equipped with transponders can hide their location or purpose, and prosecution success for most fishing misdemeanors is very low (Gross 2018). Consequently, illegal, unreported, and unregulated (IUU) fishing has hindered effective management of marine ecosystems, while one-third of assessed marine fish stocks are fished at biologically unsustainable levels (FAO 2018). and many more unassessed species and stocks are almost certainly overharvested (Pitcher and Cheung 2013). Information on maritime activity is freely available or can be purchased from data vendors (e.g., MarineTraffic and Global Fishing Watch). Most providers harvest information transmitted from vessels' automatic identification systems (AISs) or vessel monitoring systems (VMSs). Despite limitations of data derived from these systems, there are near real-time databases of fishing effort that provide opportunities to combat IUU fishing, better understand where and what fleets need management attention, illuminate key drivers of fishing behaviors, and identify overlap with marine resources and vulnerable species (Cabral et al. 2018; Kroodsma et al. 2018). But not all countries require transponders, especially for small vessels; therefore, even with these advances much of the world's fishing remains undetected.

4.2 Information Gaps

Gillnets (anchored or drifting) often generate high by-catch rates, particularly for vulnerable megafauna (e.g., marine mammals) (Lewison et al. 2004). Gillnets are simple and relatively cheap to operate and, thus, commonly used in coastal waters around the world, particularly in developing countries (Northridge et al. 2017). Tuna gillnet fisheries in the Indian Ocean have expanded since 2003. Nations, such as Iran, India, and Sri Lanka, each operate thousands of boats (Aranda 2017). Large-scale illegal gillnetting is rampant, despite a 1992 UN Resolution banning drift gillnets over 2.5 km in international waters (Ardill et al. 2013). There are multiple reports of illegal high-seas gillnet fishing by Chinese longline vessels (Cutlip 2016), and Pakistani gillnetters reportedly set 26-

km-long nets in the high seas (Moazzam 2012). Equally problematic are legal but unmonitored fisheries. Indian Ocean countries must submit catch and effort data by cell degree for purse seines and longlines for their industrial tuna sectors to the Indian Ocean Tuna Commission (IOTC), whereas no spatial information is required for gillnet vessels, which rarely have logbooks, observers, or AIS (Ardill et al. 2013). Gillnets are absent from open-source satellite maps because most of the >60,000 estimated vessels are considered artisanal or coastal, even though some are as large and fish the same areas as the industrial vessels (Aranda 2017).

4.3 Monitoring and Surveillance

Inconsistent monitoring of fisheries at national and regional scales threatens food security and marine biodiversity. Missing catch and effort information leads to an inaccurate understanding of stock status and likely contributes to unsustainable catch allowances and stock collapses (Beddington et al. 2007). Effective monitoring of fishing effort and surveillance of vessel compliance leads to better-managed fisheries that are more profitable over the long term (Sumaila et al. 2012; Pons et al. 2017). Indonesia recently reported a decrease in illegal fishing activity and increased profits for fishers after a multifaceted initiative, which included publicizing their VMS information to improve transparency, monitoring, and enforcement (Cabral et al. 2018). Better management of target stocks also has important spillover benefits for bycatch species (Burgess et al. 2018).

4.4 Taking Responsibility

The International Maritime Organization mandates AIS on large vessels, and regional fisheries management organizations, such as the IOTC, have requested better monitoring data from their member countries, but these standards must be implemented at the national level. It is, therefore, essential that developing nations receive financial and technical support and developed nations show leadership by strictly enforcing standards. This must be seen not as a sunken cost but as better prioritization of budgets to improve management and longer-term stock viability. For instance, implementing some basic monitoring and surveillance costs less than subsidizing unprofitable fisheries. Global high-seas fishing fleets received $4.2 billion in government subsidies in 2014, far exceeding the estimated $1.4 billion net economic benefit of those fisheries (Sala *et al.* 2018a). The largest subsidies are given by governments of developed countries (Japan, China, European Union), but many of these countries are underperforming in their monitoring and surveillance, especially of distant-water fleets (Bellmann et al. 2016; Sumaila et al. 2016; Tickler et al. 2018b).

Developing countries face more obstacles in balancing food provision and economic needs with marine biodiversity and ecosystem health. At the extreme is Yemen, which is believed to have several thousand gillnetters without transponders and, understandably, is yet to submit a report to the IOTC (Allison et al. 2009; Moreno and Herrera 2013). India is much more developed but also faces depleted coastal fisheries. In response, the government promoted the growth and mechanization of offshore and deep-water fleets with subsidies for engines and fish finders (Bhat and Bhatta 2006), India now ranks seventh in global seafood exports (FAO 2018), operates the region's second largest tuna gillnet sector after Iran (Aranda 2017), and is the ninth largest subsidizer among developing nations, providing approximately half a billion (U.S. $) in subsidies in 2009, mostly to enhance fishing capacity (Sumaila et al. 2016). The government has not provided for adequate monitoring of fleet expansion, even though AIS also provides safety benefits (its original purpose) such as preventing ship collisions.

Effective monitoring requires more than a few pieces of electronics and software. The government must have the infrastructure to manage the data, analyze the outputs, and respond with appropriate enforcement actions. Assuming better surveillance is necessary for successful management in the long term, investing in monitoring and surveillance is a better choice than expanding fishing capacity. However, long-term visions are supplanted by shorter-term livelihood needs unless there is a political will to improve fishing practices and tangible rewards for greater transparency (e.g., higher-value seafood products). Thus, the responsibility for improvement of fisheries monitoring also lies with seafood corporations and consuming nations. In 2012, 13 corporations controlled about 40% of the catch of the world's largest and most valuable stocks (Osterblom et al. 2015), and Japan, the United States, China, and the European Union account for over two-thirds of global seafood imports (FAO 2018). Governments of fishing countries are often shamed for their poor practices, but less public attention has focused on consumer nations or the corporations directly responsible for fishing. Concerted efforts have forced the Thai government to invest in better fisheries governance and Thai Union, one of the world's largest seafood producers, to commit to better practices (Lewis and Boyle 2017). Another example is U.S. legislation requiring imported seafood to meet stricter management standards (Williams et al. 2016). Public awareness of seafood sustainability has increased but more direct action is needed, and costs of these actions must be spread more equitably across the participators and beneficiaries of marine fisheries.

The necessary restructuring of the seafood supply chain is daunting, and monitoring and surveillance are only two pieces of the puzzle. However, they are essential because making marine activities more visible makes them more governable (Toonen and Bush 2018). Because information on fishing activity and especially fishing locations is kept tightly guarded by management agencies and by vessel owners, stronger leadership from major non-governmental organizations, UN Food and Agriculture Organisation, regional management bodies, and seafood consumers is necessary to allow for improved monitoring and surveillance.

4.5 Future Science Needs

Without demonizing developing countries or unfairly assigning blame, technological advances should be used to determine which fisheries are underperforming in monitoring and surveillance and the reasons behind this underperformance. The status of global fisheries is too urgent to continue ad hoc monitoring and surveillance, which keeps less profitable or less visible sectors (e.g., tuna drift gillnets) free of real regulatory or commercial pressure to improve. Targeting gillnet sectors in places such as India, is an opportunity for gains, while countries such as Yemen, require immediate and more extensive financial assistance. Although the need is particularly urgent in unobserved fisheries in developing countries, all parties to the global fishing fleet must be pressured to make smart investments and honest commitments to improve seafood sustainability.

5 Ecological risk assessment for data-deficient fisheries: approaches, principles and an alternative path

5.1 Abstract

Evaluating the risk that fishing and other human activities poses to marine biodiversity requires accurate information about both the threat and the impacted species. These data are often not available, especially for non-target species and non-industrial fishing sectors. Data limited approaches offer a range of options, such as Ecological Risk Assessment (ERA) methods. These ERA methods have been used extensively to estimate risk in data-poor contexts, often by incorporating expert knowledge with available quantitative or empirical data. However, expert or categorial scoring approaches may not have sound mathematical principles, leading to many haphazard applications of ERAs and potentially misleading or mathematically flawed results. As one example, we describe the underlying estimation of susceptibility to capture in fisheries that is used in ERAs and show how adapting the approach to a probabilistic framework, where the range of possible outcomes are expressed as expected mortality, can improve estimates of risk with varying availability and quality of data. We apply this framework to estimate expected mortality of marine mammals in Indian Ocean tuna gillnet fisheries and find that the probabilistic method better resolves the relative risk between highly susceptible species, and more explicitly conveys the uncertainty in the possible outcomes. Given the incessant shortage of adequate data in marine conservation—and environmental management contexts more broadly—risk assessments that incorporate scoring systems and expert knowledge will continue to be important tools. Continual improvement of the ERA approach will help researchers and practitioners apply available knowledge in the most rigorous way possible, leading to more accurate evaluations of risk and more informed management decisions.

5.2 Introduction

5.2.1 Decision making under uncertainty

Threats to marine species and ecosystems are expanding at a rate that outstrips our capacity to research and monitor the ocean environment (Díaz *et al.* 2019). We will never be equipped with as much data as we would like, but to delay urgent management decisions on account of data collection that is generally expensive and logistically difficult, is inconsistent with the precautionary approach (González-Laxe 2005). Thus, we must make the best possible decisions based off the information we have. Risk-based tools—where estimates are used to interpolate missing or uncertain data—have been refined from purely qualitative into semi-quantitative tools. Semi-quantitative scoring of risk elements has become common in marine and terrestrial conservation and resource management. These tools have gained traction in fisheries management, as mandate for assessment has increased from target species, to byproduct and bycatch, to threatened or protected species and even to habitats and ecological communities. One example of a semi-quantitative scoring tool that has become increasingly common is the Productivity Susceptibility Analysis (PSA) developed from Stobutzki et al (2001), which is an element in the Ecological Risk Assessment for the Effects of Fishing (Hobday et al. 2007; 2011).

5.2.2 Evolution of risk assessment in fisheries

Ecological risk assessments (ERAs) encompass a variety of methods to evaluate the vulnerability or susceptibility of a population or species to a threat, and are widely used in biodiversity management and conservation (e.g., Patrick *et al.* 2010; Arrizabalaga *et al.* 2011; Micheli *et al.* 2014). Hobday 2007 outlined a method (ERAEF) for assessing risk even when information is missing or highly uncertain, for instance, where there is limited information about a species' life history (e.g. distribution, habitat use, lifespan) or about the threat (e.g. intensity and location of fishing) (Hobday *et al.* 2007). The ERAEF is a hierarchical framework where ecological risks from fisheries on species (or habitats or communities) can be estimated at several levels of resolution, with increasing data requirements at each level. The hierarchical approach consists of a comprehensive but largely qualitative analysis of risk at Level 1, a more focused and semi-quantitative approach at Level 2, to a highly focused and fully quantitative "model-based" approach at Level 3. This hierarchy of approaches is efficient because many potential activities or hazards are screened out at Level 1, so that the more intensive and quantitative analyses at Level 2, and ultimately at Level 3, are limited to a subset of the higher risk activities associated with fishing. It also leads to rapid identification of high-risk activities, which in turn can lead to immediate remedial action (risk management response) where it may be inappropriate to delay action pending further analysis. The structure also allows improvement of existing tools and new tools to be included at each level as they are developed (e.g. Zhou *et al.* 2016).

Level 1 starts with the Scale, Intensity, and Consequence Analysis (SICA) tool to conduct a general risk screening that identifies the components of the system (e.g., target species, discarded species, habitats), and how different activities (e.g., discarding waste, discarding species, fishing with bait) could affect those components (e.g., capture and death of a species, discharged waste attracts other species) (Cotter and Lart 2011). This first step screens out activities that are judged as low impact, or components that are deemed less important or beyond the scope of management.

The next level of assessment combines information on productivity or exposure to a threat to assess potential risk to priority species, habitats, or communities in greater detail. The two main approaches to the Level 2 risk assessment are the Productivity Susceptibility Analysis (PSA) and the Sustainability Assessment for Fishing Effect (SAFE). The PSA is a semi-quantitative method that uses ordinal scales to provide an overall risk estimate of high, medium, or low. The SAFE risk outputs can be roughly compared to the PSA, although the SAFE uses a fully quantitative ratio scale measured by continuous quantities to categorize the risk from fishing mortality (*F*) relative to reference points for the mortality that a species or population could sustain (Zhou *et al.* 2016).

Variations of these ERA tools have been used extensively in fisheries for both targeted and incidentally caught species, but the semi-quantitative PSA is particularly common for non-target species and data poor contexts (Hordyk and Carruthers 2018; Duffy *et al.* 2019). The PSA combines two elements to evaluate risk to a population or species: 1) susceptibility to a threat such as fishing, which represents the likelihood that damage or mortality from the threat occurs and 2) biological productivity, meaning the life history traits that would allow the species to sustain or recover from the threat (e.g. fecundity, age of maturity) (Figure 5.1). This PSA approach uses existing data classified into categories and can be based on like-species or families or expert judgment in the rare case where information is entirely absent. The categories are coded with scores of 1, 2, or 3, representing low, medium, and high for two axes: productivity and susceptibility. The score for each axis is the geometric (multiplicative) mean of its components, and the Euclidean distance between the axes are combined into a single risk score for each element assessed (e.g. a species). The method is based on commonly available and existing data, such as calculating the percent overlap of a species' range and a fishery. Since the PSA for fisheries was developed in 2001 (Stobutzki *et al.* 2001), users have added or adjusted the parameters to fit different species or contexts, for instance, adding additional criteria to represent market value or animal behaviors such as schooling or seasonal migrations (Patrick *et al.* 2010; Hordyk and Carruthers 2018; Baillargeon *et al.* 2020).

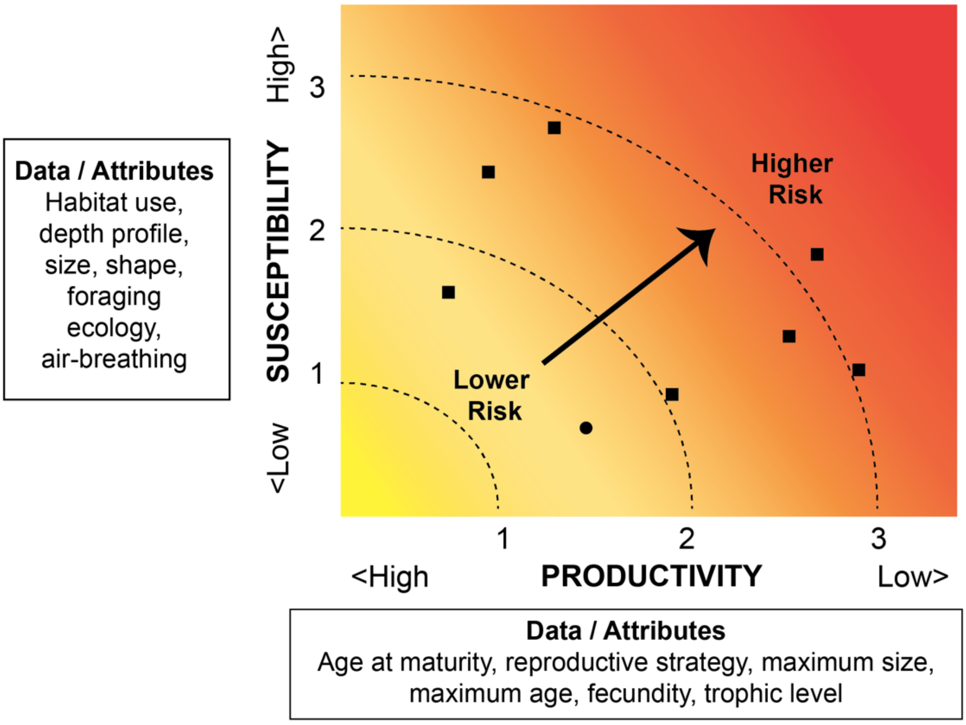

Figure 5.1: Schematic of the data and attributes used for the PSA risk calculation

Use of bins is an advantage because it overcomes the difficulties encountered with combining dissimilar quantities either within a category (e.g., age at maturity and trophic level), or between the susceptibility and productivity parameters (e.g., maximum age at maturity and spatial overlap with a fishery) to estimate the relative intrinsic rate of increase. However, a categorical scoring system is mathematically problematic because it assumes an underlying metric scale where the distance between 1-2 and 2-3 is equal. This is not necessarily true; for instance, two species scored as 2 and 3 might have more similar susceptibility to a gear compared to the distance between species in category 2 and category 1. Recent innovations for this method improved the fixed width scoring bins by allowing for continuous values scaled to fall between 1 and 3. For example, where better information on species distribution is available, a species could have a scaled score of 1.6 for overlap with a fishing gear. Interestingly, recent PSAs continue to divide the scores into fixed width bins, which washes out the precision gained by allowing continuous scaled scores (Georgeson *et al.* 2020; Lin *et al.* 2020).

An additional problem that has become apparent in the widespread adaptation of the ERA is the arbitrary combination of scoring productivity or susceptibility values (Duffy and Griffiths 2019). Susceptibility to a threat (in this case fishing) is the geometric (multiplicative) mean of the score for each parameter, for instance overlap in depth, in horizontal position, between body size and mesh size in a net. However, in some cases users have used arithmetic means (e.g. (Micheli *et al.* 2014) or have added mathematically arbitrary weights to some variables (Stobutzki *et al.* 2001; Patrick *et al.* 2010), for example, squaring the parameters for gear selectivity and horizontal overlap with fishing (e.g., Brown *et al.* 2015). The potential errors from these mathematically arbitrary assumptions increase as more variables are added (Hordyk and Carruthers 2018; Duffy and Griffiths 2019). The lack of an underlying theoretical rationale for these methods implies that equally valid but very different mathematical operations can be applied to scores, leading to very different estimates of risk for the same species and underlying data.

Although originally designed for fisheries, ERAs have been adapted to a variety of contexts, including invasive species management (Dawson *et al.* 2015), extinction risk from roadkill (Brehme *et al.* 2018), and species vulnerability to climate change (Chin *et al.* 2010; Hare *et al.* 2016). The method has evolved from integer-based expert scores across all variables to an indiscriminate mix of scores and data-derived values, which can lead to false estimates of risk and of the uncertainty of those estimates. Risk assessment tools are just one set of decision support tools for natural resource management. The ecological results are then used in cost-effectiveness or even cost-benefit analysis, and thus making sure we have transparent and unbiased estimates is critical for implanting legitimate decision-making processes. Given the increasing demand for tools to guide management in data-poor situations, how then can we improve **reliability** and **accuracy** of assessments without resorting to less **transparent** and more data hungry methods? Here we provide guidance on how to integrate different types of information—including expert scores—into calculations of risk in a mathematically robust way.

5.2.3 Definition of terms

We use catch susceptibility as used in the ERAEF PSA to demonstrate how to replace the low, medium, high scoring bins with ranked probabilities. The same logic applies to any parameter included in a PSA (e.g., market price, seasonal migrations, schooling and other behavioral responses to gear). In the ERAEF PSA (and the majority of its subsequent applications), a species' susceptibility to capture is a function of the encounter probability of the species (the horizontal and vertical overlap of the species and the gear in the water column), capture probability (conditional on encounter) for that species (e.g. whether the species is the right size and shape to become entangled in a net or whether it would be attracted to bait, often called gear selectivity in fisheries), and the severity of the outcome (e.g. whether an air-breathing animal would drown if entangled):

$$S(Capture) = \sqrt[4]{A \times E \times S \times PCM}$$

Where the encounterability is divided into two separate parameters: availability (*A*, the horizontal overlap) and encounterability (*E*, the vertical overlap), *S* is gear selectivity, and *PCM* is post-capture mortality, which can also be called post-release survival or potential lethal encounter.

In the original ERAEF PSA and most subsequent iterations, the four parameters are scored 1, 2, or 3 and multiplied to get an overall susceptibility score, which is then rescaled to the same fixed width 1-3 bins. The same approach is applied to the biological parameters (e.g., length at maturity, fecundity, growth rate) to generate a scaled score for productivity. The overall vulnerability score is calculated from the Euclidean distance between the productivity and susceptibility axes. These overall scores are usually divided into thirds to make subjective comparisons, e.g. labeling a score >3.14 as "very high" susceptibility.

5.3 Proposed framework

5.3.1 Defining risk

Risk means different things in different contexts, and it should be clear what the objective of the ERA is and how the components of risk are translated into mathematical functions. The key concept in moving to a risk framework is to define risk correctly, so that it can be broken down into appropriate parameters. Risk—as defined using probability theory—is the expectation of how likely an event is to occur and the severity of the outcome:

$$Expected\ Risk\ =\ Probability\ of\ Event\ \times\ Severity\ of\ event$$

Previous applications of the ERAEF PSA do not explicitly frame risk in this probabilistic way, so the resulting value (e.g. a catch susceptibility score) is a unitless number without any context of an event or outcome. We propose an improvement, whereby the ranked probability approach to the ERA avoids unitless scores by framing susceptibility in terms of expected mortality, where expected mortality is a function of the probability of the event occurring and the severity of the outcome (the lethality) conditional on the event occurring (Figure 5.2). The expected mortality can then be summed across the appropriate scale, such as a population or an area.

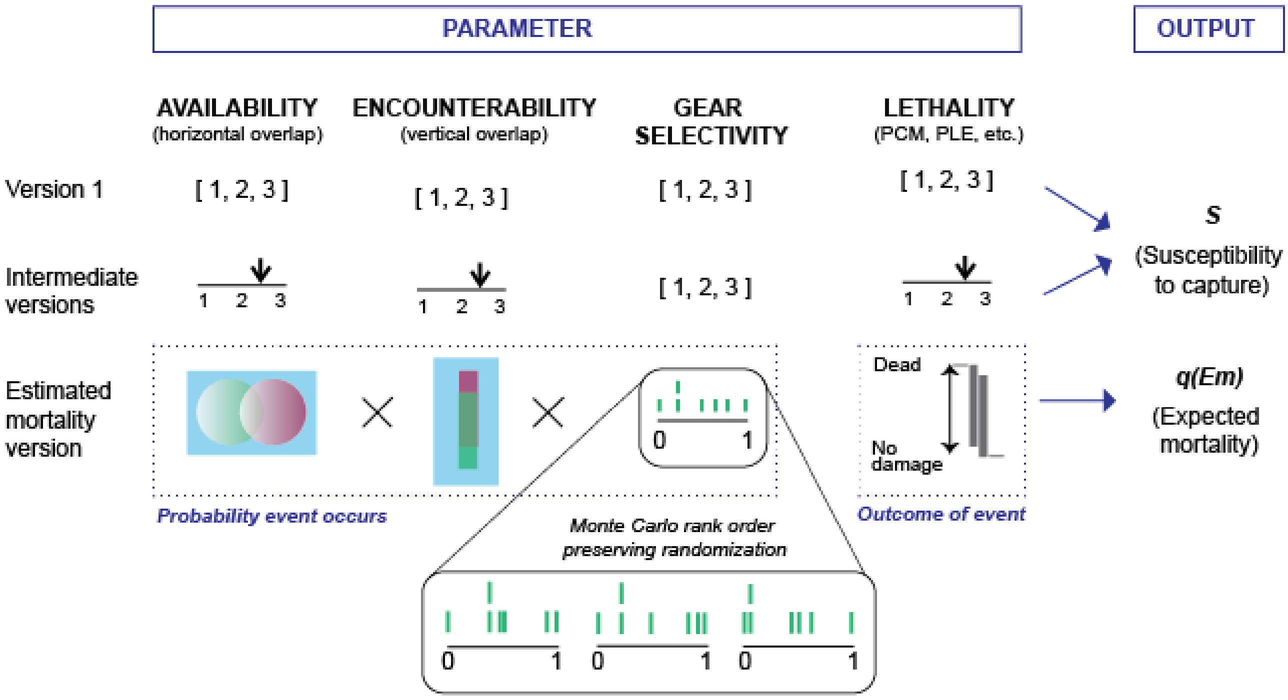

Figure 5.2: Schematic of decision structure evolution from categorical scores to the ranked probability version

5.3.2 Guiding Principles

We propose five main principles that emerge from framing the ERA methodology as a calculation of expected mortality. These principles apply not only to the context of susceptibility of a species to catch in fishing gears but to any incarnation of ERA, risk-based scoring systems, or decision tool for data deficient situations.

1. The system should be built on a well-defined mathematical framework, with consistent and meaningful operations. Arbitrary addition, multiplication, or other operations can change the meaning of the parameters and the resulting estimate of risk.

2. Do not assume more than is known. For example, many ERAs assign numbers to categories and ask experts to score different parameters of the risk equation (e.g. 1, 2, 3 for low, medium, and high selectivity of fishing gear for a species). However, scores like these imply more than an order, they also imply a distance between levels, and in this case specify that this distance is constant across the scale. Such assumptions should be explicit, and only included where supported.

3. There should be a clear order of goodness of information. Known values with no error are optimal, but extremely rare. The second-best option is a value derived from data. Where values are not known and adequate data are not available, the next best option is expert

judgement of a probability (e.g. the probability that a species encountering fishing gear will be entangled). Where experts cannot give reliable probability estimates, they can instead judge order (e.g. species A is more likely to be entangled than species B). The worst-case scenario is the information is unknown. In this case, bounding the information between 0 and 1 allows for quantification of uncertainty for each parameter and preservation of uncertainty through the subsequent mathematical operations.

4. Any assumptions should be conservative with respect to the outcome (this precautionary scoring is in the ERAs as they stand). For example, where no information is available, assume a gear is selective for an animal and the probability of a lethal encounter is high (in this case, set to 1).

5. Frame the risk as the expected value of an impact, probability of an event occurring weighted (multiplied) by the outcome of the event. The outcome may have different units than the probability of the event. The outcome may be bounded, for instance: no effect (0, 0), sublethal (0, dead], potentially lethal [0,1), or lethal (dead, dead), while the probability of capture in gear is the product of the probability of each dimension of capturability, where:

$$P(Capture) = P(horizontal\ encounter) \times P(vertical\ encounter) \times P(gear\ selectivity)$$

Using this framing, operations such as summing across locations, vessels, species, and other mathematical operations to develop decision tools (e.g. optimization, cost effectiveness) are well defined. For example, the quantitative sustainability assessment for fishing effects (SAFE) method estimates fishing mortality, and can therefore be summed (Zhou *et al.* 2016).

5.4 Worked example

5.4.1 Background

Score-based ERAs have been used extensively to estimate the impacts of fishing on non-target species, which often lack consistent monitoring and reporting. The impacts of Indian Ocean gillnet fisheries on marine mammals is an example of a context where data are severely limited for both the species and the fishing effort. The Indian Ocean is recognized as a global hotspot for megafauna diversity, but basic information about abundance and distribution is lacking for many species (Selig *et al.* 2014). The Indian Ocean harbors many cetacean species that are threatened with extinction and are considered to be extremely vulnerable to fishing impacts, particularly from gillnet sectors (Kiszka 2012; Anderson *et al.* 2020). Gillnets—a broad category of relatively cheap, simple fishing nets that can be anchored or drifting—have emerged as a major concern because they are associated with high mortality of marine mammals globally, whereas there is much greater variability in mortality from other common fishing gears depending on the species and location (Lewison *et al.* 2004; Northridge *et al.* 2017). Gillnets are common in developing countries and coastal waters around the world and have expanded into a major fishing sector in the Indian Ocean, which has large, offshore "driftnet" sectors in addition to more traditional inshore nets (Aranda 2017). Although the driftnet fisheries are essentially industrial-scale, they are categorized as "artisanal" and therefore, countries are not required to report information about where driftnet fisheries operate or how many vessels are involved (Roberson *et al.* 2019). There is even less reporting of TEP bycatch in the gillnet sectors, and where fisheries interactions are reported they are often not recorded to the species level (Aranda 2017). This makes it extremely difficult to quantify risk across a species or fishery, especially for rare, cryptic, or poorly known species like deep-diving beaked whales.

5.4.2 Methods

We use the example of marine mammals and Indian Ocean gillnet fisheries to demonstrate the utility of the ranked probability ERA methodology for assessing risk in data-poor scenarios, which are typical of non-industrial fisheries and non-target species. We compare the results of the ranked probability approach to the categorical scoring method. For this example, we use three types of probabilities for the horizontal and vertical encounterability and selectivity parameters, demonstrating different levels of information.

5.4.2.1 Probabilistic ERA method

For the horizontal dimension of encounterability, we use two spatially explicit models as proxies for density of animals and fishing gear. First, we selected 49 marine mammal distribution maps from the AquaMaps database, which has generated maps of probability of occurrence in 0.5 degree cells using models based on species-specific envelopes of environmental preference (Kaschner *et al.* 2016). The environmental envelopes include variables such as temperature, depth, and salinity, and are based on occurrence records and published databases.

To estimate density of driftnet fishing boats, we used the most recent year available (2015) from a model of fishing effort that disaggregates data by country, gear type, and engine power to create a spatially-explicit map of fishing power (Rousseau *et al.* 2019). This model builds from reconstructed catch data and incorporates information on each country's fleet across different gear types, including vessel characteristics, major ports, and distance from the coast to estimate effort in terms of engine power and fishing days per year. In this example, we are interested in the larger drift gillnets used to target tuna and tuna-like species in the Indian Ocean Tuna Commission (IOTC) management area. However, these nets can be used in many different configurations to target a wide range of species in addition to tuna and tuna-like species (Yousuf *et al.* 2009). Therefore, we removed the two lowest power categories because data for small vessels are the least complete (Rousseau *et al.* 2019), and these power categories are likely to represent smaller inshore nets.

Due to discrepancies in gillnet catch reporting across countries (partly due to the wide variety of gears included in this category), driftnet effort was extremely skewed and concentrated in a few coastal cells. Assuming that effort will not vary dramatically between neighboring cells, we replaced outlier cells that were more than two standard deviations from the mean of their neighboring cells with the neighbors' mean. We then scaled the spatially smoothed fishing effort from 0-1, where the maximum value for any one cell is 1 but there is no constraint on the sum of values across all cells (as opposed to normalizing). This gives a relative probability of driftnet fishing in each cell.

Assuming the scaled fishing effort and the AquaMaps probability of occurrence are proxies for density of fishing boats and density of animals, the product gives a value for the probability of horizontal encounterability in each grid cell.

$$P(horizontal\ encounter) = P(species\ occurrence) \times P(fishing\ presence)$$

In this example, horizontal encounterability represents the best information of the susceptibility parameters because the probability accounts for density within the overlap.

The vertical dimension of encounterability is the probability that the fishing gear and the species overlap in the water column. Driftnets are set at or slightly below the surface, and typically have a hanging depth <25m (Stequert and Marsac 1989; Novianto *et al.* 2016; Khan 2017). Here we use 50m as a conservative maximum depth. For species' depth ranges, we used the Maximum Preferred Depth from the AquaMaps model and a minimum depth of zero, as all cetaceans are air-breathing. Encounterability in the water column is the percent overlap of the species and the gear:

$$P(vertical\ encounter) = \frac{overlapping\ depth\ range}{species\ depth\ range}$$

Here we represent encounterability with a simple percent overlap, which assumes that both the fishing gear and the species are uniformly distributed throughout their depth range. In practice, additional data or expert knowledge could be incorporated to create depth profiles for fishing gears and species. For example, deep-diving beaked whales are known to spend more time at the deeper limits of their depth ranges compared to small dolphins and porpoises that congregate near the surface.

The third parameter in the susceptibility equation is the gear selectivity. In this example, we have the least information available for gear selectivity. In this case, we rank species based on empirical data (or known selectivity for physiologically similar species) and then randomly generate probabilities for each species consistent with their rank. In cases where there are many species and insufficient information for ranking individual species—such as our Indian Ocean example—the best option is to group like species and generate probabilities for each group. This is equivalent to ranking individual species and allowing ties. To demonstrate this method, we divided the 49 cetaceans into five groups based on physiological characteristics that affect their propensity for entanglement in gillnets. We then ordered the ranks based on available empirical data. To capture the uncertainty, we used a Monte Carlo process with 1000 iterations to randomly generate probabilities for the species groups, allowing ties and preserving their order (Figure 5.2).

The final component of the susceptibility calculation is the severity of the outcome conditional on the event occurring (the species is entangled in the gear). Previous studies have used a number of different terms for the outcome, including Post-Capture Mortality, Post-Release Mortality, and Potential for Lethal Encounter (Cortés *et al.* 2010; Breen *et al.* 2017; Duffy *et al.* 2019; Clavareau *et al.* 2020). Most studies use the 1-3 scores for this parameter because the effects of capture on escaped or released animals are poorly known. Instead of discrete scores, we propose quantitative intervals with overlapping ranges of possible outcomes to better accommodate the high uncertainty in the mortality parameter, especially where the behavior of fishermen is unknown if the animal were to be landed. We use four bounded quantitative intervals for lethality (Table 5.1).

Table 5.1: Intervals and descriptions of possible outcomes (lethality) if an animal is entangled in gear

Category	**Interval**	**Description**
No damage	[0,0]	Species escapes without damage that decreases fitness
Sublethal	[0,1)	Species may escape unharmed, may suffer minor to serious damage, but will not be landed
Potentially lethal	(0,1]	Species may escape with minor to serious damage, or could be landed
Lethal	[1,1]	Species is a target or like-target species and will likely be landed

The product of the first three parameters (horizontal and vertical encounterability and gear selectivity) gives the probability an animal will be entangled in fishing gear in a given cell, and the lethality intervals give a range of outcomes for that event (e.g., animal will escape without damage, animal will die). We used the mean catchability value to summarize each species' overall catchability across all cells in the IOTC area, irrespective of their range size. To explore which species are most exposed to fishing across their range, we calculated the proportion of each species range within the IOTC area that overlaps with driftnet fishing. We used a cut-off of 1 kWday/year to exclude cells with negligible fishing effort but counted species as present in any cell with probability of occurrence > 0 (the minimum possible probability is 0.01).

5.4.2.2 Comparison to categorical score approach

We repeated the analysis for the 49 species using the categorical scores method from Hobday et al. 2007. Here, availability (horizontal encounterability) is based on presence-absence of species and fishing, instead of a density distribution. We converted the smoothed and scaled 2015 fishing effort (excluding cells with effort < 1 kWday/year) to presence-absence in each cell. For species presence, we used a threshold of 0.5 as a relatively conservative probability of occurrence. Previous studies using AquaMaps distributions have found that species distributions are robust to different thresholds across a large area or a species' entire range (Kaschner *et al.* 2011; Klein *et al.* 2015; Jones *et al.* 2018a). The availability scores are not particularly sensitive to these presence-absence thresholds because, following the categorical approach, each value is binned as Low (< 10% overlap), Medium (10 – 30% overlap), or High Risk (> 30% overlap), which correspond to values to 1, 2, and 3, respectively. Encounterability (vertical overlap) is scored according to the same overlap thresholds.

For gear selectivity, previous ERA iterations used a rough selectivity rubric with four categories based off the animal's length at maturity relative to stretched mesh size (or hook size and affinity towards bait, in the case of line fishing). This rubric was designed for fish species; for cetaceans and gillnets (and for many TEP species), all species are substantially larger than the mesh size, which ranges from less than 1 cm for smaller inshore gillnets to about 20cm for the pelagic driftnets used to target tuna and tuna-like species (Aranda 2017; Hosseini *et al.* 2017). There are no available studies with selectivity scores for gillnets and all cetaceans on our list, so we used selectivity scores from previous PSAs and scored the five groups according to the most common score for species within that group (Brown *et al.* 2013, 2015; Breen *et al.* 2017).

The fourth parameter is lethality, which is also scored as Low (evidence of post-capture release and survival), Medium (species released alive), or High (species usually retained or discarded dead) (Cotter and Lart 2011). The geometric mean of these four parameters gives a score for susceptibility to capture. For a full PSA, which incorporates biological parameters for a species' resilience to fishing mortality, the overall vulnerability score is calculated from the Euclidean distance between the productivity and susceptibility axes. The space is divided into thirds to categorize the scores as Low, Medium, or High risk. Here, we are working with only one axis (susceptibility to capture), so scores from 1-1.66 are Low, 1.66 - 2.33 are Medium, and <2.33 are High risk.

5.4.3 Results and Discussion

We compared two ERA approaches for estimating cetaceans' risk of capture in gillnets. The output of the categorical score approach presented in Hobday et al. 2007 is a single unitless score for susceptibility to catch, whereas the output of our probabilistic approach is expressed as a mean probability of capture and an interval of possible outcomes across all cells in the analysis. Although expressed in different terms, both approaches are essentially estimating the potential damage or death, with the probabilistic approach more explicitly expressing this outcome at the level of an individual animal.

Overall, the relative species rankings were similar between the two approaches, with the top 10 species for mean probability of capture all scored as High catch susceptibility, except for *Feresa attenuata* (Delphinidae), which was categorized as Medium (Figure 5.3). Conversely, most of the High susceptibility species were still ranked high for probability of capture, except for *Orcaella heinsohni* (Delphinidae), which scored 3 (tying for highest catch susceptibility) but was only ranked 22 for mean probability of capture. Compared to the probabilistic approach, the categorical bins resulted in more species categorized as higher susceptibility to capture, with 14 High and 30 Medium susceptibility compared to five Low susceptibly scores. The categorical bins approach also resulted in a lot of ties; eight species tied for highest catch susceptibility (a score of 3), and there were only 11 different catch susceptibility scores so 46 of the 49 species are ranked in the top ten. In contrast, with the probabilistic approach there were no ties, and only two species fell above the top third (High), two as Medium, and the remaining 45 species as Low probability of capture.

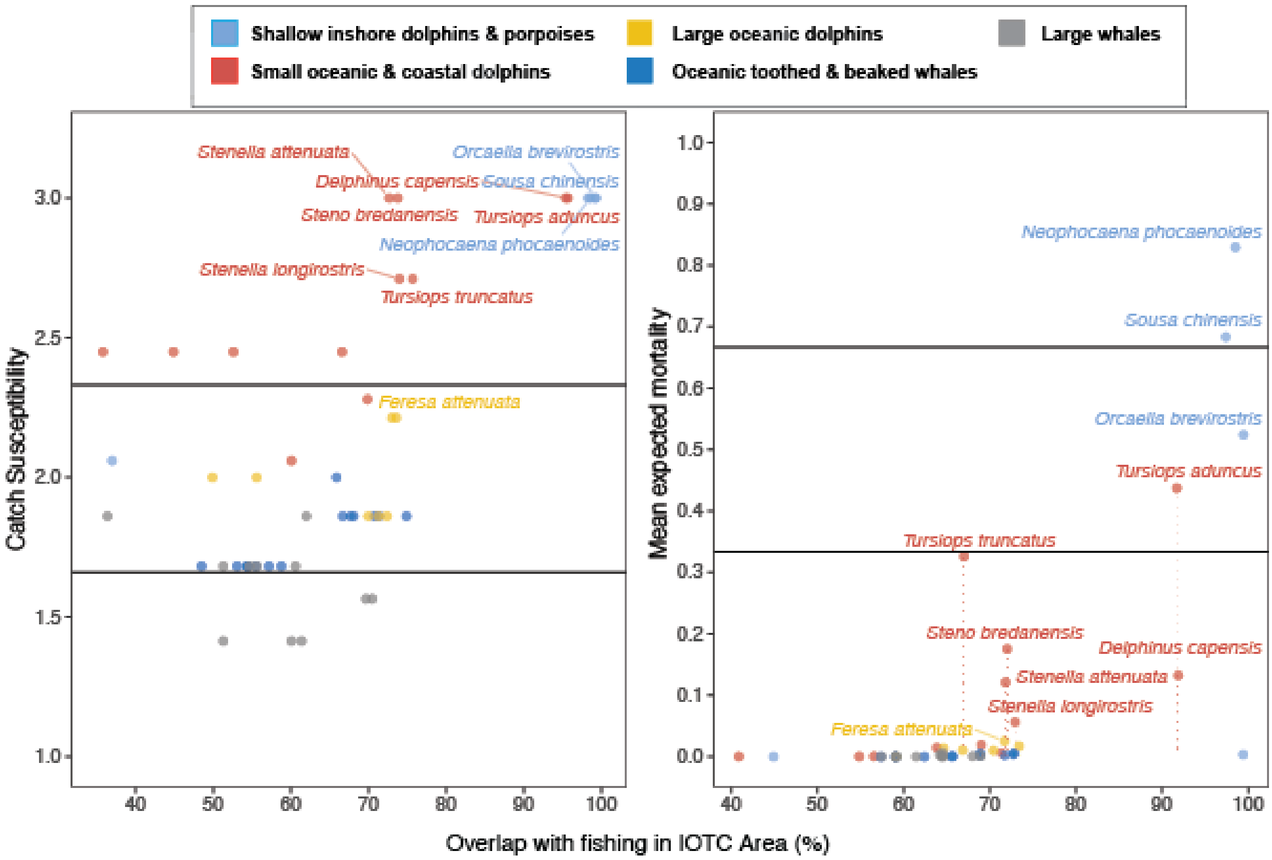

Figure 5.3 Catch susceptibility scores (Left) and mean expected mortality across all cells (Right) for 49 marine mammals occurring in the IOTC management area. Dotted lines show the lethality intervals (the range of possible outcomes if the animal is caught). Labels show species with the top 10 mean catchability scores.

More important than the Low, Medium, and High bins (which are inherently problematic for both approaches), the probabilistic approach better resolves the rankings for high-risk species—the species management should be most concerned about—and shows the distance between the probabilities of capture (Figure 5.3). The categorical approach scores four of the five shallow inshore dolphins and porpoises as 3 (the highest possible score from the four parameters used in this example), and 10 of the 12 small oceanic and coastal dolphins also score 3 or above the 2.66 cut-off for the High susceptibility category. Mathematically, the equal-distance bins mean that the eight species with scores of 3 are about twice as susceptible to being captured as the two species scoring 1.56 (*Balaenoptera edeni* and *B. brydei*, Balaenopteridae). In contrast, the new approach showed that most species had low catchability compared to three shallow inshore dolphins and porpoises (*Neophocaena phocaenoides,* Phocaenoidae, *Sousa chinensis,* Delphinidae, and *Orcaella brevirostris,* Delphinidae). Following these three species is a cluster of six small oceanic and coastal dolphin species (e.g. *Tursiops spp.,* Delphinidae), which have much lower relative probabilities of capture and also have a wider uncertainty interval for the outcome of capture (they are more likely to survive entanglement) compared to the shallow inshore species (Figure 5.3). If an air-breathing animal is entangled, gillnets tend to be more lethal than many other fishing gears (e.g. purse seines), even for large species (Johnson *et al.* 2005; Senko *et al.* 2014). All the cetacean species in our analysis are categorized as lethal or potentially lethal, except for the blue and fin whales (*Balaenoptera musculus* and *B. physalus,* Balaenopteridae), which do entangle in gear but these interactions were categorized as sublethal (Ramp *et al.* 2021). Even for a generally lethal gear, the lethality interval helps resolve differences in risk between species with similar probabilities of entanglement. The categorical score approach indicates that all the species with a susceptibility score of 3 have an equally certain outcome, whereas the probabilistic approach shows a range of possible outcomes.

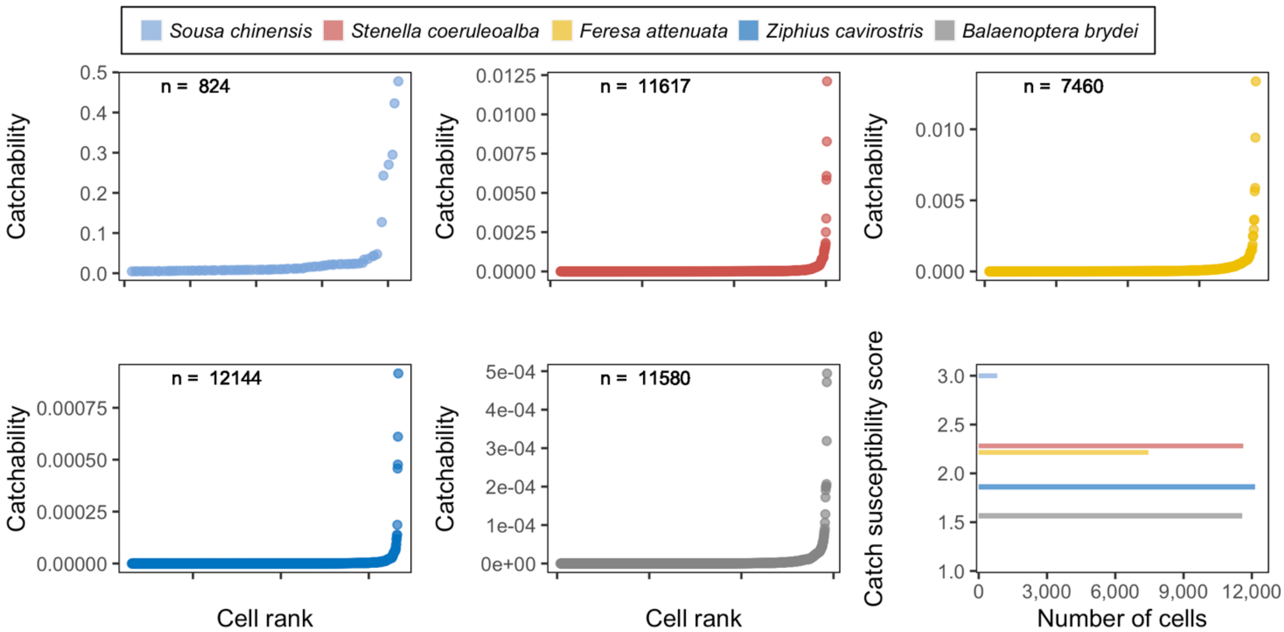

Figure 5.4: Catchability per cell for one species from each of the five species groups, compared to their catch susceptibility scores (Panel 6). The top 10% of cells for each species are shown in Panels 1-5. Cells are ordered by catchability score. Panels 1-5 are labeled with the number of cells in the IOTC area where the species has a catchability > 0. Panel 6 shows all cells with a catchability score > 0 for each species.

The categorical score approach assumes that all cells where a species overlaps with fishing effort have the same catch susceptibility. While the relative catch susceptibility scores indicate which species are at greatest risk—and therefore most in need of management interventions—there is no spatial resolution. We found that driftnet fishing effort in the Indian Ocean is concentrated in clusters of coastal cells. A comparison of probability of capture across five species' ranges (one from each species group) showed that all the species have mostly low-risk cells and the majority of their expected capture is concentrated in a small number of cells (Figure 5.4). This pattern is consistent across species with small range sizes and overall high catchability (e.g. *S. chinensis*), species with large ranges and overall low catchability (e.g. *B. brydei*), as well as across species groups (Figure 5.4, Figure 5.5).

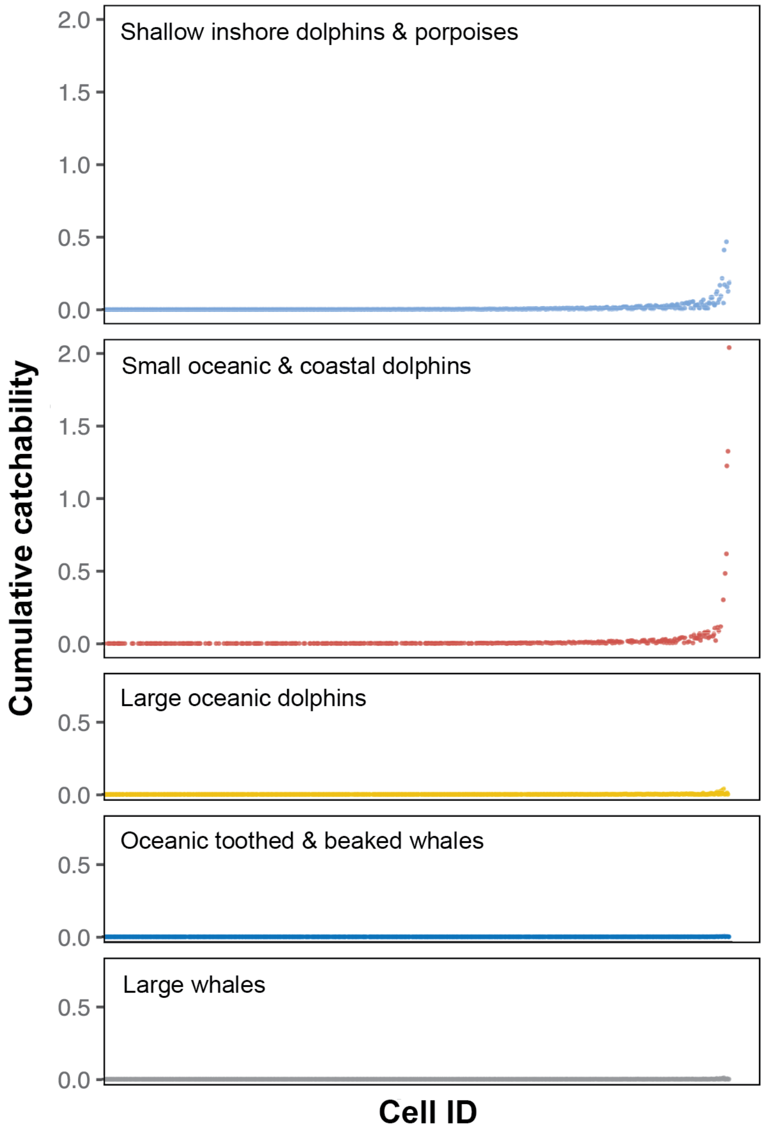

Figure 5.5: Mean catchability probabilities by cell for the five species groups. Size of dot corresponds to cumulative catchability score for each species group. Cells are ordered by ascending cumulative catchability across all species groups. Only cells in the top 5% of cumulative catchability values are shown.

In addition to showing that risk is concentrated in a small proportion of the total management area for each species and species group, we found that it is the same cells that tend to have the highest cumulative catchability scores across all 49 species (Figure 5.5). For small oceanic and coastal dolphins and shallow inshore dolphins and porpoises, the difference between low and high catchability cells is much more pronounced compared to the larger cetaceans (Figure S3.1). Thus, the spatially explicit probabilistic approach shows which species most urgently require management as well as the geographic areas where management interventions should be targeted. This means that an opportunity emerges for area-based measures such as MPAs or fishing closures that target high risk areas. The distribution of entanglement risk across cells also indicates where area-based measures are less likely to be effective in reducing species mortality; for instance, large cetaceans require management measures across their entire range, which will likely have minimal impact on reducing total number of deaths because their probability of entanglement is low. Of course, for some species preventing even a few deaths might be worth extensive and expensive management measures, which is where the biological resilience component of the PSA would help frame risk and guide management decisions.

5.5 Conclusion

The PSA is a semi-quantitative tool aimed at estimations of risk in data-poor contexts; as such, there are limits to how much the underlying quantitative assumptions can be improved (Hordyk and Carruthers 2018). Other tools exist for quantifying risk of threats such as fishing but, in many cases—such as the Indian Ocean example we demonstrate here—sufficient data are not available. In these cases, the PSA remains the most widely used option for quantifying and comparing risk. Missing or inaccurate input data will of course lead to less accurate and more uncertain results (although the direction of bias is fixed for missing data). The optimal scenario is that empirical data are available for all the parameters (horizontal and vertical encounterability and gear selectivity for the catchability example). This is rarely the case, so we demonstrate an alternative probabilistic ERA method with descending data quality for the catchability parameters. The accuracy of the probability estimate can be improved by pushing more parameters towards a location-specific density distribution; for example, incorporating information about how fishing gear and species are distributed vertically in the water column in different areas. Here, our aim was not to provide a comprehensive risk assessment for cetaceans and Indian Ocean driftnet fisheries, but to demonstrate how to use a probabilistic framework to make a more mathematically rigorous assessment of risk from limited data.

A key benefit of changing from a categorical score to a probabilistic approach is that the uncertainty of the outcome is quantified. With the score-based method, all species that score a 3 are equally susceptible to catch. In contrast, the probabilistic method separates the likelihood of the event occurring from the severity of the possible outcomes. If two species are equally likely to be entangled but have a different range of outcomes (e.g., one is more likely to escape the gear whereas the other is usually landed dead), this suggests different management interventions. For sublethal or potentially lethal encounters, measures aimed at decreasing post-capture mortality (such as safe handling and release practices) are likely to be helpful. For species that are often dead when landed, reducing mortality will depend on reducing the likelihood of entanglement (e.g. through gear modifications or temporal or spatial controls on fishing activity). In this example, driftnets are a particularly lethal gear for airbreathing species so for most cetaceans, safe release practices will have limited effectiveness. In other situations, such as comparing risk across multiple gears or taxa (e.g. cetaceans compared to chondrichthyans or purse seines compared to driftnets), separating the uncertainties of the possible outcomes from the probability of the event occurring becomes increasingly useful to guide management towards more informed decisions.

A second important benefit of our proposed ERA adaptation is that information for each parameter is better preserved through the equation. With the categorical score approach, some precision is lost each time a number is categorized into a fixed-width bin. For example, Species A with 31% horizontal and vertical overlaps with fishing gear would score a 3, same as Species B with 100% overlaps. With the probabilistic method, each parameter is multiplied (either across the species or for each cell, if the data are spatially explicit), so the probabilities of encounter would be 9.6% for Species A and 100% for Species B. Instead of forcing the variables into categorical bins, expressing the risk as a probability and an outcome provides a visualization of relative risk, which is more meaningful and standardized than a unitless score. It also shows more explicitly which parameters are driving the risk. For example, the small cetaceans with the highest probabilities of entanglement all have very narrow depth ranges that overlap perfectly with driftnets. This suggests that interventions such as setting the nets a few meters below the surface might substantially lower the overall probability of entanglement. Based on preliminary trials in Pakistan's driftnet sector, this intervention does result in lower catch rates of small cetaceans (Kiszka *et al.* In Review). This probabilistic format still allows for additional indicators of uncertainty. For example, if data are highly variable across species, then experts could add bounded intervals for data quality to indicate confidence intervals for the possible outcomes.

Although the PSA is a tool specifically designed for fisheries, the need to estimate risk while armed with only limited data is a universal problem for natural resource management, and even beyond the field of ecology. Score-based approaches are common but have led to haphazard applications (such as the evolution of the PSA ERAEF for fisheries), which can result in inaccurate estimates of risk. We illustrate how the same limited data can be used in a probabilistic instead of a score-based framework to estimate and compare risk, express the uncertainty of outcomes, and avoid making mathematically problematic assumptions. Given the current climate of rapidly changing ecosystems and evolving threats, it is crucial that we use available data rigorously and effectively to improve management outcomes.

6 Spatially explicit risk assessment for marine megafauna and Indian Ocean tuna fisheries

6.1 Abstract

Bycatch is one of the most significant threats to marine megafauna at the global scale. However, the magnitude and spatial patterns of bycatch are still poorly understood in certain regions where monitoring has been very limited, but where fisheries are expanding. The Indian Ocean is an important region for tuna fisheries, and scattered information suggests major bycatch issues involving marine megafauna. Although catch and bycatch data are relatively well documented in other regions for industrial tuna fisheries (primarily longlines and purse seines), recent estimates suggest that 35% of the catch volume in Indian Ocean tuna fisheries comes from drift gillnets, which are cheap, simple to operate, and pose a significant threat to megafauna species globally. Gillnets are poorly monitored and largely unregulated in the Indian Ocean. Here, we propose a risk assessment framework designed for data-poor contexts to present the first spatially explicit estimates of bycatch risk of sea turtles, elasmobranchs and cetaceans in Indian Ocean tuna fisheries (purse seines, longlines and driftnets). We found substantial overlap of high-risk areas across the three gears in some areas (e.g., western Indonesia), indicating potential opportunity for multi-taxa benefits by concentrating management efforts in particular coastal regions. Expected mortality in driftnets is high across the vast majority of coastal waters in the northern Indian Ocean, including in countries that have had very little engagement with regional management bodies (e.g., Myanmar and Bangladesh). In addition to species known to occur in tuna gears, we find high expected mortality from multiple gear types for many poorly known elasmobranchs that do not fall under any existing conservation and management measures. Our results show that existing bycatch mitigation measures, which focus on safe-release practices, are unlikely to be effective in reducing the substantial cumulative fishing impacts on threatened and data-poor species.

6.2 Introduction

Fishing, either targeted or incidental, is the primary threat directly driving population declines and extinction risk for many species of cetaceans, sea turtles, and elasmobranchs (Lewison *et al.* 2004; Costello *et al.* 2010; Brownell *et al.* 2019; Ripple *et al.* 2019). The risk that fishing poses varies across species, locations, and gear types, but gillnets stand out amongst the common fishing gears because they are associated with high mortality per unit of fishing effort for all three taxa globally (Lewison *et al.* 2004; Read *et al.* 2006; Reeves *et al.* 2013). Gillnets are a broad category of relatively cheap, simple to operate gears that can be anchored or drifting and are increasingly common in the coastal and continental shelf waters in developing countries (Northridge *et al.* 2017). Gillnets are the primary cause of extinction of the baiji (*Lipotes vexilifer*) and the possible imminent extinction of the vaquita (*Phocoena sinus*), and the most significant and increasing threat to a diversity of endangered marine mammals, sea turtles, elasmobranchs, and seabirds (Reeves *et al.* 2013; Lewison *et al.* 2014; Jabado *et al.* 2018; Brownell *et al.* 2019).

Tuna fisheries are some of the world's most valuable fisheries, with an annual landed value of US$12.2 billion, which comes mostly from industrial purse seine and longline sectors (Rogers *et al.* 2016). Tuna from the Indian Ocean account for 20% of the global commercial tuna catch (WWF 2020). This region is unique amongst the world's tuna fisheries because of the large gillnet sectors, especially the expansion of large pelagic gillnets ("driftnets") in addition to more traditional inshore nets (Temple *et al.* 2018). In the Indian Ocean, gillnets comprise an estimated 35% of the region's tuna catch, exceeding the catch volumes of the industrial purse seine and long line sectors (Aranda 2017). Gillnet vessels target a wide range of species in addition to the 16 tuna and tuna-like species that fall under the mandate of the Indian Ocean Tuna Commission (IOTC), and are increasingly targeting elasmobranchs in response to growing demand (Jabado *et al.* 2018). Countries are required to report information about some fishing gears to the IOTC but not about where gillnet fisheries operate or how many vessels are involved (Roberson *et al.* 2019).

Recently, gillnets (and driftnets in particular) have emerged as a primary concern amongst scientists and managers, with one report estimating that 100,000 marine mammals are caught annually in Indian Ocean tuna fisheries (Anderson *et al.* 2020). However, there is limited information available about fishing impacts on the region's megafauna (Clarke *et al.* 2014; Lewison *et al.* 2014; Garcia and Herrera 2019; Anderson *et al.* 2020), and the many loopholes in the existing regulatory framework result in very incomplete catch monitoring of sea turtles, mammals, and elasmobranchs (WWF 2020). A comparative study of ecosystem-based management approaches—including bycatch management—rated the IOTC as the worst performing Regional Fisheries Management

Organization (RFMO) for tropical tuna (Juan-Jordá *et al.* 2018). The IOTC faces considerable challenges in managing 31 contracting Parties in addition to massive distant water fleets from Europe and Asia, and compared to the other four tuna RFMOs, it has the most recently developed fisheries, countries with the lowest average per capita GDP, high economic dependency on tuna fisheries, the smallest vessels, and the most vessels (Pons *et al.* 2018; Sinan and Bailey 2020).

Previous research shows that fishing—both incidental and targeted—is a primary direct threat to megafauna in the Indian Ocean, including cetacean species (Elwen *et al.* 2011; Temple *et al.* 2018), sea turtles (Bourjea *et al.* 2008; Wallace *et al.* 2013; Williams *et al.* 2018), and elasmobranchs (Dulvy *et al.* 2014; Jabado *et al.* 2018), but data are lacking for many species, geographic areas, and fishing sectors. Bycatch rates vary across regions due to different environmental conditions, species abundances, and fishing effort dynamics, even for the same species and fishing gear, which means trends from one ocean or region may not be representative of another area (Clarke *et al.* 2014; Lewison *et al.* 2014).

In the Indian Ocean, available data suggest the most common longline bycatch species are pelagic and oceanic sharks (especially *Carcharhinus falciformis, Prionace glauca, Alopias* spp*.,* and *Pteroplatytrygon violacea,* as well as *Carcharhinus longimanus, Sphyrna spp.* and *Pseudocarcharias kamoharai*), relatively high catch rates for sea turtles (especially *Lepidochelys olivacea*), and interactions with toothed whales (especially *Grampeus griseus, Pseudorca crassidens,* and *Globicephala macrorhynchus*) (Huang and Liu 2010; Wallace *et al.* 2010; Clarke *et al.* 2014; Murua *et al.* 2018; Garcia and Herrera 2019). The purse seine fleets are known to have the lowest bycatch rates per unit of fishing effort (especially for cetaceans) and fewer species that are caught in large numbers, with bycatch dominated by *C. falciformis*, as well as *P. glauca*, *C. longimanus,* and *Isurus ocyrinchus* (Escalle *et al.* 2015; Murua *et al.* 2018; Clavareau *et al.* 2020). A wide variety of species have been reported in drift gillnets in the Indian Ocean, including notable catches of sea turtles, small and medium dolphins (especially *Stenella spp., Tursiops truncatus,* and *Delphinus delphis*), *Rhincodon typus*, and pelagic rays (e.g. Myliobatidae). Sea turtles are considered vulnerable to capture in all three gears but have lower mortality in purse seines compared to longlines and gillnets (Williams *et al.* 2018). Many oceanic and pelagic elasmobranchs that are common in long lines or purse seines are also frequently caught in driftnets, notably *Isurus spp., C. falciformis, Alopias* spp*.,* and *P. violacea* (Moazzam 2012; Garcia and Herrera 2019). However, the majority of the information about driftnets comes from the Pakistani fleet. The available reports and studies for the Indian Ocean all note the lack of quality data for megafauna bycatch relative to other regions (for all gear types), and there are many contradictory reports. For example, no shortfin makos were reported by purse seines fleets in the IOTC data (Garcia and

Herrera 2019), compared to substantial shortfin mako catch reported in a study of the Spanish purse seine fleet operating in the Indian Ocean (Clavareau *et al.* 2020).

Evaluating the risk that fishing poses to marine biodiversity requires accurate information about both the threat and the impacted species. Data limited approaches offer a range of options, such as Ecological Risk Assessment (ERA) methods, which have been used extensively to estimate risk in these data-poor contexts, often by incorporating expert knowledge with available quantitative or empirical data (Hobday *et al.* 2007; Zhou *et al.* 2013, 2016; Georgeson *et al.* 2020). Productivity susceptibility analyses—a type of ERA that compares life history characteristics and susceptibility to fisheries catch— have been widely used to estimate potential impact from fisheries for data-poor species (Arrizabalaga *et al.* 2011; Moore *et al.* 2013; Murua *et al.* 2018). Many ERA methods are based wholly or partially on categorical scores (e.g., low, medium, or high overlap with fishing), which is useful in cases with missing or highly uncertain information. However, methods that use categorial scoring may not have sound mathematical principles, leading to many haphazard applications of ERAs and potentially misleading or mathematically flawed results (Hordyk and Carruthers 2018; Baillargeon *et al.* 2020). Here, we use a new adaptation of a semi-quantitative ERA method (described in Hobday *et al.* 2007, 2011) that uses ranked probabilities instead of categorical scores to improve estimates of risk and uncertainty.

Of the many species reportedly caught in tuna fisheries and in large-scale fisheries more broadly, relatively few are actively monitored and managed by fisheries agencies (Costello *et al.* 2012; Ricard *et al.* 2012). Usually, species interact with multiple fisheries in one area or across their range, and these cumulative impacts are even more difficult to detect and monitor (Riskas *et al.* 2016). In general, multi-taxa or multi-gear studies of bycatch species are rare or lack a spatial component, and this gap is particularly glaring for the Indian Ocean (Lewison *et al.* 2014). Our goal in this study was three-fold: We 1) estimate the magnitude and location of fishing effort, including driftnets, 2) quantify the spatially explicit risk to megafauna species across the three major tuna fishing gears and 3) explore the conservation status of species at risk from fishing. We demonstrate an application of a ranked probability-based ERA method to a data-poor context that is typical of many fisheries and bycatch species, and present the first spatially explicit estimate of risk of mortality across multiple gears and taxa in the Indian Ocean. These results can serve as a baseline to guide regional management organizations such as the IOTC, national governing bodies, and NGOs to better prioritize how and where to invest limited resources in reducing fishing impacts on threatened species.

6.3 Materials and Methods

6.3.1 Species distributions and conservation statuses

Empirical data on abundance and distribution is lacking for many megafauna species in the Indian Ocean. We used species distribution maps from AquaMaps, which models species-specific envelopes of environmental preference that are based on occurrence records from published databases and include variables such as temperature, depth, and salinity (Ready *et al.* 2010). The result is a probability of occurrence for each species in each 0.5° grid cell. We selected the 405 species (348 elasmobranchs, 51 cetaceans, and 6 sea turtles) that the AquaMaps model predicts to occur within the depth range of tuna fisheries in the IOTC Area of Competence (hereafter "IOTC Area"), which covers the Indian Ocean (including the Persian Gulf and the Red Sea) to 45° and 55° South in the western and eastern Indian Ocean, respectively**.** Approximately 40% of these maps have been reviewed by experts. We used version 2020-2 of the Red List to assess species' conservation statuses (IUCN 2020).

6.3.2 Fishing effort

Reporting of catch and effort is not consistent across the tuna sectors in the Indian Ocean. Countries with fleets targeting tuna are required to report their catch to the IOTC at a maximum spatial aggregation should of 1°x1° grid cells for purse seines and 5°x5° cells for longlines (IOTC 2020). There are fewer requirements for gillnets because they are classified as artisanal gears; where gillnet catch or effort are reported, the data may refer to irregular areas (e.g. per port of unloading) (Indian Ocean Tuna Commission 2019). For a standard index of fishing effort across the three gear types, we used a global and spatially explicit model of fishing effort that reports effort in terms of engine power and fishing days (kWdays/year) for each 0.5-degree grid call (Rousseau *et al.* 2019; Rousseau 2020), and selected all grid cells within the IOTC Area.

Compared to longline and purse seine gears, there is considerable variability in the characteristics and configuration of gillnets and what species they are used to target. A variety of gillnets are used in the Indian Ocean and the country reports rarely include specific information about their gillnet fleets, such as the number of vessels that use gillnets, whether they are bottom-set or drifting, and mesh sizes used. Most fleets using driftnets to target tuna and tuna-like species in the Indian Ocean have a stretched mesh size of 13-17cm (Shahid *et al.* 2015). However, these nets can be used to target a variety of other species in addition to tunas, including demersal sharks and rays, Spanish mackerels (Scombridae), catfish (*Arius* spp.), and seabreams (Sparidae), and can be used interchangeably as bottom set gillnets and driftnets depending on the season and target species

(Shahid *et al.* 2016; Khan 2017). Vessels also frequently use multiple gears in combination, such as drift gillnets with snoods attached along the lead line or nets hung between pelagic longlines, which further complicates estimates of fishing effort (Henderson *et al.* 2007; Jabado and Spaet 2017; Yulianto *et al.* 2018; Winter *et al.* 2020). The catch data reported to the IOTC does not distinguish between larger, offshore driftnets primarily targeting tuna and smaller inshore drift or set gillnets. To focus on boats more likely using driftnets, we first removed all unpowered vessels and vessels in power categories 1 and 2, leaving only vessels >25 kW (approximately 35 HP). Second, we conducted a literature review and removed gillnet effort from countries with no reported drift gillnet fleets operating in the Indian Ocean (Table S4.3.1). Finally, we corrected for spatial skewedness by adjusting outlier cells and scaled the fishing effort from 0-1 **(**Appendix 4.1.1 Supplementary Info 1: Fishing effort). The resulting value represents a relative probability that fishing occurs in each grid cell.

6.3.3 Risk Assessment

To compare risks to species across the three tuna fishing gears, we use a semi-quantitative ecological risk assessment (ERA) that incorporates expert judgment where empirical data are not available (Hobday *et al.* 2007, 2011). This method is designed to assess risk when information is missing or highly uncertain, such as the Indian Ocean context where there is limited information for both species (e.g., distribution, abundance, habitat preferences) and fishing (e.g., intensity and location). We adapted this method to use ranked probabilities instead of discrete scales (e.g., low, medium, high or 1, 2, 3), which is the typical approach used in earlier iterations of the method.

This ERA method expressed risk in terms of a relative probability of capture and an interval of possible outcomes for an individual animal based on species and gear attributes (the per capita vulnerability). It is essentially the first half of a Productivity Susceptibility Analysis (PSA), which estimates a threat's potential impact on a species or population. A PSA incorporates information about the species' productivity (factors that influence the intrinsic rate of increase, such as reproductive rate, lifespan, and biomass) as well as its susceptibility to fisheries mortality (likelihood of encountering and entangling in fishing gear) to estimate the damage that fishing could cause to a species or population (Hobday *et al.* 2007). The biological information needed for the productivity component of the PSA is not available for most species in our focus subset; therefore, we limit this analysis to the estimated mortality in fishing gears (the susceptibility component).

The risk of capture, injury or mortality in a fishing gear is a function of availability (horizontal overlap of the species and the fishing gear), encounterability (the vertical overlap of the animal and the gear in the water column), gear selectivity (e.g., is the animal the right size and shape to become entangled in a net, is it attracted to bait), and the potential lethality if entangled. The first three parameters are probabilities and the product is the relative probability of capture, whereas the lethality is an interval indicating the range of outcomes if the animal were captured (or, "how bad is it?"). The final score can be interpreted as "expected mortality" and has an upper and lower bound:

$$Expected\ mortality_{(min)} = A\ x\ E\ x\ S\ x\ lethality_{(lower\ bound)}$$

$$Expected\ mortality_{(max)} = A\ x\ E\ x\ S\ x\ lethality_{(upper\ bound)}$$

where A = Availability, E = encounterability, and S = selectivity.

For the horizontal overlap (availability), we converted the fishing effort and species' distribution maps to raster files, then multiplied the species' probability of occurrence and the scaled fishing effort value in each grid cell using the Raster Calculator Tool in ArcMap 10.8. The probabilities are proxies for density of animals and fishing gear (assuming more fishing gear in high effort cells and more animals present in a cell with a high probability of occurrence). The probability of occurrence leads to underestimates of availability for abundant species (e.g. *C. falciformis*) compared to species with smaller population sizes (e.g. *R. typus*). Likewise, the measure of fishing effort (aggregated by year and gear type across all fishing countries) does not capture spatial and temporal variability in how much fishing gear is actually in the water within a given cell. In this per-capita framing of risk, the availability represents the probability that an individual animal and fishing gear are both present in that cell.

$$Availability_{(cell)} = P(species\ occurs) * P(fishing\ occurs)$$

This calculation of availability does not account for temporal variability (e.g., diurnal vertical migrations, time of day of fishing operations), seasonal variability (e.g., annual migrations, shifting fishing effort around the monsoon season), or different life stages of species (e.g. sea turtles and many elasmobranchs have juvenile phases with distinct life histories). These assumptions lead to overestimations of risk where the actual overlap between fishing and animals is lower than predicted, and underestimations of risk where overlap is greater than predicted because seasonal or diurnal densities coincide.

For the vertical overlap (encounterability), there is very limited information available on the vertical distribution and diving behaviour of most species. We conservatively assumed all gears are deployed from the surface to 20m for drift gillnets (Aranda 2017), 280m for purse seines (Romanov 2002), and 400m for longlines (Song *et al.* 2009). For species' depth ranges, we used depth ranges from the AquaMaps model and adjusted depths for 46 species (38 cetaceans, two sharks, and all sea turtles) based on available empirical information (Appendix 4.1.2 Supplementary Info 2: Species information). We then calculated the overlapping depth range for each species and gear types, assuming that both species and fishing gears were evenly distributed throughout the overlapping range and that the overlap was the same across all cells. This assumption leads to underestimates of catchability for species and gears that more often concentrated in the same shallow portion of their depth ranges, and overestimates of catchability for species that spend more time at depths beyond the range where most of the fishing effort is concentrated (for example, many demersal-associated elasmobranchs are less likely to encounter tuna gears than the depth overlaps suggest).

$$Encounterability_{(species,gear)} = \frac{overlapping\ depth\ range}{species\ depth\ range}$$

Less empirical information is available for the third parameter (gear selectivity) because few studies have quantified the likelihood of entanglement in fishing gears independent of species abundance and fishing effort. We compiled a database of the 405 species and used information from secondary sources to group species according to life history traits with similar propensity for entanglement and mortality in fishing gear, including body size and shape, foraging ecology, habitat use, including attraction to Fish Aggregating Devices (FADs) (Table 6.1, Table S4.3.2). We conservatively assumed that all purse seines are fishing around FADs, which has become the dominant (although not universal) practice in Indian Ocean tuna fisheries (Davies *et al.* 2014). Sets on FADs have bycatch levels approximately three times those on free-swimming sets, in addition to capturing more species (Davies *et al.* 2014; Lezama-Ochoa *et al.* 2015).We then ranked the species groups (allowing ties) by the likelihood of entanglement in each gear type, if encountered, allowing species to receive individual selectivity ranks. For example, humpback whales (*Megaptera novaeangliae*) are more often entangled in gillnets compared to other baleen whales, and thus were ranked higher for that gear (Johnson *et al.* 2005). We then randomly generated probabilities for each rank using an order-preserving Monte Carlo process in R and allowing ties.

Table 6.1: Fifteen species groups for ranking gear selectivity and assigning lethality intervals, based off habitat use, physical characteristics, and known interactions with fisheries

Taxonomic group	**Subgroup name**	**Code**	**Description**
Cetaceans	Baleen whales	BW	Coastal and oceanic baleen whales
Cetaceans	Large Oceanic dolphins	LOD	Large oceanic dolphins (beyond continental shelf)
Cetaceans	Oceanic toothed & beaked whales	OCTBW	Beaked and toothed whales (including all sperm whales) with oceanic distribution
Cetaceans	Shallow inshore dolphins & porpoises	SINDP	Nearshore species primarily in shallow (<50m) depths
Cetaceans	Small oceanic & coastal dolphins	SOCCOD	Small or medium sized dolphins found in oceanic or coastal areas primarily >50m depth
Elasmobranchs	Deep sea elasmobranchs	DSE	Benthic or demersal species anywhere along the continental shelf and upper slope >200m depth, or deep sea pelagic species >400m depth (species primarily outside the depth range of tuna gears)
Elasmobranchs	Deep shelf pelagic elasmobranchs	DSPE	Pelagic species anywhere along the continental shelf and upper slope >200m depth
Elasmobranchs	Demersal generalist elasmobranchs	DGE	Primarily feeds or lives on the bottom, occupies range of depths & range of habitats
Elasmobranchs	Inshore elasmobranchs	INE	Shallow (<100m depth), common in coastal areas (continent & island)
Elasmobranchs	Oceanic elasmobranchs	OCE	Pelagic species found in open ocean (beyond continental shelf)
Elasmobranchs	Pelagic filter feeder elasmobranchs	PFFE	Filter feeders that primarily feed or live in the pelagic zone, occupy a range of depths & range of habitats
Elasmobranchs	Pelagic generalist elasmobranchs	PGE	Primarily feeds or lives in the pelagic zone, occupies range of depths & range of habitats
Elasmobranchs	Reef elasmobranchs	RE	Known to occupy temperate and tropical reef habitat a majority of the time
Elasmobranchs	Shallow shelf elasmobranchs	SSE	Anywhere along the continental shelf <200m depth
Sea turtles	Sea turtles	ST	Six species of sea turtles (including Dermochelys coriacea)

The probability of capture is the likelihood of the event occurring. The second component of the estimate of risk is the severity of the outcome, if the event occurs. We assume the interaction is lethal unless the animal is able to escape, as there is insufficient information about compliance with safe release practices in the Indian Ocean (Zollett and Swimmer 2019). Releasing entangled animals is usually ineffective for gillnets because they are static and typically deployed overnight, so air-breathing species or elasmobranchs that need to swim to breathe are likely to drown (Zollett and Swimmer 2019). Pelagic longlines allow hooked animals to move but are usually set at depth and can also have long set times (usually more than 12 hours and sometimes more than 24 hours) (Chen *et al.* 2012; Clarke *et al.* 2014), and survival rates are highly variable for individuals that are successfully released (Carruthers *et al.* 2009). Compared to longlines and gillnets, survival rates of species released from tuna purse seines are expected to be higher for sea turtles and cetaceans, although studies are lacking (Escalle *et al.* 2015; Hamilton and Baker 2019; Zollett and Swimmer 2019). Studies suggest much lower post-release survival rates for pelagic elasmobranchs caught in purse seines (Eddy *et al.* 2016).

Once entangled, the severity of the outcome depends on physical characteristics of the animal (its ability to escape). We assigned an interval for the lethality of the outcome to each group based on available empirical information for species within that group (**Error! Not a valid bookmark self-reference.**), allowing out-of-group intervals for species where available empirical data suggest they differ from their species group in terms of the lethality of entanglement. For example, blue whales are large enough to break through drift gillnets more easily than other baleen whales. We assumed that all longline fleets use monofilament leaders, which are easier for larger species to break compared to wire leaders (Gilman 2011). However, vessels that are targeting (or sub-targeting) sharks will likely use wire leaders and there is no comprehensive information about targeting dynamics across the wide variety of longline fleets operating in the region (Ardill *et al.* 2013). Following the ERA principle of precautionary scoring, we assigned the more conservative lethality interval where empirical data were lacking (Hobday et al. 2007).

Table 6.2: Intervals and descriptions of possible outcomes (lethality) if an animal is entangled in gear

Category	**Interval**	**Description**
No damage	[0,0]	Species escapes without injury that decreases fitness
Sublethal	[0,1)	Species will most likely escape, potentially unharmed, or will suffer minor to serious injuries
Potentially lethal	(0,1]	Species may escape with minor to serious injuries, or could be landed or die during entanglement

Lethal	[1,1]	Species is a target or like-target species and will likely be landed or die during entanglement

From the three probabilities, we calculated the probability of capture and expected mortality intervals for each species and gear type in each grid cell:

$$Catchability_{(cell)} = Availability_{(cell)} * Encounterability * Selectivity$$

$$Expected\ mortality_{(min)} = Catchability_{(cell)} * Outcome_{(lower\ bound)}$$

$$Expected\ mortality_{(max)} = Catchability_{(cell)} * Outcome_{(upper\ bound)}$$

We then calculated the mean catchability and expected mortality intervals for each species across all cells where it occurred within the IOTC area and the percent overlap of each species and gear (a rough indicator of exposure to fishing, at least in the horizontal dimension).

6.4 Results

6.4.1 Species catchability and conservation status

Of the 405 species, 367 had a catchability probability greater than zero in at least one of the three gears. The species ranking highest for mean catchability across the three gears are all shallow shelf elasmobranchs, pelagic generalist elasmobranchs, or shallow inshore dolphins and porpoises, with three sea turtle species also scoring in the top 25 species (Figure 6.1, Table 6.3). The three species with the highest cumulative catchability scores are the slender weasel shark (*Paragaleus randalli*), Human's whaler shark (*Carcharhinus humani*), and Grey sharpnose shark (*Rhizoprionodon oligolinx*) (Table 6.3). In general, the species with the highest cumulative catchability scores have wide ranges and inhabit offshore pelagic regions, such as *Alopias spp., P. violacea, Sphyrna spp., C. longimanus,* and *C. falciformis.*

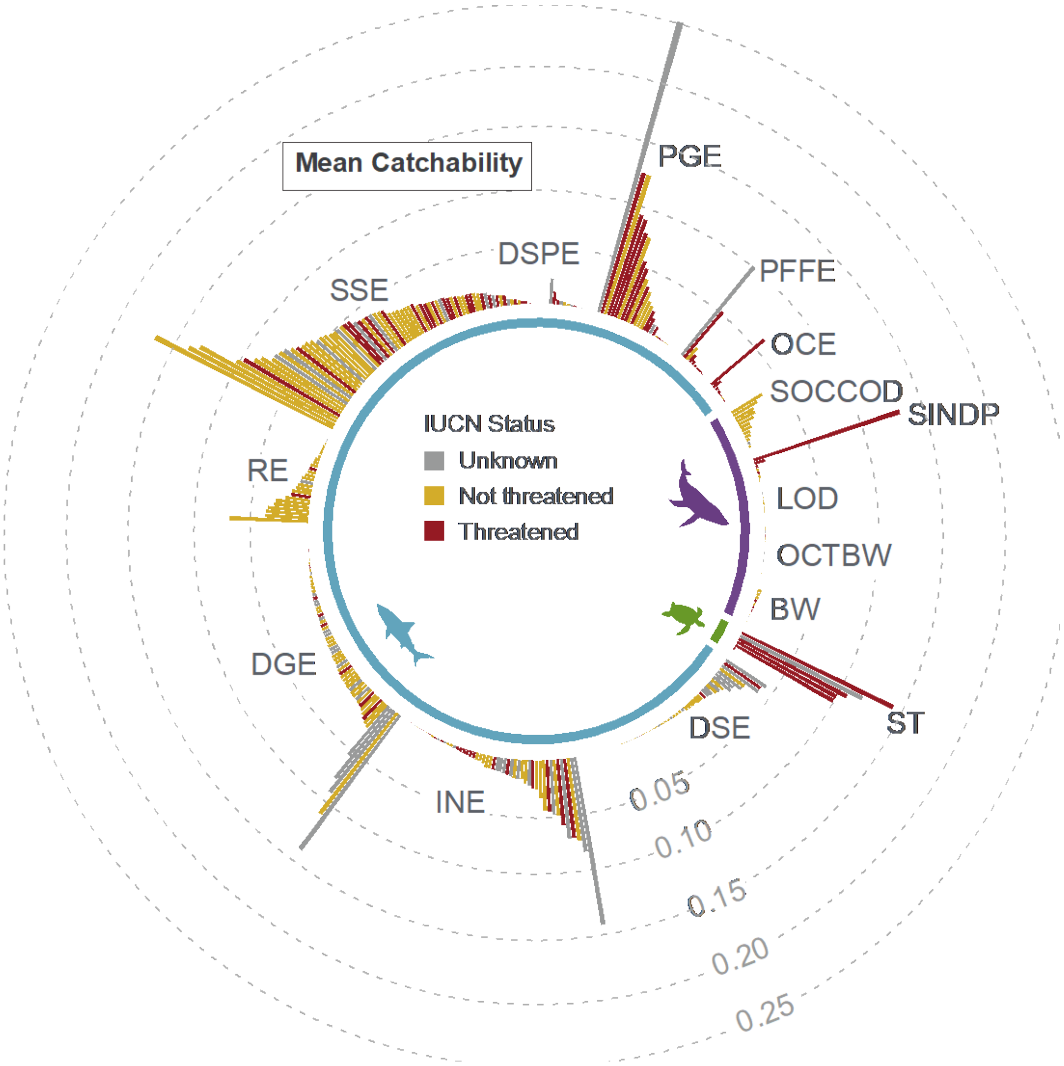

Figure 6.1: Mean catchability probabilities summed across the three gear types for species in 15 species groups, ordered first by taxonomic group (cetaceans, sea turtles, elasmobranchs) then by sub-group (See Table 6.1 for species groups). Color shows threat group (Threatened = Critically Endangered, Endangered, Vulnerable, Not Threatened = Near Threatened, Least Concern, Unknown = Data Deficient or Not Assessed.

Many of the species with the highest cumulative catchability scores are threatened or have an unknown status (Figure 6.1, Table 6.3). Overall, more than a quarter (27%) of the catchable species are threatened, with 5% (17) Critically Endangered, 8% (30) Endangered, and 14% (52) Vulnerable. The groups containing fewer species have the highest proportions of threatened species, with seven out of nine (78%) pelagic filter feeder elasmobranchs, five out of six (83%) sea turtles, six out of seven (86%) oceanic elasmobranchs, and four out of seven (57%) shallow inshore dolphins and porpoises listed as threatened. Over half (51%) of the catchable species are not threatened (Least Concern or Near Threatened), although one-fifth (21%) are listed as Data Deficient or have not been assessed by the IUCN. Oceanic toothed and beaked whales (e.g., *Mesoplodon* spp., *Kogia* spp.) have the highest proportion of Data Deficient species (60%), followed by 36% of deep shelf pelagic elasmobranchs (e.g., *Oxynotus bruniensis, Cirrhigaleus asper*) and 25% of demersal generalist elasmobranchs (e.g. *Squatina* spp*., Raja miraletus*) (Table S4.3.4). Most sea turtles and cetaceans are listed on CMS or CITES (or both), but most elasmobranchs are not, especially poorly known species and species that are widely targeted by fisheries.

Table 6.3: Conservation status information and cumulative catchability scores for the top 25 species ordered by descending mean catchability score (sum of the mean score across all gear types). Catchability sum = sum of all catchability scores across all gears and cells. Mean = mean score across all gear types and cells. CR = Critically Endangered, EN = Endangered, VU = Vulnerable, NT = Near Threatened, LC = Least Concern, DD = Data Deficient, Elasmos = elasmobranchs

Species	**Species group**	**Catchability**		**Red List**	**Appendix**	
		Mean	**Sum**		**CMS**	**CITES**
Paragaleus randalli	Shallow shelf elasmos.	0.556	132	NT	--	--
Carcharhinus humani	Pelagic generalist elasmos.	0.420	47	DD	--	--
Rhizoprionodon oligolinx	Shallow shelf elasmos.	0.359	265	LC	--	--
Carcharhinus galapagensis	Pelagic generalist elasmos.	0.314	64	LC	--	--
Carcharhinus sealei	Shallow shelf elasmos.	0.273	184	NT	--	--
Glaucostegus halavi	Shallow shelf elasmos.	0.263	195	CR	--	II
Mobula mobular	Pelagic filter feeder elasmos.	0.245	265	EN	--	--
Chaenogaleus macrostoma	Shallow shelf elasmos.	0.240	121	VU	--	--
Neophocaena phocaenoides	Shallow inshore dolphins & porpoises	0.240	80	VU	II	I
Eretmochelys imbricata	Sea turtles	0.239	300	CR	I/II	I
Sousa chinensis	Shallow inshore dolphins & porpoises	0.233	185	VU	II	I
Carcharhinus brevipinna	Shallow shelf elasmos.	0.222	202	NT	--	--
Lepidochelys olivacea	Sea turtles	0.221	176	VU	I/II	I
Chelonia mydas	Sea turtles	0.221	275	EN	I/II	I
Orcaella brevirostris	Shallow inshore dolphins & porpoises	0.211	43	EN	I/II	I
Carcharhinus sorrah	Shallow shelf elasmos.	0.207	180	NT	--	--
Brevitrygon imbricata	Shallow shelf elasmos.	0.202	169	DD	--	--
Rhynchobatus djiddensis	Shallow shelf elasmos.	0.201	71	CR	--	II
Aetomylaeus maculatus	Inshore elasmos.	0.196	108	EN	--	--
Megatrygon microps	Inshore elasmos.	0.188	141	DD	--	--
Himantura undulata	Shallow shelf elasmos.	0.183	82	VU	--	--
Carcharhinus plumbeus	Pelagic generalist elasmos.	0.182	231	VU	--	--
Carcharhinus dussumieri	Shallow shelf elasmos.	0.181	27	EN	--	--
Torpedo panthera	Demersal generalist elasmos.	0.177	42	DD	--	--
Aptychotrema vincentiana	Shallow shelf elasmos.	0.175	39	LC	--	--

Most of the highest mean expected mortality scores are for driftnets—including many threatened species—although purse seines and longlines are very high risk for several elasmobranch species (Figure 6.2: Mean expected mortality across all cells and percent range overlap with driftnets, longlines, and purse seines for Threatened, Not threatened, and Unknown status species. The 25 species with the highest mean catchability scores overall are labeled. In general, purse seines and longlines pose the greatest risk to elasmobranchs (pelagic generalists, shallow shelf, and inshore species) and proportionally more small cetaceans are ranked high for driftnets, although driftnets are high-risk for many elasmobranchs as well (Figure 6.2, Table S4.3.3). All three gears pose a high risk to sea turtles. Many species with moderately high mean catchability scores have large ranges that overlap closely with fishing effort, and thus have high cumulative risk across the IOTC Area. For example, *Caretta caretta* has high cumulative catchability in driftnets, *I. oxyrinchus* and *P. glauca* in longlines, *Mobula birostris* and *Stenella longirostris* in purse seines, and *C. longimanus, C. falciformis, P. kamoharai, Alopias spp., Sphyrna spp.* and *P.violacea* in both longlines and purse seines (Figure S4.2.1, Figure S4.2.2, Figure S4.2.3). Many species with low mean and low cumulative catchability probabilities (e.g., baleen whales) still have a large proportion of their range overlapping horizontally with fishing gears (based on presence-absence of species and fishing), especially with longlines and purse seines (Figure 6.2: Mean expected mortality across all cells and percent range overlap with driftnets, longlines, and purse seines for Threatened, Not threatened, and Unknown status species. The 25 species with the highest mean catchability scores overall are labeled. A proportionally large horizontal overlap of a species and gear does not necessarily mean the species is likely to be caught, but does indicate species-gear interactions that could be important over the extent of the species range in the IOTC Area, even if the mean catchability per cell is relatively low.

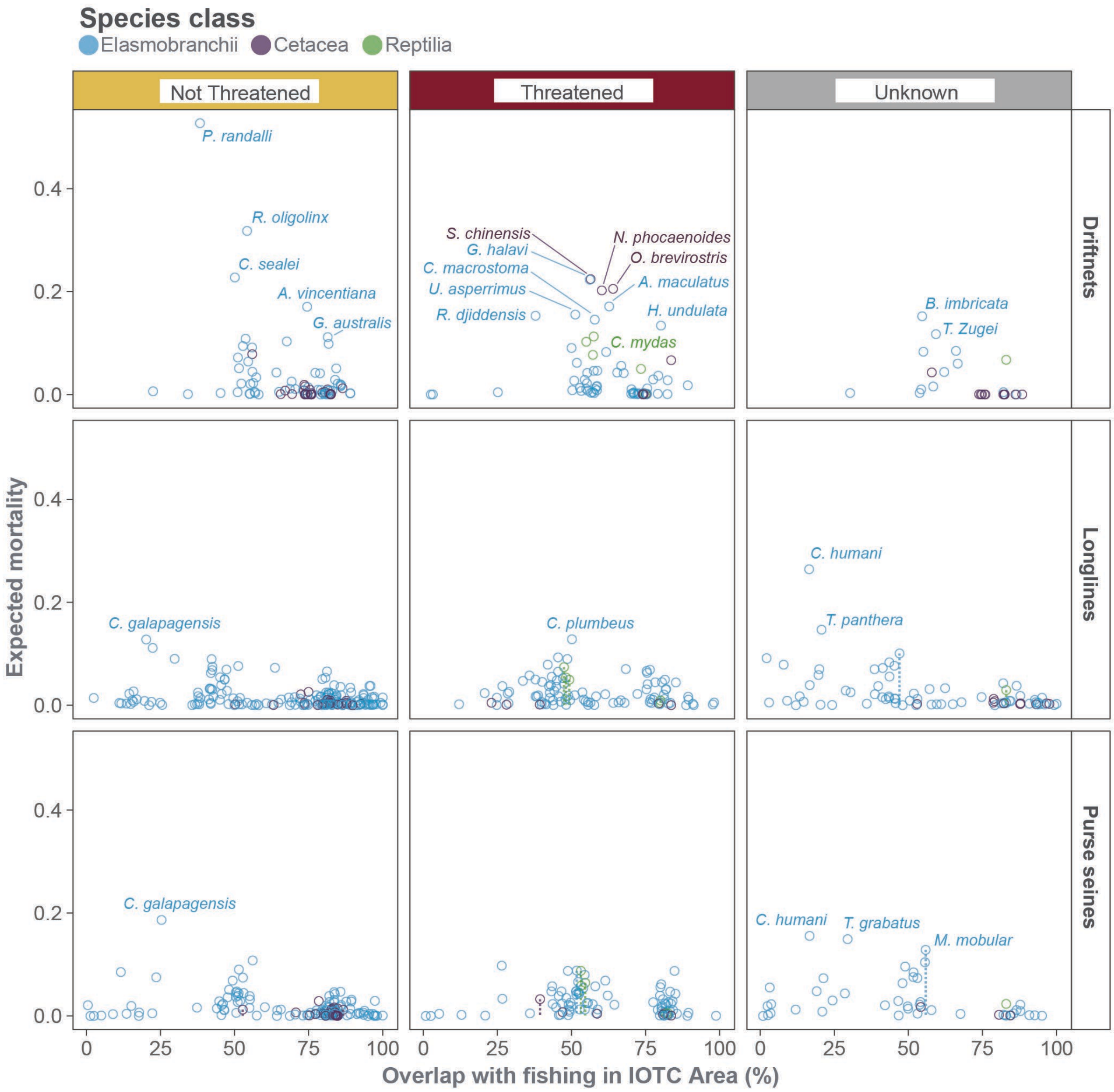

Figure 6.2: Mean expected mortality across all cells and percent range overlap with driftnets, longlines, and purse seines for Threatened, Not threatened, and Unknown status species. The 25 species with the highest mean catchability scores overall are labeled.

Overall, the potential for cumulative impacts on species is high. Two-fifths (41%) of the 367 catchable species are catchable in all three gears, 36% are catchable in two of the three gears, and 23% are only catchable in longlines (mostly deep shelf elasmobranchs). The high cumulative expected mortality scores are driven by driftnets, which have high catchability probabilities and lethality outcomes compared to longlines and purse seines, although all gears were conservatively rated as "lethal" for most species (Figure 6.2**,** Figure 6.3). In fact, most of the lethality intervals are not visible on Figure 6.2 because the species-gear combinations with the highest expected mortality scores were all scored as lethal (except for *M. mobular*). The interactions where species are more likely to escape (potentially lethal, sublethal, or no damage) are primarily cetaceans, sea turtles, and larger elasmobranchs in longlines and purse seines (Figure S4.2.4). Although less lethal potential outcomes are obviously better for the animal, these interactions also have the widest margin of uncertainty about the level of damage inflicted on the individual, as it is difficult to measure the impacts of fishing interactions on animals that escape.

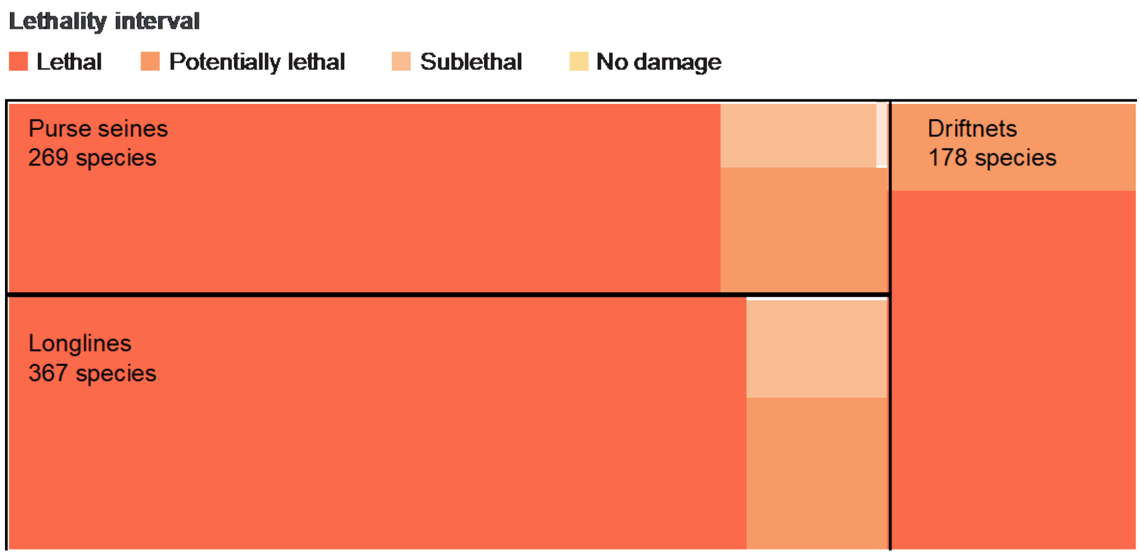

Figure 6.3: Number of species in each lethality interval for gillnets, longlines, and purse seines. Area corresponds to proportion of catchable species for each gear.

6.4.2 Comparison to available bycatch reports

The aim of this analysis is to quantify the risk of capture in tuna fishing gears, and the severity of that outcome. The estimated mortality is expressed in terms of an individual animal, which can then be summed across the population or geographic areas. The estimation of expected mortality for individual species is not directly comparable to reported bycatch in Indian Ocean tuna fisheries because available data rarely account for fishing effort (catches are given in total volume or number of individuals, not per unit of fishing), and abundance and density are not known for most non-target species. Therefore, this measure of risk cannot be translated into a total catch estimate for each species. As a rough validation of our results, we compare the ranked probability scores to available bycatch reports and find general agreement at the level of the species group (e.g., sea turtles, pelagic filter feeding elasmobranchs) and for species with high cumulative probabilities of capture (Figure S4.2.1, Figure S4.2.2, Figure S4.2.3). However, catchability scores were unexpectedly high for many demersal elasmobranchs (e.g., electric rays, guitarfish) in all three gear types. This is a function of the species ranges extending into shelf areas where the gear's possible depth range would extend to the seafloor. In reality, these species are unlikely to encounter pelagic fishing gears because they remain near the sea floor while the gear would be deployed in the pelagic zone.

6.4.3 Spatial patterns of risk

We selected motorized fishing effort in 2015 in the IOTC Area and found 22 countries fishing with driftnets, 26 countries fishing with purse seines, and 39 countries fishing with pelagic longlines. Across the IOTC area, longlines are predicted to encounter the most species (n=367), followed by purse seines (n=269) and drift gillnets (n=178) (Figure 6.3). Longlines have a large footprint and the largest depth range (0-400m and sometimes deeper), although most fishing effort occurs shallower than 300m as deeper sets are only for albacore and bigeye tuna (*Thunnus alalunga* and *T. obesus*) in some fishing grounds (Chen *et al.* 2005; Song *et al.* 2009). While fewer species are predicted to encounter driftnets, the cumulative catchability per cell is much higher than the other gears (Figure 6.4).

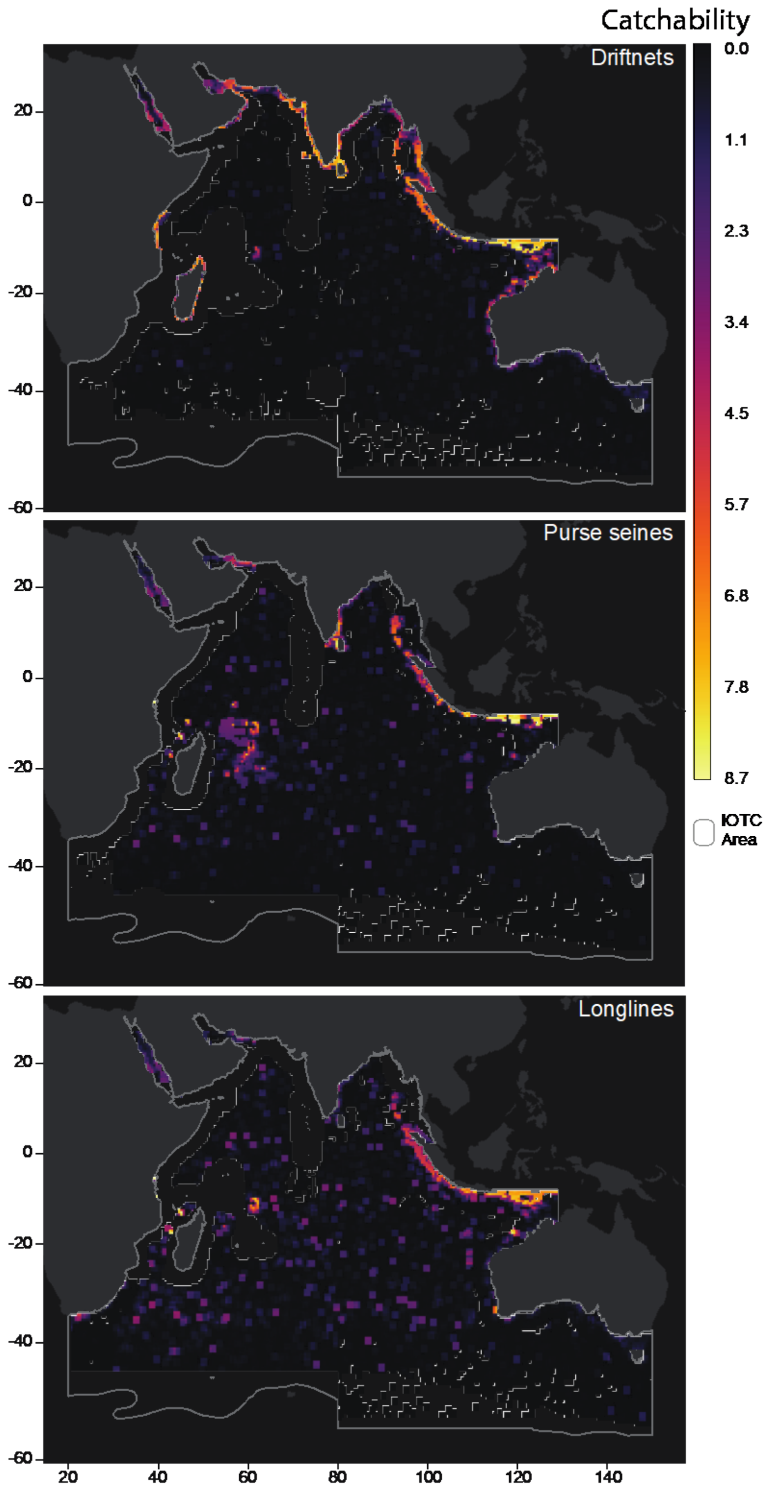

Figure 6.4: Map of cumulative catchability across all species in each grid cell for driftnets, purse seines, and longlines.

The cumulative threat from the tuna sectors is concentrated in a relatively small proportion of the IOTC area, mostly in coastal regions (Figure 6.4). Western Indonesia stands out as a high-risk area across all three gears, and there is substantial overlap in parts of the Red and Arabian Seas as well. Driftnet catchability is very high along most of the coastal areas, including regions that have lower cumulative risk from purse seines and longlines (Madagascar, Tanzania, Kenya, Iran, Pakistan, eastern India, Bangladesh, Myanmar, and north-western Australia). Compared to driftnets, high-risk longline and purse seine areas are more dispersed in offshore areas. High purse seine catchability overlaps with driftnets around Sri Lanka, the western coast of India, and in parts of the Arabian Sea. High risk areas in the Southwest Indian Ocean around Seychelles, Mauritius, and Reunion are driven primarily by purse seines.

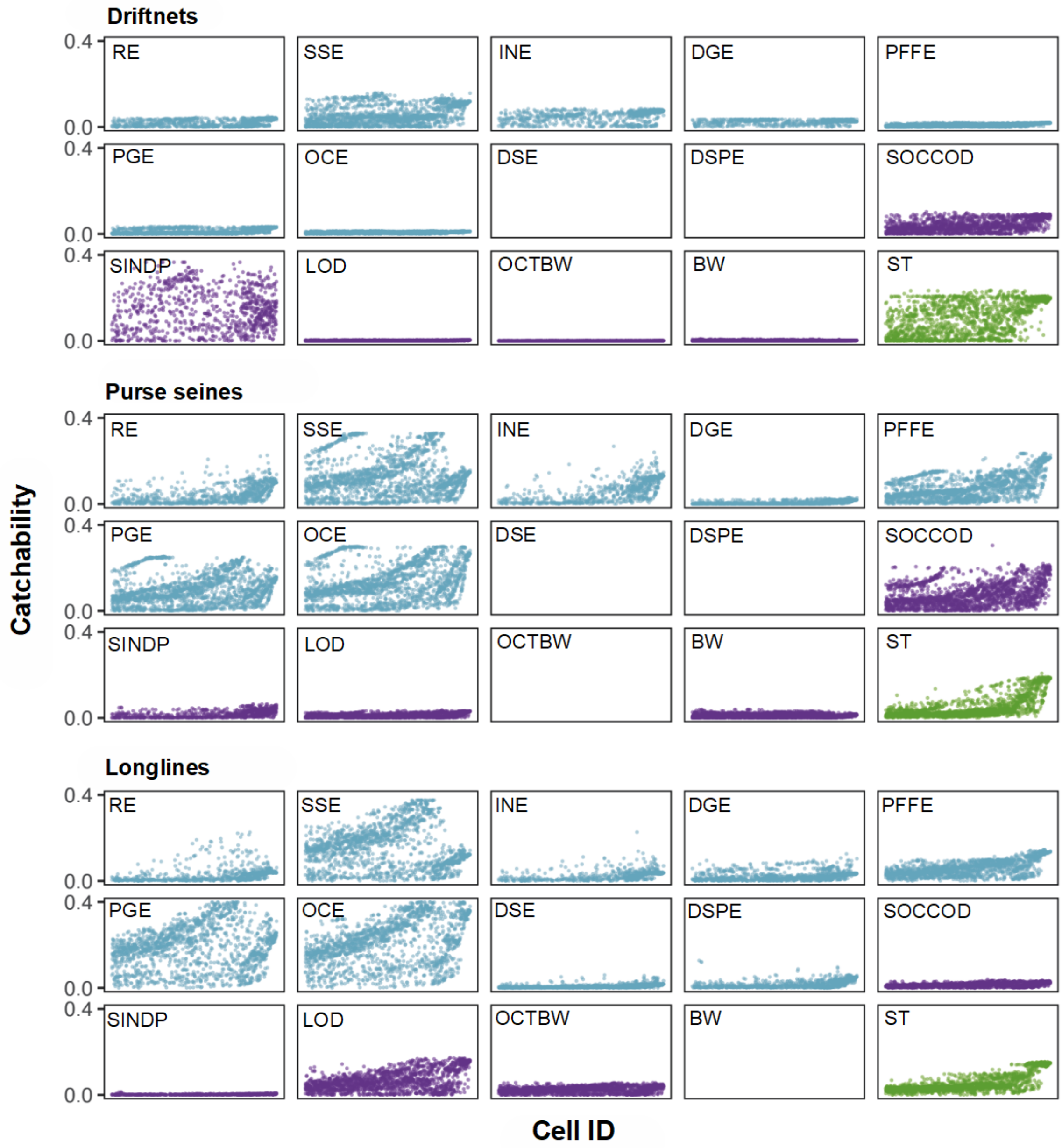

Figure 6.5: Sum of catchability scores for all species occurring in each cell for gillnets, longlines and purse seines, separated into species groups. Green is for sea turtles, purple is cetaceans, and blue is elasmobranchs. From top left: RE=reef elasmobranchs, SSE=shallow shelf elasmobranchs, INE=inshore elasmobranchs, DGE=demersal generalist elasmobranchs, PFFE=pelagic filter feeding elasmobranchs, PGE=pelagic generalist elasmobranchs, OCE=oceanic elasmobranchs, DSE=deep sea elasmobranchs, DSPE=deep shelf pelagic elasmobranchs, SOCCOD=small oceanic and coastal dolphins, SINDP=shallow inshore dolphins and porpoises, LOD=large oceanic dolphins, OCTBW=oceanic toothed and beaked whales, BW=baleen whales, ST=sea turtles. Cells are ordered by ascending cumulative catchability across all species and gears (meaning each cell's location on the x-axis is unique and comparable across all plots). The 2,037 cells in the top 10% of catchability values (for all three gears combined) are shown.

There is moderate overlap of the highest risk cells in the IOTC Area across fishing gears and species groups (Figure 6.5). For example, sea turtles (ST) have high catchability in driftnets, and most of those high-risk cells also have high catchability for sea turtles in longlines and purse seines. Shallow inshore dolphins and porpoises (SINDP) are most at risk from driftnets, but there is substantial overlap between those high-risk cells and the high-risk cells for other gears and species groups (e.g., sea turtles in all gears and the high-risk elasmobranch groups in longlines and purse seines). Overall, the pattern of high-risk cells is most similar between purse seines and longlines for all elasmobranch groups, except for deep elasmobranchs which are only predicted to encounter longlines.

6.5 Discussion

Very few fisheries management bodies require detailed records of non-target species and many countries lack the capacity even for basic monitoring of target species catch, especially countries in the Indian Ocean region (Ricard *et al.* 2012; Juan-Jordá *et al.* 2018). Due to the lack of empirical data available for the region, previous studies of bycatch in the Indian Ocean have been limited in their geographic area and number of species and fisheries considered (e.g., Bourjea *et al.* 2008; Huang and Liu 2010; Escalle *et al.* 2015). We use a rank-probability ERA method that incorporates spatially explicit models of the probability of species' occurrence (Kaschner *et al.* 2016) and fishing effort (Rousseau *et al.* 2019; Rousseau 2020) to estimate and compare the risk of capture and mortality of megafauna species across the three main tuna fishing gears in the Indian Ocean. This ERA method is designed to quantify and compare risk in terms of vulnerability of an individual animal, not to estimate the total number of animals caught in fishing gears, although these point estimates are important communication tools for management and conservation purposes (Read *et al.* 2006; Anderson *et al.* 2020).

Our results show that many cetacean, sea turtle, and elasmobranch species face substantial cumulative risks from tuna fishing sectors in the Indian Ocean, with driftnets driving the highest catchability scores. Many of the species with the greatest expected mortality across their range are listed as threatened on the Red List and have few protections. We found high risk of capture and mortality for known risk groups such as small cetaceans in driftnets (Reeves *et al.* 2013; Brownell *et al.* 2019; Anderson *et al.* 2020), mesopelagic sharks and rays in longlines and purse seines (Amande *et al.* 2012; Murua *et al.* 2018; Garcia and Herrera 2019), and sea turtles in all three gears (Varghese *et al.* 2010; Ardill *et al.* 2013; Wallace *et al.* 2013; Lewison *et al.* 2014; Ortiz *et al.* 2016). Additionally, we found that many poorly known or monitored elasmobranchs are at high risk from one or more gears (e.g., *Megatrygon microps*, *Hemigaleus microstoma*). Most of these species are rarely (if ever) specifically listed in available catch reports from the Indian Ocean, or even from other regions with more extensive bycatch monitoring.

The high-risk species that are not mentioned in reports (e.g., many species in the genus *Carcharhinus*) are either rarely caught (perhaps because they are not abundant), or the catch is not being recorded or only recorded in very aggregated groups (e.g., "pelagic sharks"). The latter is likely the case for many of the high risk pelagic and semi-pelagic elasmobranchs, which can be difficult to identify even for trained observers (Román-Verdesoto and Orozco-Zöller 2005; Smart *et al.* 2016). In contrast, the high-risk benthic or demersal elasmobranchs are probably not caught in tuna gears. These high scores are driven by the assumptions of the encounterability parameter, which assumes uniform distribution throughout the depth range and results in a high probability of encountering gear if the species' depth range overlaps closely with the depth of the fishing gear. Future analyses could refine this parameter by estimating the distribution of species and fishing effort throughout the depth range, at least by life-history group (e.g. sea turtles, benthic elasmobranchs, deep-diving whales), and could also incorporate estimates of the distribution of fishing effort in the water column. The encounterability parameter could be further improved by area-specific depth ranges, which would give a probability of encounter per cell instead of a uniform value, in the same way that availability is calculated.

Overall, cumulative expected mortality in purse seines is probably lower than our results indicate, for two main reasons. First, we assume that all purse seiners set on Fish Aggregating Devices (FADs). Although we likely overestimate expected mortality in purse seines for some species (e.g., *S. longirostris, Neophocaena phocaenoides, Eretmochelys imbricata*), known bycatch rates in purse seines set on FADs do not account for the additional mortality from ghost fishing, where pelagic sharks and sea turtles in particular can get entangled in the net hanging below the raft (Davies *et al.* 2014). Second, we assume that no bycatch mitigation tactics are in place for any gears, even for species with little market value (such as small deep-sea skates and rays). Since some Indian Ocean purse seiners do use safe release practices, which are reasonably effective for cetaceans and turtles, we likely overestimate risk to these taxa from this gear type (Bourjea *et al.* 2008; Amande *et al.* 2012; Escalle *et al.* 2015; Clavareau *et al.* 2020).

For driftnets, which have a much narrower depth range than purse seines or longlines, accounting for distribution in the water column is less relevant than separating the smaller bottom-set nets from the larger surface nets. Although we make some rough adjustments to the effort model in an attempt to subset drift gillnets targeting tuna and tuna-like species, a substantial portion of the predicted driftnet effort likely comes from vessels predominately using set gillnets. These boats are often targeting small pelagic fish such as anchovies, sardinellas, hilsa shad, and other herrings, especially around estuaries (FAO 2014; Sekadende *et al.* 2020). There is also a sizeable bottom-set gillnet sector that uses slightly larger mesh nets to target sharks and rays, particularly in the Northern

Indian Ocean (the Arabian Sea, Bay of Bengal, and western coast of Indonesia) (Henderson *et al.* 2007; Jabado *et al.* 2015). The relatively high expected mortality off Northwestern Australia is a result of large demersal gillnets targeting sharks and nearshore gillnets targeting barramundi (*Lates calcarifer)* (Gaughan and Santoro 2020).

For many species, catch rates in inshore bottom-set and offshore pelagic gillnets are likely quite different (Gillett 2011). Even if categories were rough, some standardized gillnet sub-categories would greatly improve our knowledge and understanding of this important sector. The IOTC is working to improve reporting but this will require substantial investment in helping member countries to inventory their fleets and monitor catch, especially for countries with very limited management capacity (e.g. Somalia, Yemen) (Sinan and Bailey 2020). Improving monitoring and management of the essentially unregulated gillnet sector (including both set and driftnets) should be a priority to reduce megafauna bycatch in this region. In addition to the high risk of mortality for a variety of species, gillnets are a major source of mortality in marine debris globally (Good *et al.* 2010), and are likely contributing to a growing issue of unmonitored FADs in the Indian Ocean (Davies *et al.* 2014).

Improving our understanding of the dynamics of the diverse fishing sectors in the Indian Ocean is a crucial first step in directing conservation resources and designing interventions to mitigate bycatch and protect threatened species (Teh *et al.* 2015). In general, there are two main strategies for reducing mortality in fishing gears: reducing entanglement and reducing post-release mortality (Carruthers *et al.* 2009; Senko *et al.* 2014). Techniques that reduce encounters and entanglement include time-area closures (e.g. marine protected areas or closed areas for certain seasons or gears), modifications to the gear itself (e.g. attaching acoustic pingers to nets or changing bait, hooks, leaders, or mesh size and materials), or changing how the gear is deployed (e.g. setting gillnets lower in the water column, prohibiting purse seine sets on cetaceans, or restricting use of FADs) (Gilman 2011; Senko *et al.* 2014; Northridge *et al.* 2017). The second broad strategy is to improve survivability after entanglement—usually by implementing safe release practices—although tactical measures such as shortening the time the gear is deployed can also reduce mortality (Carruthers *et al.* 2009; Zollett and Swimmer 2019). Some strategies are widely effective in mitigating bycatch of a variety of species—such as restricting FADs—although target catch rates may be affected (Gilman 2011). Other strategies are more variable depending on the context and species, and in some cases may reduce one type of bycatch but increase catch rates of another species (Gilman *et al.* 2016).

The IOTC has fewer bycatch monitoring and mitigation requirements compared to the other tuna RFMOs, and it is the only one that does not implement spatial closures or gear restrictions (Boerder *et al.* 2019). There are relatively few MPAs in the Indian Ocean, and none located in international waters. The increased piracy around Somalia initially functioned as a de facto MPA, but evidence suggests that the governance void has over time resulted in increased illegal fishing in that area (Glaser *et al.* 2019). There is a global ban on setting driftnets longer than 2.5km in the High Seas and some scattered management measures within the IOTC Area (e.g., prohibiting purse seines from intentionally encircling whale sharks or marine mammals) (Garcia and Herrera 2019). However, reports indicate high rates of noncompliance across all types of fishing regulations (e.g., gear and area restrictions) within most EEZs and on the High Seas (Jabado and Spaet 2017; WWF 2020). The only bycatch mitigation techniques that the IOTC mandates are prohibiting purse seine sets on cetaceans and whale sharks, some regulation of FADs, and some requirements for safe release practices. However, lack of a common definition for FADs limits their effective management, and the IOTC has fewer safe release requirements than the other tropical tuna RFMOs (Zollett and Swimmer 2019; Swimmer *et al.* 2020).

While safe release practices are an important component of the bycatch mitigation portfolio and can move species from a lethal to a potentially lethal or sublethal outcome, they can still have significant effects on the animal's fitness (Wilson *et al.* 2014; Adams *et al.* 2018). Furthermore, safe release is only relevant to certain species and gears. Our results show high cumulative catchability and expected lethality for many sea turtles, cetaceans and elasmobranchs, with driftnets driving the very high scores. Most species entangled in gillnets are dead by the time they are landed, so safe release practices will not mitigate the impacts of this sector. Studies show that gillnets are also difficult to effectively modify (Senko *et al.* 2014; Brownell *et al.* 2019), although there are potential modifications that have not been rigorously tested across different areas and megafauna species (e.g., type and color of net filament, type of floatline, weight of lead line, net hanging ratio) (Northridge *et al.* 2017). There has been some success using acoustic pingers to reduce gillnet bycatch of beaked whales and some small cetaceans (e.g. harbor porpoises), although they are relatively expensive to purchase and maintain (Carretta *et al.* 2008; Hamilton and Baker 2019). Thus, the most promising effort control-based solutions are likely to be tactical changes in how the gear is deployed (e.g. setting slightly below the surface) and restricting their use at certain high-risk times or areas (Hamilton and Baker 2019).

We find that the cumulative risk of capture is concentrated in a relatively small proportion of the IOTC Area near the coasts, which suggests that targeted interventions in specific geographic areas could have important benefits for a range of species. Species with high expected mortality and overlap with fisheries proportional to their range and species with high cumulative catchability should be conservation priorities, especially species that are known to be threatened or declining. We found high catchability probabilities in purse seines and longlines for many elasmobranchs, which are likely overestimates for species that spend most of their time on or near the benthos. However, it is possible that some of these species are catchable in tuna gears because the Indian Ocean has biodiverse seamounts that are relatively shallow, and many elasmobranchs make diurnal migrations through wide ranges of the water column, making them simultaneously epipelagic, mesopelagic and bathypelagic (WWF 2020). An additional concern for many species in our analysis (including demersal elasmobranchs) is additional impacts from shrimp trawlers (Oliver *et al.* 2015). The limited conservation and management measures under the IOTC mandate only cover incidental catches of a relatively short list of non-target species, which is especially concerning for elasmobranchs as fishing patterns shift and growing demand from Asian markets increasingly makes them primary or secondary target species (Jabado and Spaet 2017; WWF 2020). Better catch monitoring—especially in the essentially unmonitored gillnet sectors—will be critical for management of fishing pressure on all bycatch species and elasmobranchs in particular. Species identification is particularly labour intensive for unselective fishing gears that catch many species (e.g., small or medium-mesh gillnets) and for species that are rarely encountered or difficult to identify; thus, limited bycatch data is an issue across all ocean regions, including in many wealthy countries (Clarke *et al.* 2014; Lewison *et al.* 2014).

The current regulatory framework in the Indian Ocean has substantial limitations and loopholes that allow fishing impacts on marine megafauna to continue at unsustainable levels (WWF 2020). The IOTC alone does not have the capacity to close these loopholes; effective bycatch management in the Indian Ocean will require coordinated efforts from all of the region's RFMOs, as well as Regional Fisheries Bodies, non-governmental organizations, and the seafood industry itself. We find that cumulative risks are concentrated in coastal areas within Exclusive Economic Zones, which highlights the importance of the coastal States in managing fishing in their marine estates. Given the severely limited governance capacity of many Indian Ocean countries, improving national fisheries management institutions will require substantial assistance from wealthier governments and regional organizations (Sinan and Bailey 2020). Although voluntary, international commitments such as the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and the Convention on Migratory Species (CMS) also provide opportunities to strengthen regulations around data collection and management measures for sea turtles, cetaceans, and elasmobranchs. Currently, the CMS and CITES provide some protections to sea turtles and cetaceans but few high-risk elasmobranchs are protected by these agreements. Better catch documentation will help identify species that merit consideration of CITES or CMS listings, including the many Data Deficient cetaceans and elasmobranchs that our results suggest are potentially caught in tuna fisheries.

Despite the challenges of improving catch documentation, emerging technologies such as electronic monitoring systems are becoming increasingly feasible (Suuronen and Gilman 2020). There are promising solutions aimed at reducing bycatch that are advancing beyond gear modifications to make fishing more selective for target species; for example, integrating satellite and other data sources to build dynamic management tools and bycatch warning systems (Howell *et al.* 2015; Hazen *et al.* 2018). Given the challenging management context in the Indian Ocean and the diversity of fishers and fishing fleets, bycatch mitigation tactics will likely be intractable without early and consistent engagement with fishers and local management bodies (Gladics *et al.* 2017; McCluney *et al.* 2019; Karnad and St. Martin 2020). While baseline information on species biology and catch should remain a priority for management agencies in the Indian Ocean, there is an urgent need to implement bycatch reduction strategies, as threatened species could be declining too rapidly to wait for complete documentation of the problem.

7 Variation in fisher skill is a major determinant of bycatch rates across species, gears, and fisheries

7.1 Abstract

Fisheries bycatch continues to drive the decline of many threatened marine species such as seabirds, sharks, marine mammals, and sea turtles. Management frameworks typically treat bycatch as an inevitable externality of fishing that can be mitigated with fleet-level controls on fishing practices and effort. Yet, individual operators have agency in how, when, and where to fish, and it is widely understood that some fishers are better than others at catching fish (the "skipper effect"). If operators differ in their ability to target species, it follows that they would also have differing abilities to anti-target what they do not want to catch. We analyse variations in threatened species bycatch between individual operators from five industrial fisheries in the Australian Commonwealth, representing different geographic areas, gear types, and target species. We find that the individual vessel is a significant predictor of bycatch for 15 of the 16 species-fishery interactions and is the most important factor driving variability in bycatch of several species. This pattern is evident across bycatch types with a range of avoidance incentives, including species that represent high costs to fishers (e.g. seabirds in longlines), low costs (e.g. sea snakes), and economic value as potentially targeted byproducts (e.g. hammerhead sharks). Encouragingly, we found high performance operators in all five fishing sectors, including gears that are major concerns for causing high bycatch mortality of a wide range of species globally (e.g., set gillnets and demersal trawls). Additionally, for some species, target catch is negatively correlated with bycatch, with a few low-profit operators generating the majority of the bycatch. These results indicate there is clear potential to improve the environmental performance of fisheries with incentive-based interventions that target specific performance groups within a fleet.

7.2 Introduction

Incidental catch of marine animals in fishing gear ("bycatch") has been recognized as a serious problem for several decades and despite widespread efforts to address it, bycatch remains one of the most pressing issues in fisheries management today (Soykan *et al.* 2008; Gray and Kennelly 2018). Bycatch of threatened, endangered, or protected species (TEPs)—such as sea turtles, seabirds, elasmobranchs, and marine mammals—has gained particular attention because it has been identified as a leading cause of many species declines (McClenachan *et al.* 2012; Dulvy *et al.* 2014; Lewison *et al.* 2014). The most common approaches to reducing TEP bycatch have been top-down, command-and-control measures (e.g., effort reduction, time/area closures, technology requirements, bycatch quotas) that are implemented across the entire fleet or industry, such as a total allowable catch for particular bycatch species (Smith 2012; Lent and Squires 2017). There has been some success in reducing TEP bycatch using these conventional approaches in some fisheries; for example, prohibiting tuna purse seine sets on dolphins (Gilman 2011), requiring turtle excluder devices (TEDs) in prawn trawls (Senko *et al.* 2014), and requiring bird-scaring lines in pelagic longline fisheries (Jiménez *et al.* 2020).

Unfortunately, these conventional approaches have been far from universally successful. They have often performed worse in practice than models and trials suggested, even when the same approach is translated to a similar fishery (Gladics *et al.* 2017). For example, TEDs have been much more effective in prawn fisheries in Australia compared to the U.S. (Cox *et al.* 2007). From an economic perspective, it is not surprising that these command-and-control approaches to bycatch reduction have had limited effectiveness because they can be slow to implement, reduce target catch, stymy innovation and customization to each context, and fail to encourage continuous improvement beyond the regulatory minimum (Wilcox and Donlan 2011; Lent and Squires 2017; Squires *et al.* 2018). Importantly, they can be prohibitively difficult and expensive to enforce (Cox *et al.* 2007; Innes *et al.* 2015).

Instead, economists have urged the adoption of incentive or market-based approaches to reducing bycatch, such as transferable bycatch allowances, taxes, bonds, and insurances (Innes *et al.* 2015; Lent and Squires 2017). Unlike traditional regulation, incentive-based approaches allow individual fishery operators more flexibility and autonomy to adjust their fishing practices in ways that best fit their vessel and crew, and are the most economically efficient (Holland 2007; Innes *et al.* 2015). Incentive-based approaches have been shown to reduce finfish bycatch in major fisheries such as the Alaskan Pollock fishery and the U.S. West Coast groundfish fishery, and have been effective in other contexts, including terrestrial wildlife conservation, energy use, and carbon (Lent and Squires 2017). Yet, they have rarely been applied to fisheries bycatch. One problem is a lack of understanding of these approaches and their potential cost-effectiveness (Lent and Squires 2017). Another problem is a broader lack of understanding of how fishers behave and react to different situations (Wilen *et al.* 2002; Fulton *et al.* 2011; Van Putten *et al.* 2012).

Although most fisheries management frameworks remain focused on command-and-control measures that are implemented at the level of a fishing fleet, several studies suggest that the skill of individual fishermen (the "skipper effect") could be a driver of important and unexplained variations in fishing efficiency. A skipper's skill is some combination of managerial ability, experience and knowledge of the environment, ability to respond to rapidly changing information and conditions at sea, and numerous other factors (Squires and Kirkley 2011). It is almost impossible to describe and record the many decisions a skipper makes before, during, and after a fishing trip, even if there were an observer dedicated to the task (Wilen *et al.* 2002). Other important factors, such as vessel size and characteristics and time spent fishing, can be difficult to separate from the skipper's "skill" (e.g., the decision to use one type of equipment over another, or how much time to spend fishing that day) (Lokina 2009; Squires and Kirkley 2011). There is ongoing debate about the key components of operator skill and its importance in different contexts. For instance, whether skipper skill is equally important for all fishing methods and whether the magnitude of the effect has been reduced by technological advancements (Hilborn 1985; Russell and Alexander 1996; Viswanathan *et al.* 2002; Tidd *et al.* 2017). However, numerous studies show consistent variation in target catch rates among anglers, skippers, or fishing vessels that is not explained by environmental variables or economic inputs (e.g., Hilborn 1985; Gaertner *et al.* 1999; Marchal *et al.* 2006; Vázquez-Rowe and Tyedmers 2013). This includes technically advanced fisheries where a skipper's skill would seemingly be less important, such as the US menhaden purse seine fisheries, which have similar vessels with similar equipment owned by the same company (Ruttan and Tyedmers 2007).

Previously, the skipper effect has been explored in relation to fishing efficiency and profitability (effort and target catch). However, if fishers have differing abilities to catch what they want to catch, it follows that they would also have variable skill at avoiding things they do not want to catch. The skipper effect is relevant to any management action pertaining to the efficiency of the fishery. Thus, if it is present, it is important to consider in the development of strategies to reduce fishing impacts on threatened species. Increasing voluntary compliance and bycatch avoidance behaviours would greatly reduce enforcement costs and impacts on biodiversity, but incentivizing behaviour change requires an understanding of the behaviour of the individuals (Sutinen and Kuperan 1999; Stern 2000). To incentivize behaviour change around TEP bycatch specifically, we need to understand the ability of individual fishers to avoid bycatch if they are inclined to do so.

Untangling the skipper effect is difficult without very detailed data, which are often not available for target catch and is extremely rare for bycatch. Here, we capitalize on a rare opportunity to compare multiple high-resolution fisheries datasets that have information about both target and bycatch. We use fisheries observer data from five Australian Commonwealth fisheries sectors to answer three key questions: 1) Is there significant and predictable variation among operators in their target to bycatch ratios? 2) If so, does the pattern hold across gear types and fisheries? and 3) Do bycatch species differ in their avoidability in a consistent way? We hypothesize that there are characteristics at the operator level that lead some operators to have worse performance than others on a consistent basis and that operator skill is an important factor driving variations in bycatch across fishing fleets. Secondly, we hypothesize that, irrespective of the gear and fishery, there are high performing operators that are able to avoid bycatch while maintaining high target catch. Finally, we expect there is a spectrum of bycatch avoidability across different species and fishing methods. Overall, if the patterns we hypothesize hold true, then there is untapped potential to reduce bycatch without imposing additional controls on fishing effort and gear. This would support an alternative approach to framing management questions such as those around threatened species bycatch. It may be that it is not a random event across a fishery, but in fact is an issue of particular low performance operators. In this case, measures aimed directly at those individual operators could be an opportunity to make considerable progress towards reducing threatened species bycatch, at potentially much lower cost than common whole-of-fishery solutions.

7.3 Results

To explore patterns in bycatch among individual fishing vessels, we analysed 17,030 fishing events ("shots") from 297 vessels between 2001 and 2017. The observer datasets are from five Australian Commonwealth fisheries with different gear types or geographic areas: Northern Prawn Fishery ("prawn trawl"), Eastern Tuna and Billfish Fishery ("tuna longlines"), set gillnets, demersal longlines, and otter bottom trawls. The latter three are gear-based sub-sectors of the Southern and Southern and Eastern Scalefish and Shark Fishery (SESSF). In all five fisheries, there was considerable heterogeneity among vessels in their bycatch to target catch ratios (Figure 7.1). Several operators with the highest average target catch had some of the lowest average bycatch rates, and conversely the highest bycatch rates were from operators with lower target catch. However, the relationship was not consistent, with a slightly negative correlation for most species in the set gillnets, a positive correlation for seabirds in the demersal longlines, and no clear correlation for other species and fisheries.

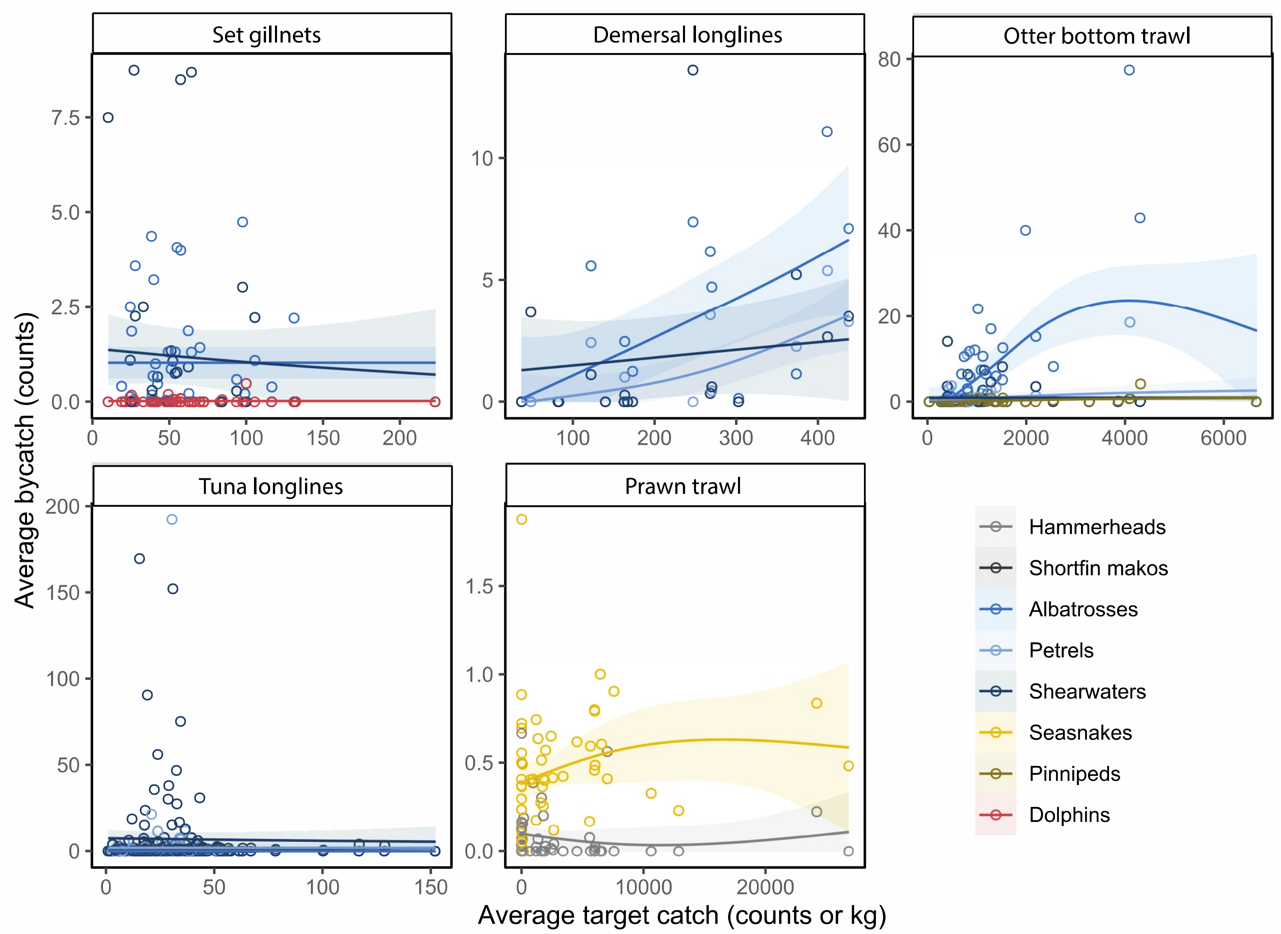

Figure 7.1: The relationship between bycatch and target catch per individual vessel for species in five fishing sectors. The data are fit with GAMs and the shaded area shows 95% confidence intervals

We used Generalized Additive Models to test which environmental factors and factors related to operator skill drive the variations in bycatch. We compared the individual vessel as a fixed versus a random effect. The random effect allows us to tell whether differences among vessels capture a significant amount of variation in the bycatch rates in the data. However, it does not focus on testing the performance of individual vessels. Moving to a fixed effect allows us to estimate a bycatch rate for each vessel, and thus identify which specific vessels have high rates. However, the fixed effect is very data hungry, as it requires estimation of a coefficient for each vessel, instead of estimating the population level variation as in the case of the random effect. The individual vessel (either as a fixed or random effect) was significant in 15 out of 16 species-fishery models, and explained anywhere from 5 to 67% of the expected deviance in those models (Table 7.1). There is no way to directly quantify the effect size of each GAM parameter; therefore, to indicate the relative importance of each variable in explaining the variation in bycatch, we first calculated the difference in the deviance explained by the best model with and without the vessel. We then estimated the importance of each variable from the models in the dredge analysis using the importance function (which sums model weights for each variable across all combinations) from the mumin package (Fisher *et al.* 2018). The individual vessel had the highest (or tied for the highest) importance score for 14 of the 16 models (Table S5.1).

Overall, the models performed well and explained anywhere from 5 to 95% of the deviance in bycatch. The models with the most unexplained deviance were albatrosses and shearwaters in the demersal longlines and set gillnets, and shortfin makos in the tuna longlines (Table 7.1). After the vessel, year was the second most important factor (judged by the importance estimates and frequency of occurrence in the best model). This is expected because there were substantial changes in the regulation of fishing practices and fleet structure in all sectors over the time period, as well as changes in the availability of bycatch species. Seasonal and geographic (latitude and longitude) factors were significant for most species as well (12 and 13 models, respectively).

Table 7.1: Significant predictor variables for the best models for 16 species-fishery interactions. Vessel (vsl) was included as a random effect unless specified as fixed (fe). Delta deviance explained is the difference in deviance between the best model and with the vessel parameter removed. Trgt catch = target catch as volume or number of individuals. Trgt clust = targeting cluster. Op. type = fishing operation for pelagic longlines (e.g., standard operations or bycatch mitigation trial). % in light = percent of shot in daylight. Shot dur = duration of shot.

Model	Trgt catch	Year	Mnth	Lat/ Lon	Trgt clust	Dpth	Op. type	% in light	Shot dur.	Vsl	Dev. %	Delta Dev.
Set gillnets												
Albatrosses				x		x	--	--	--	**x**	20.0	15.0
Shearwaters		x	x	x			--	--	--		12.0	0.0
Dolphins	x	x		x			--	--	--	**x**	72.3	66.5
Demersal longlines												
Albatrosses	x	x	x	x		x	--	--	--	**x**	27.0	27.0
Petrels	x	x		x	x	x	--	--	--	**x**	44.2	9.4
Shearwaters			x				--	--	--	**x**	16.1	16.1
Otter bottom trawl												
Albatrosses	x	x	x			x	--	--	--	**x**	51.3	14.3
Petrels	x	x					--	--	--	**x**	70.3	25.5
Shearwaters	x	x	x	x			--	--	--	**x**	66.8	16.2
Pinnipeds	x		x	x		x	--	--	--	**x**	46.3	15.8
Tuna longlines												
Albatrosses		x	x	x		--	x	x		**x**	52.3	9.2
Petrels		x	x	x		--	x	x		**x**	84.1	13.8
Shearwaters		x	x	x	x	--	x	x		**x**	82.5	9.6
Shortfin mako	x	x	x	x	x	--	x	x	x	**x(fe)**	25.3	5.2
Prawn trawl												
Hammerheads		x	x	x		x	--	--	--	**x(fe)**	95.4	35.2
Seasnakes	x	x	x	x	x		--	--	--	**x(fe)**	84.8	62.2

The association between bycatch and target catch was variable (Table 7.1). Target catch was included in 9 of the 16 best models, including all four bycatch groups in the otter bottom trawls (albatross, shearwaters, petrels, and pinnipeds), dolphins in set gillnets, shortfin makos in the tuna longlines, and two of the three seabirds in the demersal longlines. Target catch was a significant predictor of sea snake bycatch in the prawn trawl, which was unexpected as prawns are not known to be primary prey for sea snakes (Fry *et al.* 2001). Surprisingly, target catch was not included in the best model for the species most known to associate (e.g., seabirds and tuna, where fishers often use seabirds to locate the tuna), although it was close to significant and was suggested in some of the top models. This could be explained by shifting fishing practices to avoid seabird bycatch, such as adoption of bird scaring lines, night setting, and area closures (Commonwealth of Australia

2018). The bycatch that was most clearly not associated with target catch was hammerheads in the prawn trawl.

Tactical factors that were significant for some bycatch contexts included targeting cluster, which represents fishing tactics that are not easily described or are not directly recorded in the data (e.g., bait type, orientation of gear), and set duration and type of operation (e.g., standard fishing activities versus gear modification trials) in the tuna longline fishery. Some of the target clusters corresponded with known dynamics in the fisheries, such as targeting tiger prawns in different areas, months, and with a different net configuration than for shots targeting banana prawns. Targeting clusters are not well understood for the extremely multi-species SESSF fisheries. However, the clusters did capture a known dynamic in the otter bottom trawl, where highly targeted trawls are aimed at single species aggregations (e.g., orange roughy, *Hoplostethus atlanticus,* or blue grenadier, *Macruronus novaezelandiae*), whereas generalist shots are aimed at a wide variety of targets (Tuck *et al.* 2013). Although targeting cluster was only significant for four of the 16 models, some other factors related to fishing tactics also capture aspects of targeting, such as depth, location, and time of day. Several environmental factors were significant predictors of bycatch for certain species and fisheries, including time of day, depth, geographic location, and month. These factors are also related to operator skill because skippers make decisions about where and when to fish.

Our primary aim was to isolate the marginal effect of the individual operators that is not captured in tactical variables such as location and timing of fishing, while accounting for factors affecting the catchability of bycatch. We assessed the regression coefficients for individual vessels in each model to indicate the direction and strength of the relationship between the vessel and the amount of bycatch (Figure 7.2). The regression coefficients indicate that in each fishery, specific vessels are significant predictors of high bycatch shots, and others predictably have lower bycatch shots. The effect is more pronounced for certain species; for instance, petrels in the demersal longlines, otter bottom trawls, and tuna longlines, dolphins in the set gillnets, and to a lesser extent, pinnipeds in the otter bottom trawls and albatross in the otter bottom trawls and demersal longlines. Large gaps in the spread of regression coefficients indicate potential targeting behaviour. This pattern is evident for dolphins in the set gillnets and is fairly dramatic for hammerheads in the prawn trawls.

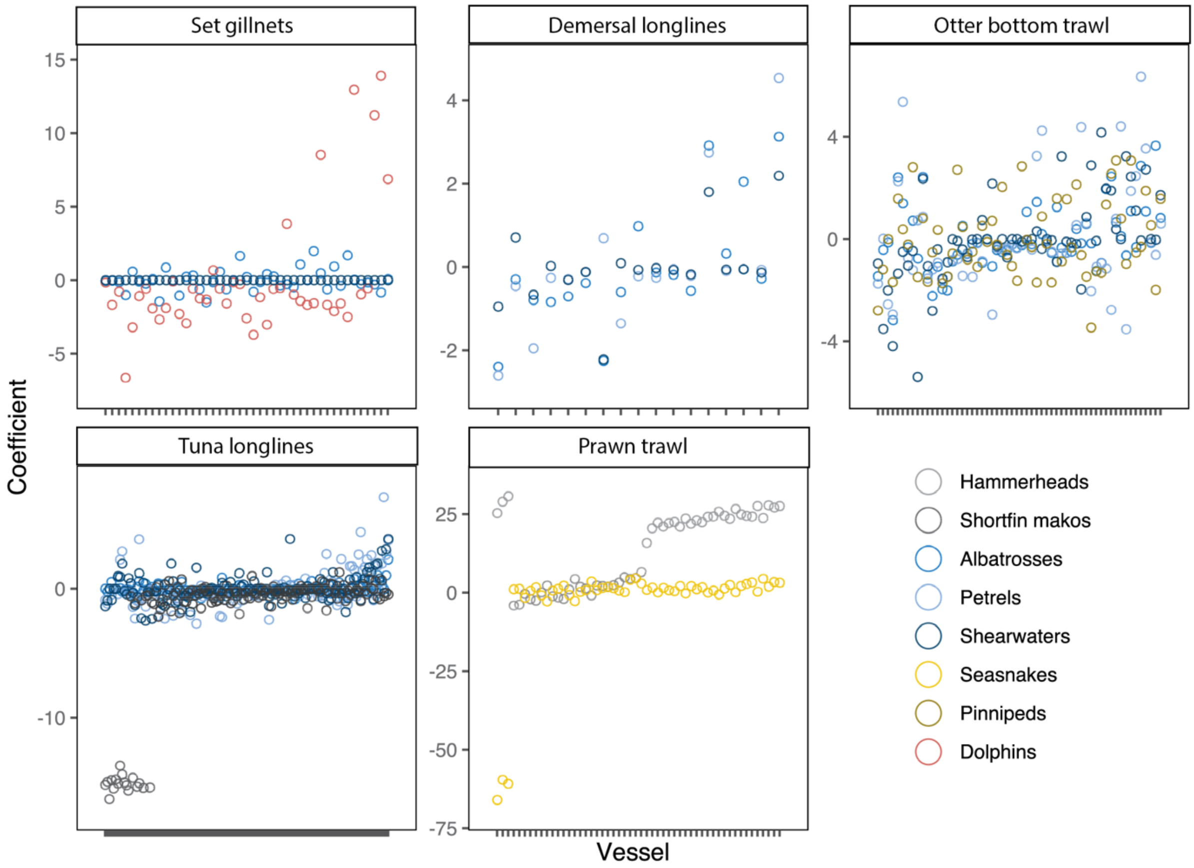

Figure 7.2: Regression coefficients for individual vessels (fixed or random effects) in the best models for species in the five fisheries. Vessels are ordered by ascending cumulative value of their regression coefficients across all species.

To indicate whether the variability among operators persisted over time, we explored operators' bycatch to target ratios over the timespan of the data. Overall, improvements in bycatch were variable across fisheries (Figure 7.3), although it is difficult to compare bycatch ratios between rare species and common species. Following a series of regulatory changes and bycatch mitigation programs, the observer data shows a dramatic reduction in seabird bycatch in the tuna longlines from a fleet-wide average of over 100 birds per shot in 2001 down to zero in 2015. These very high averages are likely inflated by bycatch mitigation trials in the early 2000s that were not normal operations, but logbooks and recent and electronic monitoring data corroborates a significant improvement in seabird bycatch overall (Phillips *et al.* 2010; Emery *et al.* 2019b). Shortfin mako catch rates in this sector were always much lower, but do not seem to decrease as seabird bycatch has. This is not surprising, as shortfin makos are a byproduct species with a catch limit per fishing trip. The SESSF sectors (set gillnets, demersal longlines, and otter bottom trawls) also underwent a series of regulatory changes related to bycatch (largely focused on the otter bottom trawls) (Tuck *et al.* 2013), and the most recent years of observer data indicates there may be some improvement in seabird bycatch. Compared to seabirds, cetacean and pinniped interactions are relatively rare and it is difficult to detect a trend in the observer data, but bycatch of these species remains a major concern (AFMA 2019b; Tulloch *et al.* 2020). There is no evidence of reduction in hammerhead bycatch in the prawn trawl, but sea snake bycatch rates seem to be decreasing. Most importantly, patterns in the observed bycatch ratios indicate that variability among operators persisted over time in all fisheries, which indicates that there remains opportunity for further improvement and reduction in rates of threatened species bycatch.

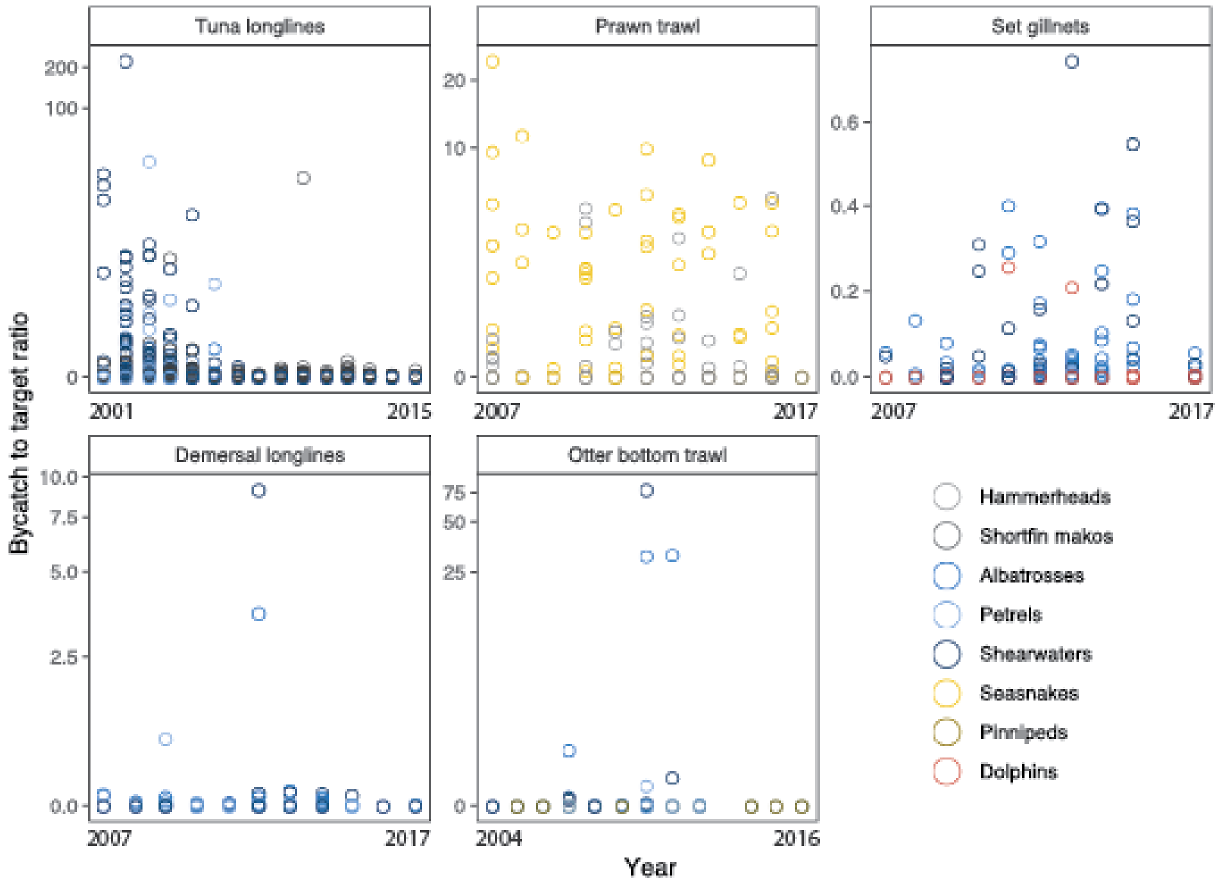

Figure 7.3: Average ratio of bycatch to target catch for individual vessels over time for five fishing sectors. Ratios are shown on a log transformed scale (except for set gillnets).

7.4 Discussion

Controlling for factors affecting bycatch availability, targeting tactics, and changes in fleet structure and management over time, we find that characteristics of the individual operators have a significant effect on bycatch levels across a range of species and fishing methods. We assume that the data provides an accurate representation of fishing activities, although there are biases and inconsistencies in observer data (Benoît and Allard 2009; Wakefield *et al.* 2018). However, we detect the pattern of operator variability over a relatively long period of time, and across five observer data sets that are known to have good accuracy (Kennelly 2020). Our results suggest that anti-targeting (avoiding) is a skill just as targeting is, and we posit three main drivers of the variable anti-targeting performance: 1) Anti-targeting may be inherently more difficult for some gears and species and therefore require greater skill; 2) Some bycatch is not very inconvenient, so there is little motivation to avoid it; and 3) There are incentives to catch some byproduct species, potentially making them clandestine targets. Notably, even in gears known to catch a wide range of bycatch species (e.g., gillnets and demersal trawls), we find that a small group of operators are able to simultaneously anti-target a range of different bycatch species, while still maintaining high target species catch. These high-performance operators present an untapped opportunity to greatly improve the environmental performance of fisheries, without necessarily mandating additional gear modifications or other command and control regulations.

The vessel effect in our analysis represents the unknown elements of operator skill and decision making that are not captured in other factors relating to fishing tactics, including managerial skills, knowledge of species or habitats, and ability to manoeuvre the vessel and haul gear. It might be that the low-bycatch operators are more conscientious about using their gear (e.g., TEDs in prawn trawls or bird scaring lines for pelagic longline sets), or that they have developed subtle innovations in their fishing practices, for instance, changing the depth or orientation of their gear in response to changing environmental conditions they observe at sea. Avoiding different types of bycatch (e.g., seabirds versus sharks) may demand different types of skills from operators. Observer coverage in the five fisheries was not sufficient for a comprehensive analysis of how individual operators performed across multiple bycatch species over time (especially for the rarer species). However, our results suggest there are several characteristic groups of operators in each fishery, although the delineation of the groups is less obvious in some sectors. There are some definite high performers that are skilled at avoiding multiple types of bycatch while maintaining high target catch, and a group of low performers with above average bycatch and below average target catch. In between these extremes are operators with low bycatch rates but also lower target catch rates, and in some cases (e.g. the demersal longlines), there is a group of operators with high target and high bycatch. Further exploration of individual vessels would be useful to detect operators that performed particularly well for certain species, but poorly for others. It may be that these are in fact skilled operators, but are more inclined to avoid certain types of bycatch.

There is a range of incentives to avoid different bycatch species, including safety hazards, damage to gear, loss of target species, or bycatch penalties, and some incentives may be more salient to fishers than others. There are also perverse incentives to catch some bycatch, such as species with market value. Our results suggest that both phenomena occur in the Commonwealth fisheries. For instance, the dramatic decrease in seabird bycatch in the tuna longlines suggests that bycatch mitigation measures were effective, and likely worked in tandem with changing attitudes within the fishery. There was a strong incentive to reduce seabird bycatch because they have no market value, cost time, and waste a hook that could have caught a tuna. Management measures further strengthened the inherent incentive to avoid catching seabirds by imposing a hefty financial penalty, where the region of the fishery with high bycatch rates was closed to fishing if the bycatch rate exceeded 0.05 birds per 1000 hooks (Trebilco *et al.* 2010). In contrast, seabird bycatch reduction in the SESSF sectors have been less successful (Phillips *et al.* 2010). This could be because the seabird bycatch mitigation equipment for otter bottom trawls, demersal longlines, and gillnets is more difficult to operate, or because there was less incentive to do so. Input controls were introduced in the SESSF (e.g., mandating the use of at least one approved bycatch mitigation device on trawls), but it was not coupled with the high bycatch penalty as in the tuna longlines (Tuck *et al.* 2013).

The significant variability in bycatch levels among operators suggests that incentives aimed at individuals could be more effective at reducing overall bycatch levels, while not punishing operators who are profitable and environmentally efficient (low impact on TEP species per unit of production). This is not how bycatch management measures are typically designed. In the tuna longline fishery, a small number of vessels were responsible for the majority of seabird bycatch, but the strict penalty is imposed across the fleet. This type of command-and-control measure can have unanticipated negative effects, at the macro scale. For example, regulations on sea turtle bycatch in the Hawaiian swordfish longline fishery resulted in a three-year fishery closure, which allowed less-regulated fleets from other countries to increase their effort and likely had a detrimental effect on overall sea turtle bycatch (Chan and Pan 2016). Although the input controls in the tuna longline fishery ultimately had very positive outcomes for seabirds, management measures directed at low performing operators could further reduce overall bycatch levels. Individual standards have been applied in a few cases, such as the multilateral dolphin conservation program for tuna purse seine fisheries in the Pacific, which assigns individual dolphin mortality limits in addition to other measures (Lent and Squires 2017). In response to increased reports of dolphin bycatch in set gillnets, a large area of the SESSF was completely closed to gillnet fishing in 2011, which significantly impacted the profits of the entire fleet. Recognising that this approach punished fishers who had avoided interactions and stymied incentives for individuals to innovate best solutions for their own vessels, the strategy was revised so that the maximum interaction rates (and the penalties for exceeding them) are applied to individual vessels (AFMA 2019a). A comprehensive report of the results has not been released, but at least one operator was temporarily banned from fishing after failing to comply (AFMA 2018).

However, even where these measures are aimed at individuals, a bycatch limit is essentially a quota that sets an acceptable level of species mortality, and thus would not be expected to drive bycatch rates to zero. Even where limits are set based on sustainability criteria, bycatch may still threaten the viability of seriously endangered populations (Komoroske and Lewison 2015). We found that variation among individual operators in their bycatch to target catch ratios persisted over time, even as regulatory conditions changed and many low-performing operators exited the fisheries (Mobsby 2018). This suggests there remains latent potential to reduce bycatch to very low levels while still maintaining target catch. Once managers understand the variability and role of individuals within fishing sectors, an important next step is to use that knowledge to design interventions that encourage continued innovation towards zero threatened species bycatch. These positive incentives (often in combination with some sort of penalty) have been successfully applied to bycatch in a few fisheries (Lent and Squires 2017), and have also been successful for other environmental problems, such as littering and marine debris (Hardesty *et al.* 2015).

The appropriate combination of incentives and penalties will vary for different bycatch contexts. For instance, sea snake bycatch may not incur enough costs or trigger social norms adequately enough to lead fishermen to avoid compared to sawfish or sea turtles, and thus may be an issue primarily of lack of effort as opposed to lack of skill. Bycatch that associates with target species, such as dolphins in the gillnet fishery, may elicit a stronger response to environmental social norms but could require more ingenuity and skill to avoid. There may also be rare bycatch incidents that are truly accidental and unpredictable; for example, there was one blue whale entanglement in the demersal longline sector over ten years of observer data. However, our results indicate this is not the norm, and that fishers do possess untapped knowledge and innovation in reducing threatened species bycatch, even for unselective or passive gears and for bycatch that associates with target species. Sea snakes and dolphins both associate with target species and are caught in high-bycatch gears (trawls and gillnets), yet there was significant variability among operators that explained a large proportion of the deviance in bycatch rates of these species.

Understanding the incentives and behaviours underlying bycatch contexts is especially pertinent for byproduct species that have value in legal or illegal markets. We found evidence of targeting (and anti-targeting) in the tuna longline fishery for shortfin makos, which is permitted but regulated, and in the prawn trawl for hammerheads, which are not supposed to be targeted. Elasmobranch bycatch is especially complex because of their market value, which can change dramatically due to shifting demand, access to markets, and regulations (Oliver *et al.* 2015). Shark targeting behaviours and dynamics require more in-depth analysis, but our results indicate that skilled fishers are able to both target and avoid a range of species simultaneously. This information could help managers identify where accidental bycatch may in fact be targeting.

Identifying high and low performing vessels with respect to bycatch is an important first step towards designing effective management actions. There has been some progress using statistical approaches to glean more information from catch data in order to standardize catch rates and detect both good and bad anomalies (e.g., Zhou *et al.* 2019; Parsa *et al.* 2020). Ideally, managers would have better quality data to work from, and electronic monitoring systems—which are gaining traction in industrial fisheries globally—are a major step towards more informed and effective fisheries management (Helmond *et al.* 2019). Better monitoring will also help managers understand the impact of fishing on bycatch species, which is often poorly known, and what species and populations merit the most concern (Moore *et al.* 2013).

Ultimately, the goal is to move from identifying patterns of high and low performing vessels, to understanding the underlying processes, and using that knowledge to inform actions. Insights into the biophysical drivers of catch and bycatch (e.g., sea surface temperature, frontal systems, isothermal layer depth) likely help explain some aspects of how high-performing operators are fishing (Scales *et al.* 2017). However, certain elements of operator skill—such as managerial skills or reacting to dynamic conditions at sea—are not captured in biophysical variables or in data from logbooks, observers, or electronic monitoring. Therefore, it is essential that management and research institutions collaborate directly with fishers to understand the more nuanced skills and behaviours that characterize good operators, and how to spread that optimal performance across the fishery (Johnson and Van Densen 2007; O'Keefe *et al.* 2014). This level of individual engagement is expensive and time consuming but would be a worthwhile investment in the long term. Enforcement is the largest expense for fisheries management globally, and increasing voluntary compliance would greatly reduce those costs (Arias 2015; Mangin *et al.* 2018). In this context, voluntary compliance could mean shifting from bycatch limits and technology requirements with an underlying enforcement program to a focus on innovation at the individual level, supported by incentives. Our results suggest that some fishers already voluntarily avoid bycatch of species that do not incur a penalty or major cost to their fishing operations, and are able to do so without compromising their economic performance. The appropriate set of incentives and management interventions could encourage further innovation from fishers, and potentially improve bycatch rates beyond what currently seems feasible. The importance of variable skills and behaviour of individual operators could extend beyond threatened species bycatch to management of other environmental impacts, such as gear abandonment and waste discharge. Although fisheries operators are notoriously resistant to change (Eayrs *et al.* 2015), the current climate of environmental and socioeconomic uncertainty could be an opportunity for a transformation in global fisheries. Increased uptake of bycatch avoidance skills and other positive environmental behaviours across fishing fleets would be a major gain for management agencies and for biodiversity at a pivotal moment in the trajectory of ocean sustainability.

7.5 Materials and Methods

7.5.1 Description of fisheries and datasets

We use observer data provided by the Australia Fisheries Management Authority for five federally managed fishing sectors in Australia: Northern Prawn Fishery ("prawn trawl"), Eastern Tuna and Billfish Fishery ("tuna longlines"), and three sub-sectors of the Southern and Southern and Eastern Scalefish and Shark Fishery (SESSF), referred to here as demersal longlines, otter bottom trawls, and set gillnets (Figure 7.4).

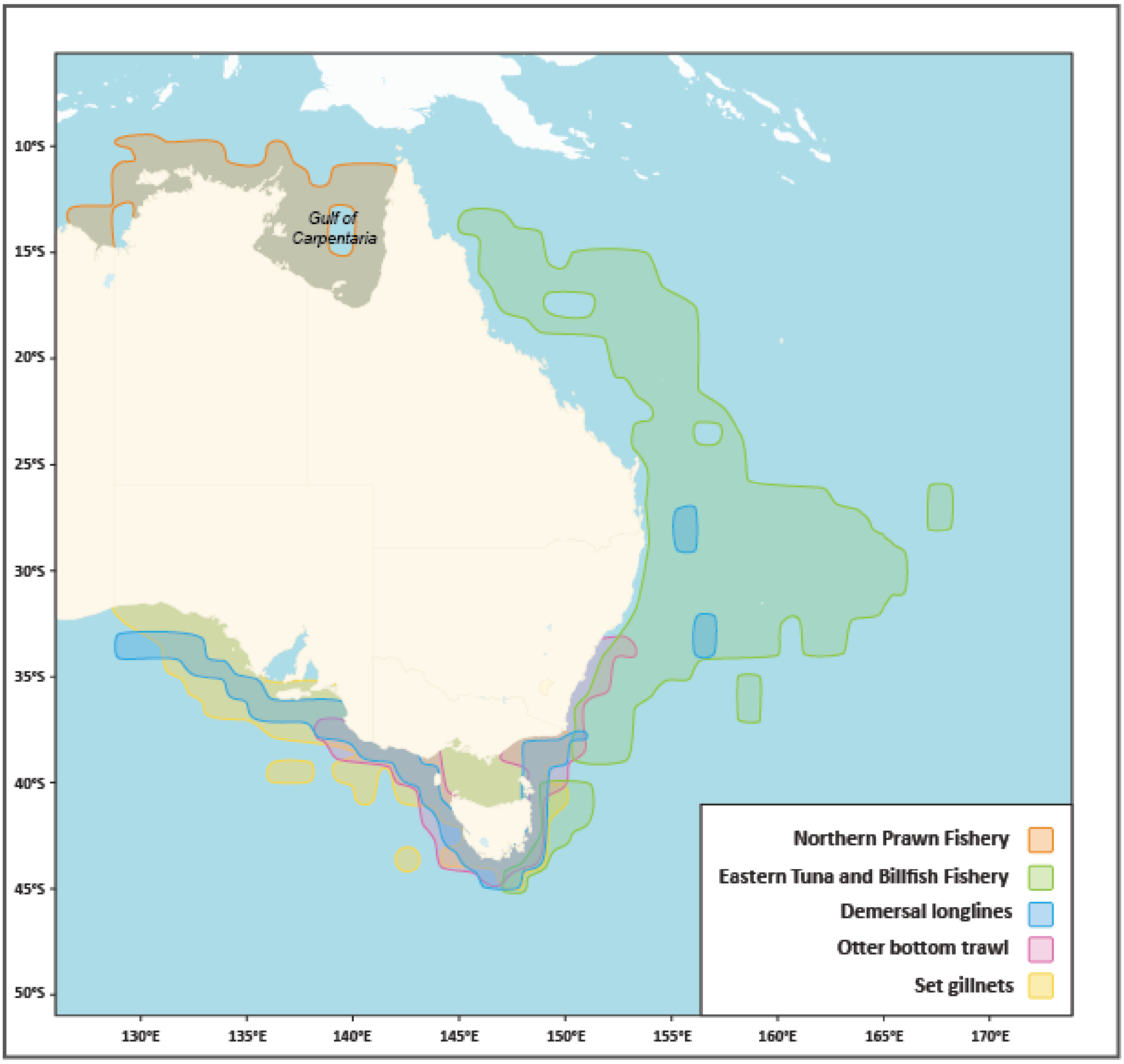

Figure 7.4: Map of the five Australian Commonwealth fisheries and their general areas of operation shown in reports from the Australian Fisheries Management Authority

7.5.1.1 Northern Prawn Fishery (prawn trawl)

The Northern Prawn Fishery extends across most of northern Australia and is the country's most valuable trawl fishery. It is essentially two distinct fisheries; a banana prawn fishery and a tiger prawn fishery, which operate during different time periods and in mostly distinct regions of the management area, and also use slightly different types of trawl gear (Brewer *et al.* 2006; Pascoe *et al.* 2012). White banana prawns (*Fenneropenaeus merguiensis*) are mostly caught during the day on the eastern side of the Gulf of Carpentaria offshore from mangrove forests, where they form dense aggregations ("boils") near the surface that are often located using spotter planes (Figure 7.4). Redlegged banana prawns *(F. indicus*) are mainly caught in the western region of the management area (Patterson *et al.* 2017), whereas tiger prawns (mainly *Penaeus esculentus* and *P. semisulcatus*) are usually caught at night closer to the seafloor and near coastal seagrass beds in the central portion of the management area (Patterson *et al.* 2017). Endeavor prawns (*Metapenaeus endeavouri* and *M. ensis*) are mainly a byproduct caught along with tiger prawns (Patterson *et al.* 2017).

Prawn trawls are known to have high environmental impact, including high bycatch rates (Kelleher 2005). Yet, the Northern Prawn Fishery received Marine Stewardship Council accreditation in 2012, largely due to the extensive effort to incorporate a range of biological and bioeconomic models into an active management framework (Pascoe *et al.* 2017). The fishery has been restructured over several decades through a series of management measures and buyback programs of less-efficient vessels, including a reduction from about 250 to 50 vessels (Patterson *et al.* 2017). In 2000, the prawn trawl introduced the compulsory use of approved turtle excluder devices (TEDs) and bycatch reduction devices (BRDs), allowing operators to select their desired combination of devices (Brewer *et al.* 2006). Overall, the TEDs and BRDs have substantially reduced catches of larger animals such as sea turtles and large elasmobranchs—although sawfish are an important exception—but have been much less effective for smaller animals such as sygnathids (seahorses, pipefish and sea dragons) and sea snakes (Gourguet *et al.* 2016; Abrantes *et al.* 2020). The scientific observer program covers about 2% of fishing days (Laird 2020).

7.5.1.2 Eastern Tuna and Billfish Fishery (tuna longlines)

The Eastern Tuna and Billfish Fishery is a pelagic longline fishery operating year-round in the EEZ and adjacent High Seas off Australia's East Coast (Figure 7.4). The main targets are yellowfin (*T. albacares*), bigeye (*T. obesus*), albacore (*T. alalunga*), and southern bluefin tuna (*T. maccoyii*), and broadbill swordfish (*Xiphias gladius*) (Emery *et al.* 2019a). Structural readjustments and new harvest strategy policies over the past two decades have reduced the number of vessels from 150 to about 40 currently active vessels, with the more economically efficient vessels remaining (Mobsby 2018). Several management interventions have aimed to reduce bycatch of protected species (seabirds, sea turtles, and marine mammals); for example, requirements to carry line cutters and de-hookers, use bird-scaring lines, and deployment of gear at night (Mobsby 2018). Wire leaders were banned in 2005 to reduce shark bycatch, although vessels are permitted to retain up to 20 individuals per trip—meaning they are actually byproduct as opposed to bycatch (Mobsby 2018). Seabird bycatch mitigation has been very successful but there is still concern about catch of other species, such as shortfin mako sharks (*Isurus oxyrinchus*), which were recently upgraded to Endangered on the IUCN Red List and are the most common protected species caught in the tuna longline fishery, and leatherback turtles, which are much rarer occurrences but are listed as Critically Endangered in the Western Pacific (Mobsby 2018; IUCN 2020). The tuna longline fishery has had a scientific observer program since 2001, which has ranged from 3.5-8% of fishing effort (Kirby and Ward 2014).

7.5.1.3 Southern and Eastern Scalefish and Shark Fishery (SESSF)

The SESSF is a multispecies, multigear, and multisector fishery with a management area covering almost half of Australia's fishing area and has the largest catch volumes of any Commonwealth fishery (Mobsby 2018). Many SESSF stocks were overfished (and some remain overfished); thus, it was one of the first fisheries targeted by the Commonwealth government's structural adjustment programs to reduce fishing effort and improve economic efficiency (Mobsby 2018). Overall, observer coverage has increased since the program was implemented in 1992, with required coverage varying according to the sub-sector and area (e.g. 100% observation is required near certain marine mammal colonies and closure areas) (Emery *et al.* 2020).

We focus on three gear types used in SESSF fishing sub-sectors: bottom set gillnets, otter bottom trawls, and auto-demersal longlines (referred to here as "demersal longlines"—"auto" refers mainly to how the hooks are baited) (Figure 7.4). The gillnet sector mainly targets sharks—primarily gummy sharks (*Mustelus antarcticus*), sawsharks (Pristiophoridae), and elephant fish (Callorhinchidae)—whereas the otter bottom trawls predominantly target eight teleost species or genera and the auto-demersal longline subsector primarily targets four deep-water teleosts (Wayte *et al.* 2007; Zhou *et al.* 2011; AFMA 2020). However, all three sectors catch and retain hundreds of other teleosts and elasmobranch species, most of which are not directly monitored or managed under a quota system (Zhou *et al.* 2011). In addition to these byproduct species, the SESSF sectors also catch a variety of protected species groups, including marine mammals (cetaceans and pinnipeds), seabirds, large sharks (e.g., shortfin makos and hammerheads, *Sphyrna* spp.) and sygnathids. Bycatch of pinnipeds and cetaceans is frequently cited as a major environmental concern for the SESSF (Tuck *et al.* 2013; Woodhams *et al.* 2020).

7.5.2 Fisheries observer data

The observer programs were instated at different times for the five fisheries. We obtained scientific observer data from 2001-2015 for the tuna longlines, 2007-2017 for the prawn trawl, 1992-2017 for the set gillnets and demersal longlines, and 1992 to 2016 for the otter bottom trawl. The scientific monitoring program for the latter three sectors was originally designed to collect data on target species, and the focus only expanded to TEP species in the early 2000s (Bergh *et al.* 2009). Thus, we excluded the early years from the analysis because almost no bycatch records appeared in the observer data prior to 2007 for the demersal longlines and set gillnets, and prior to 2004 in the otter bottom trawl. Since 2015, electronic monitoring systems are slowly replacing at-sea observers in these Commonwealth fisheries.

In order to account for species-specific dynamics that affect bycatch availability, we maintained the highest possible taxonomic resolution in the analysis of bycatch. Species-level identification by scientific observers is generally accurate for easily identified species (e.g., shortfin makos) and to the genus or family level for common species (e.g., shearwaters), but is less reliable for rare or similar looking species (e.g., different species of shearwaters) (Trebilco *et al.* 2010). We identified candidate bycatch groups of seabirds (albatrosses, petrels, and shearwaters), elasmobranchs (shortfin makos, hammerhead and winghead sharks, and sawfish), sea turtles, syngathids, and marine mammals (pinnipeds and dolphins). The majority of the dolphin bycatch records are for common dolphins (*Delphinus delphis*), and the pinnipeds are primarily Australian fur seals (*Arctocephalus pusillus doriferus*).

7.5.3 Statistical Analyses

7.5.3.1 Targeting cluster analysis

Fishers in multispecies fisheries often use different fishing tactics to target subgroups of targets species (Zhou *et al.* 2019). Sometimes the targeting practices are well-understood by fisheries managers (e.g., in the ETBF, swordfish are targeted with shallow night sets, often using fluorescent sticks attached to the lines) (Campbell and Young 2012). These different tactics affect the catchability of bycatch species but can be difficult to define and record. We used model-based clustering of the target species recorded in the observer data to define subgroups of target species and assign a targeting cluster to each fishing event. The cluster analysis was done in the R statistical language (R Core Team 2019) using the mixtools package (Benaglia *et al.* 2009), which uses a mixture of beta distributions to describe the probability of each target species occurring in a single fishing event. An advantage of the mixtools infrastructure, compared to other tools for finite mixture modelling, is that it considers the ratio of target species counts in each shot, as opposed to just the frequency of each species. We fit the mixture model using the expectation-maximization (EM) algorithm, limiting it to a maximum of 15 clusters, and compared models of increasing complexity, selecting the model that corresponded to the first minimum in AIC values (Peel and McLachlan 2000). For the SESSF sectors, which have many targets, we selected candidate target species first by selecting the 15 species with the highest total catch volumes and then by the most non-zero catches (how frequently that species is caught). We compared the AIC values to select the cluster model that best describes the data. We then used the best fitted model to classify each fishing event as one of the targeting types, assigning it randomly in the case of ties.

7.5.3.2 Exploring the relationship between catch and bycatch

For the measure of target catch, we used the sum of the number of individuals of the target species from each shot. For the SESSF sectors, which do not have a defined list of targets, we used all retained catch as the target catch (recorded as number of individuals for the set gillnet sector and as weights for the otter bottom trawl and demersal longlines). For the ETBF, we included only the five main target species (albacore, bigeye, yellowfin, and southern bluefin tuna, and broadbill swordfish) in the count of target catch. We combined the retained and discarded shortfin mako catch because they are a byproduct species. All bycatch is recorded as counts. Our focus was on exploring whether operators could avoid bycatch interactions altogether; therefore, we measured bycatch as animals that interacted with the gear but escaped as well as animals that were caught (this mostly applies to seabirds).

To explore the relationship between catch and bycatch, we first examined the data graphically using and used a generalised additive model implemented in the mgcv package in R (Wood 2015). This exploratory analysis indicated different relationships between bycatch and target catch depending on the species and fishery. In most cases the relationship appeared to be monotonic, but not always linear or in the same direction. For some species-fisheries interactions, there was no evidence of a correlation between target catch and bycatch. To evaluate the factors driving variations in bycatch, we used a generalized additive model (GAM) with a Tweedie distribution, which are good for handling very zero inflated data because they are a mixture of Poisson and Gamma distributions (Shono 2008). We incorporated the targeting type as a factor, as well as environmental and tactical factors that could affect the availability of bycatch, including year, month, depth of the fishing activity, latitude, longitude, and their interaction, time of day (percent of the shot that was in daylight), and whether it was a standard fishing trip or an experimental project (such as testing bycatch mitigation technologies). Not all variables were available or relevant to all fisheries. Each model included an offset for fishing effort, measured as thousands of hooks deployed for the tuna longlines and the duration of the fishing event for the other fisheries (number of hooks was not available for the demersal longlines).

We used a series of steps to select the best model. First, we compared two global models—with all factors included, along with a term for the vessel, as either a fixed or a random effect—to a null model of each bycatch species. We used the dredge function from the mumin package in R (Barton 2015) to compare all possible combinations of factors in the best global model (with vessel and observer as either fixed or random effects), then selected the model with the lowest AIC as the best model. If there were multiple models within 2 AIC units, we selected the simpler one with fewer factors, or the lowest AIC score if they had the same number of factors. We assessed the final model to verify the data were not over-dispersed and that the model captured the important patterns in the data. We excluded several species groups due to rarity of bycatch records: sea turtles in the tuna longlines and prawn trawl, marine mammals in the tuna longlines, sygnathids in the otter bottom trawl, and sawfish in the prawn trawl. The final analysis included 16 models of species or species-groups for the five fisheries.

8 Conclusion

8.1 Overview

The aim of this thesis was to investigate the threat that large-scale fishing poses to marine biodiversity at multiple geographic scales. I explored key gaps in our knowledge of fishing impacts that could present opportunities for conservation gains for marine species. I began with a broad global analysis of the political distribution of marine biodiversity (**Chapter 2**), then focused on the global seafood supply chain and the conservation status of seafood species (**Chapter 3**). I then presented an example of an important regional-scale gap in fisheries management in a data-poor biodiversity hotspot (**Chapter 4**), proposed an improvement to a widely used ecological risk assessment tool (**Chapter 5**), quantified and compared risks from multiple fishing sectors across different groups of megafauna species (**Chapters 6**), and, at the finest spatial scale, analysed variations in the performance of individual fishers with respect to threatened species bycatch (**Chapter 7**).

I found that transboundary collaboration is relevant to the protection of a much broader suite of marine species than is typically considered in multinational conservation instruments, especially poorly studied taxa such as many invertebrates, plants, algae, and deep-sea animals (**Chapter 2).** However, many large and well-studied species are also in a form of 'conservation purgatory' because of their value as seafood commodities, and there are various mismatches between the focus of marine conservation efforts and the scale of impacts of global seafood consumption on biodiversity (**Chapter 3**). Catch monitoring and documentation remains a serious challenge, and many regions still lack basic information about how many boats are fishing and what species they catch (**Chapter 4**). Estimates of risk to biodiversity that draw on expert elicitation can be improved by replacing binned categorical scores with ranked probabilities **(Chapter 5).** Many species of conservation concern have high risk of capture and mortality from multiple industrial scale fishing sectors, including species that do not appear in available catch reports (**Chapter 6).** The role that individual fisheries operators play in driving fishing impacts on threatened species is generally overlooked in management frameworks, and they present an important opportunity to improve the environmental performance of fisheries (**Chapter 7**).

In general, I found that marine biodiversity conservation is plagued by some rudimentary but persistent problems across a variety of scales. Compared to land, the ocean is relatively unexplored by ecologists and basic abundance and distribution information is lacking for the vast proportion of marine species (Crespo *et al.* 2019). This is directly relevant to biodiversity conservation and fisheries management because the intricately connected and dynamic nature of many marine ecosystems—and the threats they face—makes them difficult to protect effectively with terrestrial conservation paradigms (Carr *et al.* 2003). As a result, many species are not adequately protected to curb population declines (Klein *et al.* 2015; Jones *et al.* 2020). Despite these complex challenges, important opportunities for management and conservation actions exist at many different scales and in many different forms, from mathematical methods for making more informed conclusions from limited data, to broad international policy instruments. These findings highlight some of the opportunities to improve conservation outcomes and provide baseline information to inform management actions, including information about the distribution of marine biodiversity and the conservation status of exploited species, fisheries management priorities in an understudied region and an improved method for assessing risk, and a novel perspective for approaching mitigation of threatened species bycatch. In this concluding chapter, I provide a synthesis of how this work advances our knowledge of fisheries management and biodiversity conservation. I examine the limitations of my methods and findings, reflect on the implications of each chapter for reducing fishing impacts on marine species, and highlight important areas for further investigation and exploration.

8.2 Key findings and significance for conservation and management

8.2.1 Political distribution of marine biodiversity and implications for international conservation instruments

In **Chapter 2**, I quantified and mapped the political distributions of marine species and showed that marine biodiversity is extremely transboundary, with the vast majority of species distributions spanning many nations. However, most international conservation initiatives are implemented by individual countries, with no requirements for multinational coordination, and existing multinational management mechanisms are limited to a relatively small number of iconic habitats, commercially exploited, or highly migratory species (Fidelman and Ekstrom 2012; Lascelles et al. 2014; Palacios-Abrantes et al. 2020). Maintaining ecosystem integrity across species' ranges is important even for small-range, non-migratory, or sessile species (Carr et al. 2017). There is growing recognition of the value of less visible or less charismatic species and the ecosystem services they provide, and the importance of protecting a wider suite of marine biodiversity (Coleman and Williams 2002; Worm et al. 2006). Therefore, the highly transboundary nature of marine biodiversity has significant implications for the design and implementation of international conservation goals, especially as nations focus on their individual contributions to global conservation targets such as the Convention on Biological Diversity. These findings show that it is essential to create mechanisms that facilitate the transfer of funds, technology, and capacity building across countries and regions. Given the enormous volume of the ocean and the limits of our current knowledge, it is clear that collaboration and data sharing must extend to sectors that have not been actively engaged in biodiversity conservation, such as the mining, shipping, and renewable energy industries (Maureaud et al. 2020). This work aims to provide a baseline for identifying priority regions, countries, and extra-transactional actors for more integrated and collaborative governance.

8.2.2 Advancing conservation of seafood species

In **Chapter 3**, I focused on one key threat (overfishing) and one group of species (commercially exploited fish and invertebrates) that are often overlooked in conservation frameworks. Public awareness and concern for fishing and seafood sustainability issues is increasing (Lam 2016), but research on fishing impacts on threatened species has focused on charismatic megafauna (e.g. sea turtles and whales) that are primarily caught incidentally in large-scale fisheries (Erisman *et al.* 2017). Little was known about the magnitude and extent of legal commercial fishing of threatened seafood species. The global fisheries supply chain is plagued by severe environmental and social issues but, encouragingly, there are instances where raising the profile of marine conservation issues has catalysed management actions and positive outcomes for biodiversity and for people (Hall and Mainprize 2005; Österblom *et al.* 2011; Hardesty *et al.* 2015). This work aimed to do just that: inventory the status of threatened seafood species from a global conservation perspective and the countries driving industrial scale catch and trade of those species. We have an opportunity to leverage consumer and corporate awareness and the power of emerging technologies to improve seafood sustainability across the fisheries supply chain, including in wealthy countries with actively managed fisheries (Probst 2019).

However, juggling economic and environmental objectives and collaborating across disciplines is not a trivial task, and outcomes have not always been optimal for biodiversity or for livelihoods. Ecolabels and sustainable seafood guides—which have emerged as common tools for communicating information about seafood—are an important example of the difficulties in engaging consumers in more sustainable seafood behaviours (Roheim *et al.* 2018). There has been pushback against some of the major seafood ecolabels labels, particularly the Marine Stewardship Council, essentially for greenwashing (Christian *et al.* 2013; Gutierrez *et al.* 2016). The Marine Stewardship Council was born from a collaboration between one of the largest conservation NGOs (World Wildlife Fund) and consumer goods companies (Unilever). I advocate for these cross-sectoral collaborations in **Chapter 2** (transboundary conservation and data-sharing), **Chapter 3** (catch documentation and seafood traceability), and **Chapter 4** (building capacity for fisheries monitoring and surveillance), but I recognise they are not always successful. These setbacks underscore the difficulty of balancing economic and conservation objectives in the context of the global seafood supply chain (Bailey *et al.* 2018; Roheim *et al.* 2018). Global initiatives and cross-sectoral working groups are gaining traction and have had successes in other marine conservation contexts like plastics and debris (Hardesty *et al.* 2015), but fishing is a particularly complex challenge. The temptation of shorter-term profits—to the detriment of marine ecosystem health and social equity—remains a fundamental obstacle to more sustainable fisheries. The power of consumer demand must be bolstered by structural changes across the fisheries supply chain, and these changes will require coordinated efforts from a variety of actors. Although I did not delve into any specific collaborative mechanism in depth, these chapters can help illuminate critical gaps in fisheries management and threatened species protections, and for what areas and species these coordinated actions are most needed.

8.2.3 Monitoring fisheries

I limited the analysis of global seafood trade to larger-scale fisheries because data are most available for those sectors, not because they are necessarily more important for reducing fishing impacts on threatened species. Extensive work has been done to date that emphasizes the necessity of managing (and defining) recreational and non-industrial fishing sectors, which are extremely valuable both from a cultural and a food security perspective and are expanding in many areas of the world (Grafeld et al. 2017; Pauly 2017). **Chapter 4** highlights one of these management gaps—tuna driftnet fisheries in the Indian Ocean—and introduces a case study of this extremely data-poor region and megafauna biodiversity hotspot (Roberson et al. 2019). I argue for the need to prioritize limited resources to address the most important gaps in monitoring and surveillance of fisheries globally, including fisheries that are not categorized as industrial. Improving monitoring and surveillance is a necessary precursor to achieving better enforcement and governance, which is essential to improving sustainability of fisheries (Pons et al. 2017; Burgess et al. 2018). Although technologies for monitoring fisheries and tracing seafood are advancing (Lewis and Boyle 2017; Kamilaris et al. 2019), endeavours such as Global Fishing Watch, electronic catch monitoring schemes, national Vessel Monitoring Systems, or block-chain based seafood traceability are impossible without some basic management infrastructure. Given the expense and difficulty of monitoring fishing activity, especially in developing countries, it is important to identify and prioritize the sectors that merit the most immediate assistance with capacity building. I also point out examples of poorly prioritized government spending on fisheries, particularly subsidies aimed at increasing fishing capacity without complementary spending on better monitoring and management (Sala *et al.* 2018b; Sumaila *et al.* 2019b).

8.2.4 Assessing risk in data-poor fisheries

Poor catch documentation is an issue even for valuable target species in technologically advanced fisheries, but arguably a more urgent problem for species that are not considered to be primary targets (i.e., byproduct or bycatch species) (Komoroske and Lewison 2015). Lack of reliable catch data leads to high uncertainty about fishing impacts and species' conservation statuses (Moore et al. 2013). Better monitoring technologies are available but realistically their widespread implementation is a long way off, especially in less developed countries and fisheries (Österblom 2014; Future of Fish 2015; Doumbouya et al. 2017). Ecological risk assessments (ERAs) are a commonly used tool in these data-limited contexts, and often incorporate expert knowledge where empirical data are not available. However, the calculations of risk are not always done in a mathematically robust or consistent way, which can lead to inaccurate conclusions and misunderstanding about the uncertainty of those estimates. In my fifth and sixth chapters, I propose an adaptation of an ERA and use a case study of megafauna bycatch in Indian Ocean tuna fisheries to demonstrate its utility. My results provide a spatially explicit estimate of the location and magnitude of the drift gillnet threat compared to the industrial purse seine and longline sectors, and indicate that all three fishing gears are likely impacting a much larger suite of species than existing data show. The adjusted ERA method allows better quantification of relative risk and the uncertainty of outcomes even using the same sub-optimal data, which is valuable for managers who need to prioritise the highest risk species and allocate scarce research and management resources.

These findings highlight opportunities for multi-taxa benefits by concentrating management efforts on particular high-risk areas where gillnet fishing is concentrated. However, these high-risk areas are dynamic in space and time, which adds considerable complexity to any area-based management measures in a region with very limited management capacity. Dynamic management measures such as move-on rules and seasonally transient protected areas would likely be more efficient and impose less cost on fisheries compared to static regulations (Runge et al. 2014; Dunn et al. 2016), but require more resources and management infrastructure and may be infeasible for Indian Ocean coastal countries. Protection of species with lower average risk but large ranges and high cumulative overlap with multiple fishing sectors (e.g., baleen and beaked whales) remains an additional challenge, and effective protection will require multilateral collaboration and coordination beyond the tuna fisheries sectors (Lascelles et al. 2014; Di Sciara et al. 2016).

The silver lining of this inauspicious management situation is that many ‘low-hanging fruits’ still remain for reducing bycatch impacts on species in the Indian Ocean. The door is wide open for a range of approaches that would all help advance bycatch mitigation, including controls on fishing effort, gear modifications, and lower cost catch and vessel monitoring systems. There is a particularly strong case for engaging with fishers in any of these pursuits, as much of the region's fishing effort is from socially or economically marginalized groups that could possess important knowledge and insight into strategies for confronting some of these problems (Bennett 2019; McCluney et al. 2019; Karnad and St. Martin 2020).

8.2.5 Policy portfolios for fisheries management

In **Chapters 2-6**, I explore management gaps at the level of international commitments, regional organizations, and individual nations. These co-occurring regulatory layers—although often uncoordinated and haphazardly applied—are still related and complementary in important ways. Although international treaties and agreements have been criticized for being too generic to prevent site-level environmental degradation (among other criticisms) (Agardy 2005), they have been surprisingly useful in fisheries management contexts. Broad international conservation commitments can set policies in motion that ripple all the way down to affect how an individual fishing vessel behaves at sea. For example, much of Australia's fisheries bycatch regulation is a result of the Environmental Protection and Biodiversity Conservation Act 1999, which was created, in part, to fulfil the country's obligations as a signatory of the CBD (Miller *et al.* 2018b). Ratification of international fisheries agreements (e.g., the UN Compliance Agreement and the Straddling Fish Stocks Agreement) seems to also have spill-over benefits for species not covered by the agreement, and improves countries' fisheries management overall (Melnychuk et al. 2021). In general, countries with more fisheries management measures have stronger management performance and less overfishing (Fulton et al. 2014; Melnychuk et al. 2021). In contrast to seafood ecolabels and certification schemes, which have diminished effectiveness when there are too many available (Gutierrez et al. 2016), more seems to be better in the fisheries management context. This underscores the importance of approaching threatened species bycatch and marine biodiversity conservation more broadly with multiple solutions at multiple scales.

8.2.6 The role of individual fishers in driving bycatch threats and solutions

Chapter 7 capitalizes on high-resolution observer data from Australian Commonwealth fisheries to explore fine-scale patterns in fishing impacts at the level of individual operators. It is generally accepted that there is a "skipper effect" that drives variability in operator profitability and performance regarding their target catch (e.g., Hilborn 1985; Squires and Kirkley 2011), but this phenomenon had not been rigorously tested for non-target catch. I find significant variability between operators and while the magnitude of the effect varies, the pattern occurs across geographic areas, types of bycatch species, and fishing sectors. Four of the five gears included in this analysis (shrimp trawls, otter trawls, pelagic longlines, and gillnets) have been highlighted as major concerns globally for their consistently poor bycatch performance and relative lack of improvement even with mitigation measures (Lewison et al. 2014; Savoca et al. 2020). Importantly, my results identified individuals in each of these high-impact sectors who had low bycatch rates and high target catch rates. Tapping into these skilled operators could help us progress past the low-hanging fruits and address that obstinate "last 10%" of bycatch that has proved extremely hard to eliminate (Savoca et al. 2020).

Compared to other models for similar fisheries that predict catch and bycatch based on environmental and biophysical factors (e.g., sea surface temperature, isothermal layer depth, frontal systems), the effect of individual vessels was a stronger predictor of bycatch and explained a larger portion of the deviance in bycatch rates for most species (Bromhead et al. 2012; Scales et al. 2017, 2018). This underscores the importance of individuals within the system and suggests that bycatch is not a random event across a fishery, which has significant implications for how fisheries are managed. It is also possible that the skipper effect extends to other deleterious activities besides threatened species bycatch. There could be small groups of "regular offenders" across a range of behaviours, such as gear abandonment, accurate logbook reporting, or mistreatment of crew, and the magnitude of the operator effect may vary for different behaviours (Putt and Anderson 2007; Sampson 2011). This would provide important insight and guidance for how to approach compliance more broadly.

The larger objective of identifying behaviour patterns in fishers is of course to change those behaviours for the better. Behaviour change is a large and accelerating field of research and not surprisingly (since we are talking about human beings), the consensus is that behaviour is complex and there is no panacea for catalysing change (Sutinen and Kuperan 1999; Keane et al. 2008; Kurland et al. 2017). It is well understood that fisher behaviour is influenced by many factors beyond economic incentives; thus, management actions that target specific behaviours and drivers can be more effective than traditional management measures, if they are informed by an understanding of the context and implemented in an appropriate way (Hatcher et al. 2000; Österblom et al. 2011; Petrossian 2014; Thomas et al. 2016; Mackay et al. 2018). It is clear that lasting behaviour change requires a combination of approaches that are appropriate to each context (Arias 2015; Reddy et al. 2017). My findings suggest that behaviour change interventions should target performance-based groups of individuals within a fishery (e.g., the target and bycatch "high performers"). Behavioural studies show that social norms and ties among networks can influence fisher behaviour in important and predictable ways, and better understanding of these networks can help managers identify pathways for change (Jentoft 2004; Grafton 2005; Bodin and Crona 2009;

Arlinghaus et al. 2013). For example, fishers in the Hawaiian longline fishery segregated into information-sharing groups that followed ethnic lines, and these groups correlated with different shark bycatch mitigation patterns (Barnes et al. 2016). In this case, encouraging information sharing across ethnic groups could help spread positive bycatch mitigation behaviours.

Another potentially important social structure that could drive patterns in fisher behaviour is the nature of the fishing company, as industrial fishing vessels are typically part of a conglomerate (which could have several layers of ownership) (Carmine et al. 2020). Since corporations are influential social networks with strong and distinct cultures, and most industrial fishing vessels belong to large transnational companies (Österblom 2014; Österblom et al. 2016), it would be important to explore patterns of threatened species bycatch (and other environmental behaviours) across boat owners and seafood companies. Like finding key influencers within social networks, targeting owners or key members of companies could be an effective strategy to accelerate behaviour change in fisheries.

8.3 Research limitations and future research priorities

8.3.1 Data limitations

I used seven global databases of species statuses and distributions and fishing catch, trade and effort, five large observer datasets from the Australian Commonwealth fisheries, and several publicly available supplementary datasets. I am conscious about not complaining about the problems and inconsistencies inherent in all of these data sources because I was not the person who painfully compiled national fisheries reports to piece together global catch reconstructions, dug through handwritten species occurrence records to tune distribution models, or clung to the winch of a tuna longliner identifying shearwaters in a Southern Ocean storm. All of these data sources have limitations and there is always a trade-off between accuracy and resolution and the breadth and scope of the information. Therefore, I will focus on the most important limitations of my results and conclusions and highlight key areas for further investigation.

I use global databases of species range maps and models of fishing effort, which are derived from the above-mentioned data sources, to predict how species are distributed across political boundaries (**Chapter 2**) and to what extent they will encounter fishing gear (**Chapters 5 and 6**). The IUCN range maps are drawn by experts and AquaMaps are generated from environmental preference models. Both sources treat all areas in a species' range equally, without considering different life stages, key habitat areas, migrations, or seasonal patterns in density and distribution. Likewise, the model of fishing effort is highly aggregated across space and time. In these chapters I do not attempt to predict or account for climate-driven shifts in species ranges or climate change impacts on fisheries, which are both large and active fields of research (e.g., Pinsky et al. 2013; García Molinos et al. 2015; Hobday et al. 2015; Free et al. 2019; Brito-Morales et al. 2020). These analyses are based on traditional approaches that map species and threats to provide an intelligible summary of complex and dynamic systems, but the spatial and temporal aggregation limits what these maps can represent. New mapping approaches are being developed to quantify and visualize more complex or dynamic processes and relationships, such as ecosystem services and benefit flows across human and natural systems, in order to guide more informed management decisions (Drakou et al. 2017).

These static and relatively low-resolution results are intended to identify large scale patterns that merit much closer inspection. For example, small-island nations with high transboundary biodiversity will need research and capacity support from wealthier nations (**Chapter 2**), and area-specific management measures could protect multiple taxa from multiple tuna fishing gears in the Indian Ocean (**Chapter 6**). Effective management actions will require more specific information, such as how spatial overlap of fishing and species varies across seasons and years and what types of actions would benefit taxa that are overlooked in existing biodiversity protections. Although it is somewhat unsatisfying to make maps that are too broad or unsophisticated to inform specific conservation actions, knowledge of both regional and local dynamics is important to catalyse management across multiple scales (Drakou et al. 2017; Friedman et al. 2018). Maps tell a story, which is often the first step towards policy change.

A second major source of uncertainty in my conclusions is the limitations of available fisheries catch, effort, and trade data (**Chapters 3, 5, 6, and 7**). Fisheries data are notoriously unreliable because of the difficulty and expense of making direct observations (e.g., onboard observer programs) and because of the general reluctance among fishers to share information about their fishing activities (Mangi et al. 2015). Better catch documentation is a top priority for management agencies globally and will also help improve our knowledge of where species occur and how their ranges are shifting, which is crucial information for conservation assessments (D'Eon-Eggertson et al. 2015; Kennelly 2020). New tools for monitoring fishing are becoming more technologically and economically feasible (e.g., Mangi et al. 2015; Venturelli et al. 2017; Toonen and Bush 2018; Probst 2019). An interesting benefit of investing in new monitoring and surveillance systems is that they can improve the quality and value of traditional data sources. For example, the groundfish hook-and-line fishery in British Columbia, Canada, implemented electronic monitoring as part of a suite of technical measures. Reviewing the footage is labour intensive and only ten percent of it is audited but there are significant penalties for any discrepancies between the videos and the captain's logbook (Stanley et al. 2015). The accuracy of the logbooks has improved substantially and they now provide much better information than before, including information about other aspects of harvesting that are reportedly helpful for the fishers to manage their operations (Stanley et al. 2011).

However, the utility of these new data sources will be diminished if we do not also build mechanisms to access and share data, both within and across institutions (Sequeira et al. 2019; Maureaud et al. 2020). It is unfathomable how much valuable biodiversity data currently sits unutilised, buried in handwritten reports, data sheets, and hard drives in offices and storage cupboards around the world. New data from satellite radar, electronic monitoring systems, or vessel monitoring systems will suffer the same fate if these new tools are not complemented by capacity building efforts to make use of the data. For example, in many countries the national vessel monitoring data is collected and managed by national maritime security units, which tend to be staffed by people with military-type training, and there may be few data scientists or statisticians who can analyse the data correctly and use it to its full potential.

8.3.2 Improving value of information from limited data

Along with securing better sources of data and learning how to use them, a second important area of investigation is how to make better use of existing data. In **Chapters 5 and 6**, I proposed an improvement to a widely used ERA method. Although the probabilistic approach is a mathematical improvement over ordinal scores (e.g., binning the risk of entanglement in fishing gear as low, medium, high), there remain several important flaws in my approach. One challenge is that it is difficult to express and summarise non-point estimates, where there is a range (or interval) of possible outcomes. In the ERA example, there is a point estimate of the likelihood a species is captured in any one grid cell. For species that are targeted by that gear or are definitively not-targeted and will escape unharmed, the outcome is also a point-estimate (0% chance of mortality versus 100% chance of mortality). But in between these two outcomes is a range of possibilities (e.g., the species may escape unharmed or it may suffer serious injuries) and in these cases, the estimated mortality is an interval instead of a point estimate. The interval can be shown on a graph (e.g., Figure 5.3, Figure 6.2, Figure S4.2.4), but the cumulative risk across the species' range could be better communicated using interval statistics methods (Ferson et al. 2007; Zaman et al. 2011).

Related to the problem of quantifying uncertain outcomes is a broader issue of inherent subjectivity in risk assessment approaches. Expert elicitation or judgment is a useful solution to missing or poor data, but the quality of the information derived from experts can vary widely depending on how the information is presented and how the elicitation is conducted (Hemming et al. 2018, 2020). Assessments of many species (e.g., over 400 species in Chapter 6) are challenging because expert fatigue will affect the quality of species-level rankings or judgments (Hemming et al. 2018). To avoid this problem, I first grouped the species by traits that affect their likelihood of entanglement and mortality in fishing gear. However, it is difficult to group species by traits, especially poorly known species or species that use a wide variety of habitats, because there is no standardised system for classifying marine habitats or the species that inhabit them. Terms such as oceanic, pelagic, coastal, or inshore are not clearly defined, so classification schemes must be tailored to each application (Spalding et al. 2007; Costello 2009).

The parameters in the calculation of risk must also be tailored to each application. Some examples will lend themselves to a more systematic and literal interpretation of each dimension; for example, metrics based largely on mesh size are sensible for comparing the likelihood of entanglement of fish in nets, because different mesh sizes are used to target different species (Cotter and Lart 2011; Zhou et al. 2016). In contrast, all species considered in **Chapter 6** are larger than the mesh size of a typical Indian Ocean driftnet, making the gear selectivity ranking less straightforward. Although the parameters are meant to represent independent components of the risk of capture, it is difficult to maintain these strict distinctions when ranking species. For example, the gear selectivity parameter is the risk of entanglement assuming the animal encounters the gear, and the probability of encountering the gear should be expressed only in the encounterability parameter. However, the encounterability parameter is based solely on minimum and maximum depths. This results in unrealistically high catchability estimates for species that are the right size and shape to be entangled but are extremely unlikely to ever encounter that gear (e.g., a benthic skate and a drift gillnet). One possible solution is adding more nuance to the parameters, for instance, incorporating information about how species are distributed in the water column into the vertical encounterability calculation. Likewise, the lethality intervals could be adjusted to account for a wider range of possible outcomes, such as species that are likely to be caught but released as opposed to dead when landed. Ultimately, there will always be inconsistencies and information biases in these risk assessment methods but they will remain important tools for prioritising conservation and management, especially for data-poor fisheries (Gallagher *et al.* 2012; Baillargeon *et al.* 2020; Good *et al.* 2020). Further sensitivity analyses of different rankings and species groupings will help improve this method and provide a better sense of how these uncertainties are propagated through the estimation of risk.

A fundamental challenge for fisheries management is the difficulty of predicting events that rarely occur or are rarely recorded. I encountered these black swan events in **Chapters 5 and 6** in the context of ERAs and in **Chapter 7** in the context of identifying patterns in fishing impacts more broadly. Even the relatively high-quality Commonwealth observer datasets did not have sufficient records for some important bycatch species groups (e.g., sygnathids, sea turtles, sawfish). I used GAMs fit with Tweedie distributions to handle the very over-dispersed data, but statistical methods for zero-inflated count data is a large area of research unto itself and the methods are continuously advancing. There are several variations of the GAM approach I used that would be worth exploring, as well as alternate approaches based on Bayesian frameworks (Zhou et al. 2019; Parsa et al. 2020). Of course, there is a limit to how much information can be massaged from observations of very rare events.

At some point, more data are required to reduce the uncertainty and allow managers to make more informed decisions. Although I do not attempt to evaluate specific management actions, these findings can inform future research from a value of information perspective, where collecting information is valued for its potential to improve management compared to other uses of those resources (Hansen and Jones 2008). A guiding principle that has emerged from quantitative studies of value of information is that the most valuable information is related to the component of the system that you plan to manage (Davis et al. 2019). In the case of the Australian Commonwealth fisheries, this suggests that researching fishers and their fishing behaviours could theoretically be more useful for managing threatened species bycatch than researching the bycatch species, especially for rare bycatch species that would require greater data collection effort. In the Indian Ocean context, where data are scarce across all components of the system and there are few active management measures, knowledge of the fishing effort and fleet dynamics may also be a higher priority than biological information. Although, in this case there could be a valid argument for devoting all resources to implementing bycatch mitigation measures even if many uncertainties remain (Hansen and Jones 2008).

8.3.3 Defining objectives for biodiversity conservation and sustainable fisheries

These chapters are based on the general assumption that catching less threatened species and less bycatch is better for marine biodiversity, thereby leading to more sustainable fisheries. The idea of reducing catch of threatened species is somewhat contentious in fisheries because there is incongruity between conservationist perspectives on threatened species (often using the Red List criteria for extinction risk) and a fisheries management definition of threatened species (based on principles of maximum sustainable yield) (Salomon et al. 2011; Davies and Baum 2012). There is less contention around the assumption that less bycatch of threatened non-target (or usually non-target) species is a positive outcome for biodiversity, and therefore for fisheries sustainability (Hilborn et al. 2015). Still, I do not attempt to define threatened species or sustainable fisheries or unpack the many complexities of these concepts. I consciously only address one aspect of sustainable fishing: overfishing of species that are protected or listed as threatened with extinction. Thus, this work contributes to one piece of a much larger conversation about seafood sustainability, which includes carbon footprints, socioeconomic sustainability, and ecosystem based management (Pikitch et al. 2004; Hilborn et al. 2015).

8.4 Concluding remarks

There is not always consensus among scientists and managers about what constitutes overfishing and what defines a threatened species. However, there is no debate that overfishing—whether targeted or incidental—is a serious socioeconomic and environmental problem for fisheries globally, and a primary threat to marine biodiversity and to humanity (Costello *et al.* 2012; Burgess *et al.* 2018). It is also clear that we need a variety of approaches at a variety of scales to address overfishing. The overall aim of this thesis was to identify gaps in our understanding and management of fishing impacts on biodiversity, with a lens towards finding different types of solutions to overfishing. These solutions can be broadly categorized as technical, regulatory, and social approaches (Hall and Mainprize 2005). These six chapters touch on all three types of solutions to some extent, although regulatory and social solutions were a much larger focus than technical solutions (e.g., specific modifications to fishing gears, technologies to monitor fisheries catch and effort). **Chapters 2-6** explored key gaps in fisheries regulations and biodiversity management at multiple scales, and **Chapter 7** investigated how leveraging the social components of fisheries (individual behaviours) could present an opportunity for more effective regulation. Actual implementation of these solutions will of course require much more than data and theory; it will require complementary actions from scientists, governments, industries, and civil society. The challenge is that the problems facing marine biodiversity are wicked and immense. Fortunately, there are many opportunities for both small and large-scale actions to effect change. We certainly have not exhausted our creativity and capacity to innovate better solutions that reduce the harm fishing causes to biodiversity.

9 References

Abrantes, K., Barnett, A., Kyne, P.M., et al. (2020) Can sawfish bycatch within the Northern Prawn Fishery be mitigated using an electric field? Fisheries Research and Development Corporation Project No 2016-058.

Adams, K.R., Fetterplace, L.C., Davis, A.R., Taylor, M.D. and Knott, N.A. (2018) Sharks, rays and abortion: The prevalence of capture-induced parturition in elasmobranchs. *Biological Conservation* **217**, 11–27.

AFMA (2018) AFMA's dolphin strategy starts to bite. Available at: https://www.afma.gov.au/news-media/news/afmas-dolphin-strategy-starts-bite [Accessed November 11, 2020].

AFMA (2019a) Gillnet Dolphin Mitigation Strategy. Canberra, Australia.

AFMA (2020) Methods and Gear. Available at: https://www.afma.gov.au/fisheries-management/methods-and-gear/longlining [Accessed December 3, 2020].

AFMA (2019b) Southern and Eastern Scalefish and Shark Fishery: Management Arrangements Booklet 2019. Canberra, Australia.

Agardy, T. (2005) Global marine conservation policy versus site-level implementation: the mismatch of scale and its implications. *Marine Ecology Progress Series* **300**, 242–248.

Agardy, T., Bridgewater, P., Crosby, M.P., et al. (2003) Dangerous targets? Unresolved issues and ideological clashes around marine protected areas. *Aquatic Conservation: Marine and Freshwater Ecosystems* **13**, 353–367.

Allison, E.H., Perry, A.L., Badjeck, M.C., et al. (2009) Vulnerability of national economies to the impacts of climate change on fisheries. *Fish and Fisheries* **10**, 173–196.

Amande, M.J., Chassot, E., Chavance, P., Murua, H., Molina, A.D. De and Bez, N. (2012) Precision in bycatch estimates: the case of tuna purse-seine fisheries in the Indian Ocean Monin. *ICES Journal of Marine Science* **69**, 1501–1510.

Anderson, C.M., Krigbaum, M.J., Arostegui, M.C., et al. (2019) How commercial fishing effort is managed. *Fish and Fisheries* **20**, 268–285.

Anderson, R., Herrera, M., Ilangakoon, A., Koya, K., Moazzam, M., Mustika, P. and Sutaria, D. (2020) Cetacean bycatch in Indian Ocean tuna gillnet fisheries. *Endangered Species Research* **41**, 39–53.

Anderson, S.C., Flemming, J.M., Watson, R. and Lotze, H.K. (2011a) Rapid global expansion of invertebrate fisheries: Trends, drivers, and ecosystem effects. *PLoS ONE* **6**, 1–9.

Anderson, S.C., Flemming, J.M., Watson, R. and Lotze, H.K. (2011b) Serial exploitation of global sea cucumber fisheries. *Fish and Fisheries* **12**, 317–339.

Anticamara, J.A., Watson, R., Gelchu, A. and Pauly, D. (2011) Global fishing effort (1950-2010): Trends, gaps, and implications. *Fisheries Research* **107**, 131–136.

Aranda, M. (2017) Description of tuna gillnet capacity and bycatch in the IOTC Convention Area.

Ardill, D., Itano, D. and Gillett, R. (2013) A review of bycatch and discard issues in Indian Ocean tuna fisheries.

Ardron, J.A., Rayfuse, R., Gjerde, K. and Warner, R. (2014) The sustainable use and conservation of biodiversity in ABNJ: What can be achieved using existing international agreements? *Marine Policy* **49**, 98–108.

Arias, A. (2015) Understanding and managing compliance in the nature conservation context. *Journal of Environmental Management* **153**, 134–143.

Arlinghaus, R., Abbott, J.K., Fenichel, E.P., et al. (2019) Opinion: Governing the recreational dimension of global fisheries. *Proceedings of the National Academy of Sciences of the United States of America* **116**, 5209–5213.

Arlinghaus, R., Cooke, S.J. and Potts, W. (2013) Towards resilient recreational fisheries on a global scale through improved understanding of fish and fisher behaviour. *Fisheries Management and Ecology* **20**, 91–98.

Aron, W., Burke, W. and Freeman, M.M.R. (2000) The whaling issue. *Marine Policy* **24**, 179–191.

Arrizabalaga, H., De Bruyn, P., Diaz, G.A., et al. (2011) Productivity and susceptibility analysis for species caught in Atlantic tuna fisheries. *Aquatic Living Resources* **24**, 1–12.

Bailey, M., Packer, H., Schiller, L., Tlusty, M. and Swartz, W. (2018) The role of corporate social responsibility in creating a Seussian world of seafood sustainability. *Fish and Fisheries* **19**, 782–790.

Baillargeon, G.A., Tlusty, M.F., Dougherty, E.T. and Rhyne, A.L. (2020) Improving the productivity-susceptibility analysis to assess data-limited fisheries. *Marine Ecology Progress Series* **644**, 143–156.

Barkin, J.S. and DeSombre, E.R. (2013) Do we need a global fisheries management organization? *Journal of Environmental Studies and Sciences* **3**, 232–242.

Barnes, M.L., Lynham, J., Kalberg, K. and Leung, P. (2016) Social networks and environmental outcomes. *Proceedings of the National Academy of Sciences* **113**, 6466–6471.

Barton, K. (2015) Package 'mumin.' 18.

Basurto, X. and Nenadovic, M. (2012) A systematic approach to studying fisheries governance. *Global Policy* **3**, 222–230.

Batavia, C., Nelson, M.P., Darimont, C.T., Paquet, P.C., Ripple, W.J. and Wallach, A.D. (2019) The elephant (head) in the room: A critical look at trophy hunting. *Conservation Letters* **12**, 1–6.

Beddington, J.R., Agnew, D.J. and Clark, C.W. (2007) Current problems in the management of marine fisheries. *Science* **316**, 1713–6.

Bejder, M., Johnston, D.W., Smith, J., Friedlaender, A. and Bejder, L. (2016) Embracing conservation success of recovering humpback whale populations: Evaluating the case for downlisting their conservation status in Australia. *Marine Policy* **66**, 137–141.

Belhabib, D., Sumaila, U.R. and Le Billon, P. (2019) The fisheries of Africa: Exploitation, policy, and maritime security trends. *Marine Policy* **101**, 80–92.

Bellmann, C., Tipping, A. and Sumaila, U.R. (2016) Global trade in fish and fishery products: An overview. *Marine Policy* **69**, 181–188.

Benaglia, T., Chauveau, D., Hunter, D.R. and Young, D.S. (2009) Mixtools: An R package for analyzing finite mixture models. *Journal of Statistical Software* **32**, 1–29.

Bennett, N.J. (2019) In Political Seas: Engaging with Political Ecology in the Ocean and Coastal Environment. *Coastal Management* **47**, 67–87.

Benoît, H.P. and Allard, J. (2009) Can the data from at-sea observer surveys be used to make general inferences about catch composition and discards? *Canadian Journal of Fisheries and Aquatic Sciences* **66**, 2025–2039.

Bergh, M., Gayland, J., Martens, K., Knuckey, I. and Koopman, M. (2009) Revised Sampling Regime for the Southern and Eastern Scalefish and Shark Fishery – Final Report, AFMA Project F2008/0627.

Bhat, M.G. and Bhatta, R. (2006) Mechanization and technical interactions in multi-species Indian fisheries: Implications for economic and biological sustainability. *Marine Policy* **30**, 237–248.

Bianchi, C.N. and Morri, C. (2000) Marine biodiversity of the Mediterranean Sea: Situation, problems and prospects for future research. *Marine Pollution Bulletin* **40**, 367–376.

Blanchard, C. (2017) Fragmentation in high seas fisheries : Preliminary reflections on a global oceans governance approach. *Marine Policy* **84**, 327–332.

Bodin, Ö. and Crona, B.I. (2009) The role of social networks in natural resource governance: What relational patterns make a difference? *Global Environmental Change* **19**, 366–374.

Boerder, K., Schiller, L. and Worm, B. (2019) Not all who wander are lost: Improving spatial protection for large pelagic fishes. *Marine Policy* **105**, 80–90.

Boulais, O. (2019) Exploring Provenance of Tuna using Distributed Ledgers. Available at: https://viral.media.mit.edu/pub/tunaprovenance [Accessed November 18, 2019].

Bourjea, J., Nel, R., Jiddawi, N., Koonjul, M. and Bianchi, G. (2008) Sea Turtle Bycatch in the West Indian Ocean: Review, Recommendations and Research Priorities. *Western Indian Ocean J. Mar. Sci* **7**, 137–150.

Breen, P., Brown, S., Reid, D. and Rogan, E. (2017) Where is the risk? Integrating a spatial distribution model and a risk assessment to identify areas of cetacean interaction with fisheries in the northeast Atlantic. *Ocean and Coastal Management* **136**, 148–155.

Brehme, C.S., Hathaway, S.A. and Fisher, R.N. (2018) An objective road risk assessment method for multiple species: ranking 166 reptiles and amphibians in California. *Landscape Ecology* **33**, 911–935.

Brewer, D., Heales, D., Milton, D., Dell, Q., Fry, G., Venables, B. and Jones, P. (2006) The impact of turtle excluder devices and bycatch reduction devices on diverse tropical marine communities in Australia's northern prawn trawl fishery. *Fisheries Research* **81**, 176–188.

Brito-Morales, I., Schoeman, D.S., Molinos, J.G., et al. (2020) Climate velocity reveals increasing exposure of deep-ocean biodiversity to future warming. *Nature Climate Change* **10**, 576–581.

Bromhead, D., Clarke, S., Hoyle, S., Muller, B., Sharples, P. and Harley, S. (2012) Identification of factors influencing shark catch and mortality in the Marshall Islands tuna longline fishery and management implications. *Journal of Fish Biology* **80**, 1870–1894.

Brown, S.L., Reid, D. and Rogan, E. (2013) A risk-based approach to rapidly screen vulnerability of cetaceans to impacts from fisheries bycatch. *Biological Conservation* **168**, 78–87.

Brown, S.L., Reid, D. and Rogan, E. (2015) Spatial and temporal assessment of potential risk to cetaceans from static fishing gears. *Marine Policy* **51**, 267–280.

Brownell, R.L., Reeves, R.R., Read, A.J., et al. (2019) Bycatch ingillnet fisheries threatens critically endangeredsmall cetaceansand other aquatic megafauna. *Endangered Species Research* **40**, 285–296.

Burden, M. and Fujita, R. (2019) Better fisheries management can help reduce conflict, improve food security, and increase economic productivity in the face of climate change. *Marine Policy* **108**, 103610.

Burgess, M.G., Mcdermott, G.R., Owashi, B., et al. (2018) Protecting marine mammals, turtles, and birds by rebuilding global fisheries. *Science* **1258**, 1255–1258.

Cabral, R.B., Mayorga, J., Clemence, M., et al. (2018) Rapid and lasting gains from solving illegal fishing. *Nature Ecology & Evolution* **2**, 650–658.

Campbell, L.M. and Gray, N.J. (2019) Area expansion versus effective and equitable management in international marine protected areas goals and targets. *Marine Policy* **100**, 192–199.

Campbell, R.A. and Young, J.W. (2012) Monitoring the behaviour of longline gears and the depth and time of fish capture in the Australian Eastern Tuna and Billfish Fishery. *Fisheries Research* **119–120**, 48–65.

Carmine, G., Mayorga, J., Miller, N.A., et al. (2020) Who is the high seas fishing industry? *One Earth* **3**, 730–738.

Carr, M.H., Neigel, J.E., Estes, J.A., Andelman, S., Warner, R.R. and Largier, J.L. (2003) Comparing marine and terrestrial ecosystems: Implications for the design of coastal marine reserves. *Ecological Applications* **13**, 90–107.

Carr, M.H., Robinson, S.P., Wahle, C., et al. (2017) The central importance of ecological spatial connectivity to effective coastal marine protected areas and to meeting the challenges of climate change in the marine environment. *Aquatic Conservation: Marine and Freshwater Ecosystems* **27**, 6–29.

Carretta, J. V., Barlow, J. and Enriquez, L. (2008) Acoustic pingers eliminate beaked whale bycatch in a gill net fishery. *Marine Mammal Science* **24**, 956–961.

Carruthers, E.H., Schneider, D.C. and Neilson, J.D. (2009) Estimating the odds of survival and identifying mitigation opportunities for common bycatch in pelagic longline fisheries. *Biological Conservation* **142**, 2620–2630.

CBD (2011) Report of the Tenth Meeting of the Conference of the Parties to the Convention on Biological Diversity. Montreal, Canada.

Chan, H.L. and Pan, M. (2016) Spillover Effects of Environmental Regulation for Sea Turtle Protection in the Hawaii Longline Swordfish Fishery. *Marine Resource Economics* **31**, 259–279.

Chen, I.C., Lee, P.F. and Tzeng, W.N. (2005) Distribution of albacore (Thunnus alalunga) in the

Indian Ocean and its relation to environmental factors. *Fisheries Oceanography* **14**, 71–80.

Chen, W., Song, L., Li, J., Xu, W. and Li, D. (2012) Optimum soak time of tuna longline gear in the Indian Ocean. *Fourteenth Working Party on Tropical Tunas, Mauritius, 24–29 October 2012* **1**, 24–29.

Chin, A., Kyne, P.M., Walker, T.I. and McAuley, R.B. (2010) An integrated risk assessment for climate change: Analysing the vulnerability of sharks and rays on Australia's Great Barrier Reef. *Global Change Biology* **16**, 1936–1953.

Christian, C., Ainley, D., Bailey, M., et al. (2013) A review of formal objections to Marine Stewardship Council fisheries certifications. *Biological Conservation* **161**, 10–17.

Clarke, S., Sato, M., Small, C., Sullivan, B., Inoue, Y. and Ochi, D. (2014) Bycatch in longline fisheries for tuna and tuna-like species. A global review of status and mitigation measures.

Clavareau, L., Sabarros, P.S., Escalle, L., et al. (2020) Elasmobranch bycatch distributions and mortality: Insights from the European tropical tuna purse-seine fishery. *Global Ecology and Conservation* **24**.

Coleman, F. and Williams, S.L. (2002) Overexploiting marine ecosystem engineers:potential consequences for biodiversity. *Trends in Ecology & Evolution* **17**, 02330–8.

Collette, B.B., Carpenter, K.E., Polidoro, B. a, et al. (2011) High Value and Long Life — Double Jeopardy for Tunas and Billfi shes. *Science* **333**, 291–292.

Collier, K.J., Probert, P.K. and Jeffries, M. (2016) Conservation of aquatic invertebrates: concerns, challenges and conundrums. *Aquatic Conservation: Marine and Freshwater Ecosystems* **26**, 817–837.

Commonwealth of Australia (2018) Threat Abatement Plan for the incidental catch (or bycatch) of seabirds during oceanic longline fishing operations (2018). Canberra.

Cooke, S.J. and Cowx, I.G. (2006) Contrasting recreational and commercial fishing: Searching for common issues to promote unified conservation of fisheries resources and aquatic environments. *Biological Conservation* **128**, 93–108.

Cooke, S.J. and Cowx, I.G. (2004) The role of recreational fishing in global fish crises. *Bioscience* **54**, 857–859.

Cortés, E., Arocha, F., Beerkircher, L., et al. (2010) Ecological risk assessment of pelagic sharks caught in Atlantic pelagic longline fisheries. *Aquatic Living Resources* **23**, 25–34.

Costello, C., Ovando, D., Hilborn, R., Gaines, S.D., Deschenes, O. and Lester, S. (2012) Status and

Solutions for the World's Unassessed Fisheries. *Science* **338**, 517–520.

Costello, M.J. (2015) Biodiversity: The known, unknown, and rates of extinction. *Current Biology* **25**, R368–R371.

Costello, M.J. (2009) Distinguishing marine habitat classification concepts for ecological data management. *Marine Ecology Progress Series*.

Costello, M.J. and Ballantine, B. (2015) Biodiversity conservation should focus on no-take Marine Reserves: 94% of Marine Protected Areas allow fishing. *Trends in Ecology and Evolution* **30**, 507–509.

Costello, M.J., Coll, M., Danovaro, R., Halpin, P., Ojaveer, H. and Miloslavich, P. (2010) A Census of Marine Biodiversity Knowledge, Resources, and Future Challenges. *PLoS ONE* **5**, e12110.

Cotter, J. and Lart, W. (2011) A Guide for Ecological Risk Assessment of the Effects of Commercial Fishing (ERAEF).

Cox, T.M., Lewison, R.L., Žydelis, R., Crowder, L.B., Safina, C. and Read, A.J. (2007) Comparing effectiveness of experimental and implemented bycatch reduction measures: The ideal and the real. *Conservation Biology* **21**, 1155–1164.

Crespo, G.O., Dunn, D.C., Gianni, M., Gjerde, K., Wright, G. and Halpin, P.N. (2019) High-seas fish biodiversity is slipping through the governance net. *Nature Ecology & Evolution* **3**, 1273–1276.

Crowder, L.B. and Murawski, S.A. (2017) Fisheries Bycatch : Implications for Management. *Fisheries* **2415**.

Cullis-Suzuki, S. and Pauly, D. (2010) Failing the high seas: A global evaluation of regional fisheries management organizations. *Marine Policy* **34**, 1036–1042.

Cutlip, K. (2016) Our Analyst Nets Another IUU Bust. Available at: http://globalfishingwatch.org/impacts/our-analyst-nets-another-iuu-bust/.

D'Eon-Eggertson, F., Dulvy, N.K. and Peterman, R.M. (2015) Reliable Identification of Declining Populations in an Uncertain World. *Conservation Letters* **8**, 86–96.

Dankel, D.J., Skagen, D.W. and Ulltang, Ø. (2008) Fisheries management in practice: Review of 13 commercially important fish stocks. *Reviews in Fish Biology and Fisheries* **18**, 201–233.

Davies, R.W.D., Cripps, S.J., Nickson, A. and Porter, G. (2009) Defining and estimating global marine fisheries bycatch. *Marine Policy* **33**, 661–672.

Davies, T.D. and Baum, J.K. (2012) Extinction Risk and Overfishing: Reconciling Conservation and Fisheries Perspectives on the Status of Marine Fishes. *Scientific Reports* **2**, 561.

Davies, T.K., Mees, C.C. and Milner-Gulland, E.J. (2014) The past, present and future use of drifting fish aggregating devices (FADs) in the Indian Ocean. *Marine Policy* **45**, 163–170.

Davis, K.J., Chadès, I., Rhodes, J.R. and Bode, M. (2019) General rules for environmental management to prioritise social ecological systems research based on a value of information approach. *Journal of Applied Ecology* **56**, 2079–2090.

Dawson, J., Oppel, S., Cuthbert, R.J., et al. (2015) Prioritizing islands for the eradication of invasive vertebrates in the United Kingdom overseas territories. *Conservation Biology* **29**, 143–153.

Dent, F. and Clarke, S. (2015) State of the global market for shark products. *FAO Fishereis and Aquaculture Technical paper No. 590.*, 187.

Díaz, S., Settele, J., E.S., B., et al. (2019) Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Bonn, Germany.

Diz, D., Johnson, D., Riddell, M., et al. (2017) Mainstreaming marine biodiversity into the SDGs: The role of other effective area-based conservation measures (SDG 14.5). *Marine Policy*, 1–11.

Donaldson, M.R., Burnett, N.J., Braun, D.C., Suski, C.D., Hinch, S.G., Cooke, S.J. and Kerr, J.T. (2016) Taxonomic bias and international biodiversity conservation research. *Facets* **1**, 105–113.

Doney, S.C., Ruckelshaus, M., Duffy, J.E., et al. (2012) Climate Change Impacts on Marine Ecosystems. *Annu. Rev. Mar. Sci* **4**, 11–37.

Doumbouya, A., Camara, O.T., Mamie, J., et al. (2017) Assessing the Effectiveness of Monitoring Control and Surveillance of Illegal Fishing: The Case of West Africa. *Frontiers in Marine Science* **4**, 50.

Drakou, E.G., Pendleton, L., Effron, M., Ingram, J.C. and Teneva, L. (2017) When ecosystems and their services are not co-located: Oceans and coasts. *ICES Journal of Marine Science* **74**, 1531–1539.

Duarte, C.M., Agusti, S., Barbier, E., et al. (2020) Rebuilding marine life. *Nature* **580**, 39–51.

Duffy, L.M. and Griffiths, S.P. (2019) Assessing attribute redundancy in the application of productivity-susceptibility analysis to data-limited fisheries. *Aquatic Living Resources* **32**.

Duffy, L.M., Lennert-Cody, C.E., Olson, R.J., Minte-Vera, C. V. and Griffiths, S.P. (2019) Assessing vulnerability of bycatch species in the tuna purse-seine fisheries of the eastern Pacific Ocean. *Fisheries Research* **219**.

Dulvy, N.K., Fowler, S.L., Musick, J.A., et al. (2014) Extinction risk and conservation of the world's sharks and rays. *eLife* **3**, e00590.

Dulvy, N.K., Jennings, S., Goodwin, N.B., Grant, A. and Reynolds, J.D. (2005) Comparison of threat and exploitation status in North- East Atlantic marine populations. *Journal of Applied Ecology* **42**, 883–891.

Dulvy, N. K., Sadovy, Y. and Reynolds, J. D. (2003). Extinction vulnerability in marine populations. *Fish and Fisheries*. **4,** 25-64.

Dulvy, N.K., Simpfendorfer, C.A., Davidson, L.N.K., Fordham, S. V., Bräutigam, A., Sant, G. and Welch, D.J. (2017) Challenges and Priorities in Shark and Ray Conservation. *Current Biology* **27**, R565–R572.

Dunn, D.C., Harrison, A.L., Curtice, C., et al. (2019) The importance of migratory connectivity for global ocean policy. *Proceedings of the Royal Society B: Biological Sciences* **286**.

Dunn, D.C., Maxwell, S.M., Boustany, A.M. and Halpin, P.N. (2016) Dynamic ocean management increases the efficiency and efficacy of fisheries management. *Proceedings of the National Academy of Sciences of the United States of America* **113**, 668–73.

Eayrs, S., Cadrin, S.X. and Glass, C.W. (2015) Managing change in fisheries: a missing key to fishery-dependent data collection? *ICES Journal of Marine Science* **72**, 1152–1158.

Eddy, C., Brill, R. and Bernal, D. (2016) Rates of at-vessel mortality and post-release survival of pelagic sharks captured with tuna purse seines around drifting fish aggregating devices (FADs) in the equatorial eastern Pacific Ocean. *Fisheries Research* **174**, 109–117.

Elfes, C.T., Cristiane, T., Livingstone, S.R. and Suzanne, R. (2013) Fascinating and Forgotten : the Conservation Status of Marine Elapid Snakes. *Herpetological Conservation and Biology* **8**, 37–52.

Elwen, S.H., Findlay, K.P., Kiszka, J. and Weir, C.R. (2011) Cetacean research in the southern African subregion: a review of previous studies and current knowledge. *African Journal of Marine Science* **33**, 469–493.

Emery, T., Marton, N., Woodhams, J. and Curtotti, R. (2020) Southern and Eastern Scalefish and Shark Fishery.

Emery, T.J., Noriega, R., Williams, A.J. and Larcombe, J. (2019a) Changes in logbook reporting by commercial fishers following the implementation of electronic monitoring in Australian Commonwealth fisheries. *Marine Policy* **104**, 135–145.

Emery, T.J., Noriega, R., Williams, A.J. and Larcombe, J. (2019b) Measuring congruence between electronic monitoring and logbook data in Australian Commonwealth longline and gillnet fisheries. *Ocean and Coastal Management* **168**, 307–321.

Erisman, B., Heyman, W., Kobara, S., Ezer, T., Pittman, S., Aburto-Oropeza, O. and Nemeth, R.S. (2017) Fish spawning aggregations: where well-placed management actions can yield big benefits for fisheries and conservation. *Fish and Fisheries* **18**, 128–144.

Escalle, L., Capietto, A., Chavance, P., et al. (2015) Cetaceans and tuna purse seine fisheries in the Atlantic and Indian Oceans: Interactions but few mortalities. *Marine Ecology Progress Series* **522**, 255–268.

Estes, J.A., Heithaus, M., Mccauley, D.J., Rasher, D.B. and Worm, B. (2016) Megafaunal Impacts on Structure and Function of Ocean Ecosystems.

Failler, P., Touron-Gardic, G. and Traore, M.S. (2019) Is Aichi Target 11 Progress Correctly Measured for Developing Countries? *Trends in Ecology and Evolution* **34**, 875–879.

FAO (2016) Common Oceans: Global Sustainable Fisheries Management and Biodiversity Conservation in Areas Beyond National Jurisdiction.

FAO (2014) The state of world fisheries and aquaculture 2014. Rome, Italy.

FAO (2018) The State of World Fisheries and Aquaculture 2018 - Meeting the sustainable development goals. Rome.

FAO (2020) The State of World Fisheries and Aquaculture 2020. Rome.

FAO Fisheries and Aquaculture (2021) ASFIS List of Species for Fishery Statistics Purposes. Available at: http://www.fao.org/fishery/collection/asfis/en [Accessed January 16, 2021].

Fernandes, P.G., Ralph, G.M., Nieto, A., et al. (2017) Coherent assessments of Europe’s marine fishes show regional divergence and megafauna loss. *Nature Ecology & Evolution* **1**, 0170.

Ferson, S., Kreinovich, V., Hajagos, J., Oberkampf, W. and Ginzburg, L. (2007) Experimental Uncertainty Estimation and Statistics for Data Having Interval Uncertainty. Albuquerque, USA.

Fidelman, P. and Ekstrom, J.A. (2012) Mapping seascapes of international environmental arrangements in the Coral Triangle. *Marine Policy* **36**, 993–1004.

Finkbeiner, E.M., Bennett, N.J., Frawley, T.H., et al. (2017) Reconstructing overfishing: Moving beyond Malthus for effective and equitable solutions. *Fish and Fisheries* **18**, 1180–1191.

Fisher, R., Wilson, S.K., Sin, T.M., Lee, A.C. and Langlois, T.J. (2018) A simple function for full-subsets multiple regression in ecology with R. *Ecology and Evolution* **8**, 6104–6113.

Foley, C.M., Lynch, M.A., Thorne, L.H. and Lynch, H.J. (2017) Listing Foreign Species under the Endangered Species Act: A Primer for Conservation Biologists. *BioScience* **67**, 627–637.

Free, C.M., Thorson, J.T., Pinsky, M.L., Oken, K.L., Wiedenmann, J. and Jensen, O.P. (2019) Impacts of historical warming on marine fisheries production. *Science* **363**, 979–983.

Freed, S., Dujon, V., Granek, E.F. and Mouhhidine, J. (2016) Enhancing small-scale fisheries management through community engagement and multi-community partnerships: Comoros case study. *Marine Policy* **63**, 81–91.

Friedman, A. (2019) Beyond "not undermining": Possibilities for global cooperation to improve environmental protection in areas beyond national jurisdiction. *ICES Journal of Marine Science* **76**, 452–456.

Friedman, K., Garcia, S.M. and Rice, J. (2018) Mainstreaming biodiversity in fisheries. *Marine Policy* **95**, 209–220.

Froese, R. (2004) Keep it simple: Three indicators to deal with overfishing. *Fish and Fisheries* **5**, 86–91.

Froese, R. and Pauly, D. (2019) FishBase.

Fry, G.C., Milton, D.A. and Wassenberg, T.J. (2001) The reproductive biology and diet of sea snake bycatch of prawn trawling in Northern Australia: Characteristics important for assessing the impacts on populations. *Pacific Conservation Biology* **7**, 55–73.

Fulton, E.A., Smith, A.D.M., Smith, D.C. and Johnson, P. (2014) An integrated approach is needed for ecosystem based fisheries management: Insights from ecosystem-level management strategy evaluation. *PLoS ONE* **9**.

Fulton, E.A., Smith, A.D.M., Smith, D.C. and Van Putten, I.E. (2011) Human behaviour: The key source of uncertainty in fisheries management. *Fish and Fisheries* **12**, 2–17.

Future of Fish (2015) Making Sense of Wild Seafood Supply Chains.

Gaertner, D., Pagavino, M. and Marcano, J. (1999) Influence of fishers' behaviour on the catchability of surface tuna schools in the venezuelan purse-seiner fishery in the Caribbean Sea. *Canadian Journal of Fisheries and Aquatic Sciences* **56**, 394–406.

Gaines, S.D., White, C., Carr, M.H., Palumbi, S.R. and Levin, S.A. (2010) Designing marine reserve networks for both conservation and fisheries management. *PNAS* **107**, 18286–18293.

Gallagher, A.J., Kyne, P.M. and Hammerschlag, N. (2012) Ecological risk assessment and its application to elasmobranch conservation and management. *Journal of Fish Biology* **80**, 1727–1748.

García Molinos, J., Halpern, B.S.S., Schoeman, D.S.S., et al. (2015) Climate velocity and the future global redistribution of marine biodiversity. *Nature Climate Change* **6**, 4–11.

Garcia, A. and Herrera, M. (2019) Assessing the Contribution of Purse Seine Fisheries to Overall Levels of Bycatch in the Indian Ocean.

Gattuso, J.P., Magnan, A.K., Bopp, L., et al. (2018) Ocean solutions to address climate change and its effects on marine ecosystems. *Frontiers in Marine Science* **5**.

Gaughan, D.J. and Santoro, K. (2020) Status reports of the fisheries and aquatic resources of Western Australia 2018/19: The State of the Fisheries.

Georgeson, L., Rigby, C.L., Emery, T.J., et al. (2020) Ecological risks of demersal fishing on deepwater chondrichthyan populations in the Southern Indian and South Pacific Oceans. *ICES Journal of Marine Science*.

Gillett, R. (2011) Bycatch in small-scale tuna fisheries: A global study.

Gilman, E. (2015) Status of international monitoring and management of abandoned, lost and discarded fishing gear and ghost fishing. *Marine Policy* **60**, 225–239.

Gilman, E., Chaloupka, M., Swimmer, Y. and Piovano, S. (2016) A cross-taxa assessment of pelagic longline by-catch mitigation measures: conflicts and mutual benefits to elasmobranchs. *Fish and Fisheries* **17**, 748–784.

Gilman, E.L. (2011) Bycatch governance and best practice mitigation technology in global tuna fisheries. *Marine Policy* **35**, 590–609.

Gjerde, K.M. (2012) Challenges to protecting the marine environment beyond national jurisdiction. *International Journal of Marine and Coastal Law* **27**, 839–847.

Gladics, A.J., Melvin, E.F., Suryan, R.M., Good, T.P., Jannot, J.E. and Guy, T.J. (2017) Fishery-specific solutions to seabird bycatch in the U.S. West Coast sablefish fishery. *Fisheries Research* **196**, 85–95.

Glaser, S.M., Roberts, P.M. and Hurlburt, K.J. (2019) Foreign Illegal, Unreported, and Unregulated Fishing in Somali Waters Perpetuates Conflict. *Frontiers in Marine Science* **6**.

González-Laxe, F. (2005) The precautionary principle in fisheries management. *Marine Policy* **29**, 495–505.

Good, T.P., June, J.A., Etnier, M.A. and Broadhurst, G. (2010) Derelict fishing nets in Puget Sound and the Northwest Straits: Patterns and threats to marine fauna. *Marine Pollution Bulletin* **60**, 39–50.

Good, T.P., Samhouri, J.F., Feist, B.E., Wilcox, C. and Jahncke, J. (2020) Plastics in the Pacific: Assessing risk from ocean debris for marine birds in the California Current Large Marine Ecosystem. *Biological Conservation* **250**, 108743.

Gourguet, S., Thébaud, O., Jennings, S., et al. (2016) The Cost of Co-viability in the Australian Northern Prawn Fishery. *Environmental Modeling and Assessment* **21**, 371–389.

Grafeld, S., Oleson, K.L.L., Teneva, L. and Kittinger, J.N. (2017) Follow that fish: Uncovering the hidden blue economy in coral reef fisheries. *PLoS ONE* **12**, 1–25.

Grafton, R.Q. (2005) Social capital and fisheries governance. *Ocean and Coastal Management* **48**, 753–766.

Granek, E.F., Madin, E.M.P., Brown, M.A., et al. (2008) Engaging recreational fishers in management and conservation: Global case studies. *Conservation Biology* **22**, 1125–1134.

Gray, C.A. and Kennelly, S.J. (2018) Bycatches of endangered, threatened and protected species in marine fisheries. *Reviews in Fish Biology and Fisheries* **28**, 521–541.

Gregory, M.R. (2009) Environmental implications of plastic debris in marine settings-entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. *Philosophical Transactions of the Royal Society B: Biological Sciences* **364**, 2013–2025.

Grose, S.O., Pendleton, L., Leathers, A., Cornish, A. and Waitai, S. (2020) Climate Change Will Re-draw the Map for Marine Megafauna and the People Who Depend on Them. *Frontiers in Marine Science*.

Gross, M. (2018) Eyes on our planet. *Current Biology* **28**, R89–R92.

Gutierrez, N.L., Defeo, O., Bush, S.R., Butterworth, D.S., Roheim, C.A. and Punt, A.E. (2016) The current situation and prospects of fisheries certification and ecolabelling. *Fisheries Research* **182**, 1–6.

Haas, B., McGee, J., Fleming, A. and Haward, M. (2020) Factors influencing the performance of regional fisheries management organizations. *Marine Policy* **113**, 103787.

Hall, M., Gilman, E., Minami, H., Mituhasi, T. and Carruthers, E. (2017) Mitigating bycatch in tuna fisheries. *Reviews in Fish Biology and Fisheries* **27**, 881–908.

Hall, M.A., Alverson, D.L. and Metuzals, K.I. (2000) By-catch: Problems and solutions. *Marine Pollution Bulletin* **41**, 204–219.

Hall, S.J. and Mainprize, B.M. (2005) Managing by-catch and discards: How much progress are we making and how can we do better? *Fish and Fisheries* **6**, 134–155.

Hamilton, S. and Baker, G.B. (2019) Technical mitigation to reduce marine mammal bycatch and entanglement in commercial fishing gear: lessons learnt and future directions. *Reviews in Fish Biology and Fisheries* **29**, 223–247.

Hanich, Q., Campbell, B., Bailey, M. and Molenaar, E. (2015) Research into fisheries equity and fairness-addressing conservation burden concerns in transboundary fisheries. *Marine Policy* **51**, 302–304.

Hansen, G.J.A. and Jones, M.L. (2008) The Value of Information in Fishery Management. *Fisheries* **33**, 340–348.

Hardesty, B.D., Good, T.P. and Wilcox, C. (2015) Novel methods, new results and science-based solutions to tackle marine debris impacts on wildlife. *Ocean and Coastal Management* **115**, 4–9.

Hare, J.A., Morrison, W.E., Nelson, M.W., et al. (2016) A vulnerability assessment of fish and invertebrates to climate change on the northeast u.s. continental shelf. *PLoS ONE* **11**, 1–30.

Harrison, A.L., Costa, D.P., Winship, A.J., et al. (2018) The political biogeography of migratory marine predators. *Nature Ecology and Evolution* **2**, 1571–1578.

Hatcher, A., Jaffry, S., Thébaud, O., Bennett, E., Thibaud, O., Thebaud, O. and Bennett, E. (2000) Normative and Social Influences Affecting Compliance with Fishery Regulations. *Land Economics* **76**, 448–461.

Hazen, E.L., Scales, K.L., Maxwell, S.M., et al. (2018) A dynamic ocean management tool to reduce bycatch and support sustainable fisheries. *Science Advances* **4**, 1–8.

He, J. (2018) From country-of-origin labelling (COOL) to seafood import monitoring program (SIMP): How far can seafood traceability rules go? *Marine Policy* **96**, 163–174.

Helmond, A.T.M. Van, Catchpole, T., Mortensen, L.O., et al. (2019) Electronic monitoring in fisheries : Lessons from global experiences and future opportunities. *Fish and Fisheries*, 1–28.

Hemming, V., Burgman, M.A., Hanea, A.M., McBride, M.F. and Wintle, B.C. (2018) A practical guide to structured expert elicitation using the IDEA protocol. *Methods in Ecology and*

Evolution **9**, 169–180.

Hemming, V., Hanea, A.M., Walshe, T. and Burgman, M.A. (2020) Weighting and aggregating expert ecological judgments. *Ecological Applications* **30**, 1–17.

Henderson, A.C., McIlwain, J.L., Al-Oufi, H.S. and Al-Sheili, S. (2007) The Sultanate of Oman shark fishery: Species composition, seasonality and diversity. *Fisheries Research* **86**, 159–168.

Hewitt, J.E., Ellis, J.I. and Thrush, S.F. (2016) Multiple stressors, nonlinear effects and the implications of climate change impacts on marine coastal ecosystems. *Global change biology* **22**, 2665–2675.

Hilborn, R. (2007a) Defining success in fisheries and conflicts in objectives. *Marine Policy* **31**, 153–158.

Hilborn, R. (1985) Fleet Dynamics and Individual Variation: Why Some People Catch More Fish than Others. *Canadian Journal of Fisheries and Aquatic Sciences* **42**, 2–13.

Hilborn, R. (2007b) Managing fisheries is managing people: What has been learned? *Fish and Fisheries* **8**, 285–296.

Hilborn, R., Amoroso, R.O., Anderson, C.M., et al. (2020) Effective fisheries management instrumental in improving fish stock status. *Proceedings of the National Academy of Sciences of the United States of America* **117**, 2218–2224.

Hilborn, R., Fulton, E.A., Green, B.S., Hartmann, K., Tracey, S.R. and Watson, R.A. (2015) When is a fishery sustainable? *Canadian Journal of Fisheries and Aquatic Sciencesjournal* **72**, 1433–1441.

Hilborn, R. and Hilborn, U. (2012) *Overfishing : What Everyone Needs to Know®*. Oxford University Press USA - OSO, Oxford, UNITED STATES.

Hilborn, R. and Ovando, D. (2014) Reflections on the success of traditional fisheries management. *ICES Journal of Marine Science* **71**, 1040–1046.

Hilborn, R., Stokes, K., Maguire, J.J., et al. (2004) When can marine reserves improve fisheries management? *Ocean and Coastal Management* **47**, 197–205.

Hobbs, C.A.D., Potts, R.W.A., Bjerregaard Walsh, M., Usher, J. and Griffiths, A.M. (2019) Using DNA Barcoding to Investigate Patterns of Species Utilisation in UK Shark Products Reveals Threatened Species on Sale. *Scientific Reports* **9**, 1–10.

Hobday, A., Fuller, M., Griffiths, S., et al. (2007) Ecological Risk Assessment for Effects of Fishing.

Hobday, A.J., Bell, J.D., Cook, T.R., Gasalla, M.A. and Weng, K.C. (2015) Reconciling conflicts in pelagic fisheries under climate change. *Deep-Sea Research Part II: Topical Studies in Oceanography* **113**, 291–300.

Hobday, A.J., Smith, A.D.M., Stobutzki, I.C., et al. (2011) Ecological risk assessment for the effects of fishing. *Fisheries Research* **108**, 372–384.

Holland, D.S. (2007) Managing environmental impacts of fishing: input controls versus outcome-oriented approaches. *International Journal of Global Environmental Issues* **7**, 255.

Hopkins, G.R. and Brodie, E.D. (2015) Occurrence of Amphibians in Saline Habitats: A Review and Evolutionary Perspective. *Herpetological Monographs* **29**, 1–27.

Hordyk, A.R. and Carruthers, T.R. (2018) A quantitative evaluation of a qualitative risk assessment framework: Examining the assumptions and predictions of the Productivity Susceptibility Analysis (PSA). *PLoS ONE* **13**, e0198298.

Hornborg, S., Svensson, M., Nilsson, P. and Ziegler, F. (2013) By-catch impacts in fisheries: Utilizing the iucn red list categories for enhanced product level assessment in seafood LCAS. *Environmental Management* **52**, 1239–1248.

Hosch, G. and Blaha, F. (2017) *Seafood traceability for fisheries compliance: country-level support for catch documentation schemes*.

Hosseini, S.A., Kaymarm, F., Behzady, S., Kamaly, E. and Darvishi, M. (2017) Drift Gillnet Selectivity for Indo-Pacific King Mackerel, Scomberomorus guttatus, Using Girth Measurements in the North of Persian Gulf. *Turkish Journal of Fisheries and Aquatic Sciences* **17**, 1145–1156.

Howell, E.A., Hoover, A., Benson, S.R., Bailey, H., Polovina, J.J., Seminoff, J.A. and Dutton, P.H. (2015) Enhancing the TurtleWatch product for leatherback sea turtles, a dynamic habitat model for ecosystem-based management. *Fisheries Oceanography* **24**, 57–68.

Huang, H.W. and Liu, K.M. (2010) Bycatch and discards by Taiwanese large-scale tuna longline fleets in the Indian Ocean. *Fisheries Research* **106**, 261–270.

Hutchings, J.A. (2000) Collapse and recovery of marine fishes. *Nature* **406**, 882–885.

Indian Ocean Tuna Commission (2019) Available Datasets. Available at: www.iotc.org/data/datasets [Accessed May 5, 2019].

Innes, J., Pascoe, S., Wilcox, C., Jennings, S. and Paredes, S. (2015) Mitigating undesirable impacts in the marine environment: a review of market-based management measures. *Frontiers in Marine Science* **2**, 1–12.

IOTC (2020) Available Datasets. Available at: https://www.iotc.org/data/datasets [Accessed March 5, 2020].

IPBES (2019) Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Bonn, Germany.

IUCN (2019) IUCN red list of threatened species. Available at: http://www.iucnredlist.org [Accessed September 13, 2019].

IUCN (2020) The IUCN Red List of Threatened Species. Available at: https://www.iucnredlist.org [Accessed September 16, 2020].

Jabado, R.W., Al Ghais, S.M., Hamza, W. and Henderson, A.C. (2015) The shark fishery in the United Arab Emirates: An interview based approach to assess the status of sharks. *Aquatic Conservation: Marine and Freshwater Ecosystems* **25**, 800–816.

Jabado, R.W., Kyne, P.M., Pollom, R.A., et al. (2018) Troubled waters: Threats and extinction risk of the sharks, rays and chimaeras of the Arabian Sea and adjacent waters. *Fish and Fisheries* **19**, 1043–1062.

Jabado, R.W. and Spaet, J.L.Y. (2017) Elasmobranch fisheries in the Arabian Seas Region: Characteristics, trade and management. *Fish and Fisheries* **18**, 1096–1118.

Jackson, J.B.C. (2001) Historical Overfishing and the Recent Collapse of Coastal Ecosystems. *Science* **293**, 629–637.

Jacquet, J. and Pauly, D. (2008) Funding priorities: Big barriers to small-scale fisheries. *Conservation Biology* **22**, 832–835.

Jennings, S. and Kaiser, M.J. (1998) *The effects of fishing on marine ecosystems*, (Vol. 34). Elsevier Masson SAS.

Jentoft, S. (2004) Institutions in fisheries: What they are, what they do, and how they change. *Marine Policy* **28**, 137–149.

Jiménez, S., Domingo, A., Winker, H., et al. (2020) Towards mitigation of seabird bycatch: Large-scale effectiveness of night setting and Tori lines across multiple pelagic longline fleets. *Biological Conservation* **247**, 108642.

Johnson, A., Salvador, G., Kenney, J., Robbins, J., Kraus, S., Landry, S. and Clapham, P. (2005) Fishing gear involved in entanglements of right and humpback whales. *Marine Mammal Science* **21**, 635–645.

Johnson, T.R. and Van Densen, W.L.T. (2007) Benefits and organization of cooperative research for fisheries management. *ICES Journal of Marine Science* **64**, 834–840.

Jones, K.R., Klein, C.J., Grantham, H.S., et al. (2020) Area Requirements to Safeguard Earth's Marine Species. *One Earth* **2**, 188–196.

Jones, K.R., Klein, C.J., Halpern, B.S., et al. (2018a) The Location and Protection Status of Earth's Diminishing Marine Wilderness. *Current Biology* **28**, 2506-2512.e3.

Jones, K.R., Klein, C.J., Halpern, B.S., Friedlander, A.M., Possingham, H.P. and Watson, J.E.M. (2018b) The Location and Protection Status of Earth's Diminishing Marine Wilderness. *Current Biology* **28**, 1–7.

Juan-Jordá, M.J., Murua, H., Arrizabalaga, H., Dulvy, N.K. and Restrepo, V. (2018) Report card on ecosystem-based fisheries management in tuna regional fisheries management organizations. *Fish and Fisheries* **19**, 321–339.

Kamilaris, A., Fonts, A. and Prenafeta-Boldύ, F.X. (2019) The rise of blockchain technology in agriculture and food supply chains. *Trends in Food Science & Technology* **91**, 640–652.

Kappel, C. V. (2005) Losing pieces of the puzzle: Threats to marine, estuarine, and diadromous species. *Frontiers in Ecology and the Environment* **3**, 275–282.

Kapsenberg, L. and Cyronak, T. (2019) Ocean acidification refugia in variable environments. *Global Change Biology* **25**, 3201–3214.

Kark, S., Tulloch, A., Gordon, A., Mazor, T., Bunnefeld, N. and Levin, N. (2015) Cross-boundary collaboration: Key to the conservation puzzle. *Current Opinion in Environmental Sustainability* **12**, 12–24.

Karnad, D. and St. Martin, K. (2020) Assembling marine spatial planning in the global south: International agencies and the fate of fishing communities in India. *Maritime Studies* **19**, 375–387.

Kaschner, K., Kesner-Reyes, K., Garilao, C., Rius-Barile, J., Rees, T. and Froese, R. (2016) AquaMaps: Predicted range maps for aquatic species. Available at: www.aquamaps.org.

Kaschner, K., Tittensor, D.P., Ready, J., Gerrodette, T. and Worm, B. (2011) Current and future patterns of global marine mammal biodiversity. *PLoS ONE* **6**.

Keane, A., Jones, J.P.G., Edwards-Jones, G. and Milner-Gulland, E.J. (2008) The sleeping policeman: Understanding issues of enforcement and compliance in conservation. *Animal Conservation* **11**, 75–82.

Kelleher, K. (2005) Discards in the world's marine fisheries - an update.

Kennelly, S.J. (2020) Bycatch Beknown: Methodology for jurisdictional reporting of fisheries discards – Using Australia as a case study. *Fish and Fisheries* **21**, 1046–1066.

Khan, M.W. (2017) Pakistan's National Report to the Scientific Committee of the Indian Ocean Tuna Commission, 2017: IOTC-2017-SC20-NR20 Rev_1.

Kincaid, K., Rose, G. and Devillers, R. (2017) How fisher-influenced marine closed areas contribute to ecosystem-based management: A review and performance indicator scorecard. *Fish and Fisheries* **18**, 860–876.

Kirby, D.S. and Ward, P. (2014) Standards for the effective management of fisheries bycatch. *Marine Policy* **44**, 419–426.

Kiszka, J., Moazzam, M., Boussarie, G., Shahid, U., Khan, B. and Naway, R. Using crew-based observer data to test a low-mitigation method to reduce cetacean bycatch in Indian Ocean gillnet fisheries. *Aquatic Conservation-Marine and Freshwater Ecosystems [In Review]*.

Kiszka, J.J. (2012) Bycatch assessment of vulnerable megafauna in coastal artisanal fisheries in the southwest Indian Ocean. *Final report for the South West Indian Ocean Fisheries Project (SWIOFP)*, 113.

Kittinger, J.N., Teh, L.C.L., Allison, E.H., et al. (2017) Committing to socially responsible seafood: *Science* **356**, 912–913.

Klein, C.J., Brown, C.J., Halpern, B.S., Segan, D.B., McGowan, J., Beger, M. and Watson, J.E.M. (2015) Shortfalls in the global protected area network at representing marine biodiversity. *Nature Scientific Reports* **5**, 17539.

Komoroske, L.M. and Lewison, R.L. (2015) Addressing fisheries bycatch in a changing world. *Frontiers in Marine Science* **2**, 1–11.

Kroodsma, D.A., Mayorga, J., Hochberg, T., et al. (2018) Tracking the global footprint of fisheries. *Science* **359**, 904–908.

Kurland, J., Pires, S.F., McFann, S.C. and Moreto, W.D. (2017) Wildlife crime: A conceptual integration, literature review, and methodological critique. *Crime Science* **6**.

Laffoley, D., Dudley, N., Jonas, H., MacKinnon, D., MacKinnon, K., Hockings, M. and Woodley, S. (2017) An introduction to 'other effective area-based conservation measures' under Aichi Target 11 of the Convention on Biological Diversity: Origin, interpretation and emerging ocean issues. *Aquatic Conservation: Marine and Freshwater Ecosystems* **27**, 130–137.

Laird, A. (2020) Northern Prawn Fishery data summary 2019. Australia.

Lam, M.E. (2016) The Ethics and Sustainability of Capture Fisheries and Aquaculture. *Journal of Agricultural and Environmental Ethics* **29**, 35–65.

Lascelles, B., Notarbartolo Di Sciara, G., Agardy, T., et al. (2014) Migratory marine species: Their status, threats and conservation management needs. *Aquatic Conservation: Marine and Freshwater Ecosystems* **24**, 111–127.

Lent, R. and Squires, D. (2017) Reducing marine mammal bycatch in global fisheries: An economics approach. *Deep-Sea Research Part II: Topical Studies in Oceanography* **140**, 268–277.

Lewis, S.G. and Boyle, M. (2017) The Expanding Role of Traceability in Seafood: Tools and Key Initiatives. *Journal of Food Science* **82**, A13–A21.

Lewison, R.L., Crowder, L.B., Read, A.J. and Freeman, S.A. (2004) Understanding impacts of fisheries bycatch on marine megafauna. *Trends in Ecology and Evolution* **19**, 598–604.

Lewison, R.L., Crowder, L.B., Wallace, B.P., et al. (2014) Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. *Proceedings of the National Academy of Sciences of the United States of America* **111**, 5271–6.

Lezama-Ochoa, N., Murua, H., Chust, G., et al. (2015) Biodiversity in the by-catch communities of the pelagic ecosystem in the Western Indian Ocean. *Biodiversity and Conservation* **24**, 2647–2671.

Lin, C.Y., Wang, S.P., Chiang, W.C., Griffiths, S. and Yeh, H.M. (2020) Ecological risk assessment of species impacted by fisheries in waters off eastern Taiwan. *Fisheries Management and Ecology* **27**, 345–356.

Link, J.S. and Watson, R.A. (2019) Global ecosystem overfishing: Clear delineation within real limits to production. *Science Advances* **1**, 1–12.

Lokina, R.B. (2009) Technical efficiency and the role of skipper skill in artisanal Lake Victoria fisheries. *Environment and Development Economics* **14**, 497–519.

Mace, G.M., Collar, N.J., Gaston, K.J., et al. (2008) Quantification of extinction risk: IUCN's system for classifying threatened species. *Conservation Biology* **22**, 1424–1442.

Mackay, M., Jennings, S., van Putten, E.I., Sibly, H. and Yamazaki, S. (2018) When push comes to shove in recreational fishing compliance, think 'nudge.' *Marine Policy* **95**, 256–266.

Mackelworth, P. (2012) Peace parks and transboundary initiatives: Implications for marine conservation and spatial planning. *Conservation Letters* **5**, 90–98.

Maguire, J.-J., Sissenwine, M., Csirke, J., Grainger, R. and Garcia, S. (2006) The state of world highly migratory, straddling and other high seas fishery resources and associated species.

Le Manach, F., Chaboud, C., Copeland, D., et al. (2013) European union's public fishing access agreements in developing countries. *PLoS ONE* **8**.

Mangi, S.C., Dolder, P.J., Catchpole, T.L., Rodmell, D. and de Rozarieux, N. (2015) Approaches to fully documented fisheries: Practical issues and stakeholder perceptions. *Fish and Fisheries* **16**, 426–452.

Mangin, T., Costello, C., Anderson, J., et al. (2018) Are fishery management upgrades worth the cost? *PLoS ONE* **13**, 1–24.

Marchal, P., Andersen, B., Bromley, D., et al. (2006) Improving the definition of fishing effort for important European fleets by accounting for the skipper effect. *Canadian Journal of Fisheries and Aquatic Sciences* **63**, 510–533.

Marinesque, S., Kaplan, D.M. and Rodwell, L.D. (2012) Global implementation of marine protected areas: Is the developing world being left behind? *Marine Policy* **36**, 727–737.

Mason, N., Ward, M., Watson, J.E.M., Venter, O. and Runting, R.K. (2020) Global opportunities and challenges for transboundary conservation. *Nature Ecology & Evolution*.

Matsuoka, T., Nakashima, T. and Nagasawa, N. (2005) A review of ghost fishing: scientific approaches to evaluation and solutions. *Fisheries Science* **71**, 691–702.

Maureaud, A., Frelat, R., Pécuchet, L., et al. (2020) Are we ready to track climate-driven shifts in marine species across international boundaries? - A global survey of scientific bottom trawl data. *Global Change Biology*, 1–17.

Mazaris, A.D., Schofield, G., Gkazinou, C., Almpanidou, V. and Hays, G.C. (2017) Global sea turtle conservation successes. *Science Advances* **3**, e1600730.

McCauley, D., Pinsky, M., Palumbi, S., Estes, J., Joyce, F. and Warner, R. (2015) Marine defaunation: Animal loss in the global ocean. *Science* **347**, 6219.

McCauley, D.J., Jablonicky, C., Allison, E.H., Golden, C.D., Joyce, F.H., Mayorga, J. and Kroodsma, D. (2018) Wealthy countries dominate industrial fishing. *Science Advances* **4**, eaau2161.

McClenachan, L., Cooper, A.B., Carpenter, K.E. and Dulvy, N.K. (2012) Extinction risk and bottlenecks in the conservation of charismatic marine species. *Conservation Letters* **5**, 73–80.

McClenachan, L., Cooper, A.B. and Dulvy, N.K. (2016) Rethinking Trade-Driven Extinction Risk in Marine and Terrestrial Megafauna. *Current Biology* **26**, 1640–1646.

McCluney, J.K., Anderson, C.M. and Anderson, J.L. (2019) The fishery performance indicators for global tuna fisheries. *Nature Communications* **10**, 1–9.

Melnychuk, M.C., Kurota, H., Mace, P.M., et al. (2021) Identifying management actions that promote sustainable fisheries. *Nature Sustainability*.

Melnychuk, M.C., Peterson, E., Elliott, M. and Hilborn, R. (2017) Fisheries management impacts on target species status. *Proceedings of the National Academy of Sciences* **114**, 178–183.

Micheli, F., Leo, G. De, Butner, C., Martone, R.G. and Shester, G. (2014) A risk-based framework for assessing the cumulative impact of multiple fisheries. *Biological Conservation* **176**, 224–235.

Miller, A.M.M., Bush, S.R. and Mol, A.P.J. (2014) Power Europe: EU and the illegal, unreported and unregulated tuna fisheries regulation in the West and Central Pacific Ocean. *Marine Policy* **45**, 138–145.

Miller, N.A., Roan, A., Hochberg, T., Amos, J. and Kroodsma, D.A. (2018a) Identifying Global Patterns of Transshipment Behavior. *Frontiers in Marine Science* **5**, 1–9.

Miller, R.L., Marsh, H., Cottrell, A. and Hamann, M. (2018b) Protecting migratory species in the australian marine environment: A cross-jurisdictional analysis of policy and management plans. *Frontiers in Marine Science* **5**, 1–13.

Mills, D.J., Westlund, L., Graaf, G. de, Kura, Y., Willman, R. and Kelleher, K. (2011) Under-reported and undervalued: small-scale fisheries in the developing world. In: *Small-scale fisheries management: frameworks and approaches for the developing world*. pp 1–15.

Di Minin, E., Leader-Williams, N. and Bradshaw, C.J.A. (2016) Banning Trophy Hunting Will Exacerbate Biodiversity Loss. *Trends in Ecology and Evolution* **31**, 99–102.

Moazzam, M. (2012) Status report on bycatch of tuna gillnet operations in Pakistan. IOTC–2012–WPEB08–13. Cape Town.

Mobsby, D. (2018) Australian fisheries and aquaculture statistics 2017, Fisheries Research and Development Corporation project 2018-134. Canbera.

Moore, J.E., Curtis, K.A., Lewison, R.L., et al. (2013) Evaluating sustainability of fisheries bycatch mortality for marine megafauna: A review of conservation reference points for data-limited populations. *Environmental Conservation* **40**, 329–344.

Moore, J.E., Wallace, B.P., Lewison, R.L., Žydelis, R., Cox, T.M. and Crowder, L.B. (2009) A review of marine mammal, sea turtle and seabird bycatch in USA fisheries and the role of policy in shaping management. *Marine Policy* **33**, 435–451.

Mora-Soto, A., Palacios, M., Macaya, E.C., et al. (2020) A high-resolution global map of giant kelp (Macrocystis pyrifera) forests and intertidal green algae (Ulvophyceae) with sentinel-2 imagery. *Remote Sensing* **12**, 1–20.

Mora, C., Myers, R.A., Coll, M., et al. (2009) Management Effectiveness of the World's Marine Fisheries. *Plos Biology* **7**.

Moreno, G. and Herrera, M. (2013) Estimation of fishing capacity by tuna fishing fleets in the Indian Ocean. IOTC–2013–SC16–INF04. Victoria, Seychelles.

Murawski, S.A. (2000) Definitions of overfishing from an ecosystem perspective. *ICES Journal of Marine Science* **57**, 649–658.

Murua, H., Santiago, J., Coelho, R., et al. (2018) Updated Ecological Risk Assessment (ERA) for shark species caught in fisheries managed by the Indian Ocean Tuna Commission (IOTC).

Nelson, M.P., Bruskotter, J.T., Vucetich, J.A. and Chapron, G. (2016) Emotions and the Ethics of Consequence in Conservation Decisions: Lessons from Cecil the Lion. *Conservation Letters* **9**, 302–306.

Neubauer, P., Jensen, O.P., Hutchings, J.A. and Baum, J.K. (2013) Resilience and recovery of overexploited marine populations. *Science* **340**, 347–349.

Northridge, S., Coram, A., Kingston, A. and Crawford, R. (2017) Disentangling the causes of protected-species bycatch in gillnet fisheries. *Conservation Biology* **31**, 686–695.

Novianto, D., Nugroho, A.F. and Zedta, R.R. (2016) Composition and abundance of pelagic shark caught by drift gillnet in Cilacap oceanic fishing port, Indonesia. IOTC-2016-WPEB12-17 COMPOSITION.

O'Hara, C.C., Afflerbach, J.C., Scarborough, C., Kaschner, K. and Halpern, B.S. (2017) Aligning marine species range data to better serve science and conservation. *Plos One* **12**, e0175739.

O'Hara, C.C., Villaseñor-Derbez, J.C., Ralph, G.M. and Halpern, B.S. (2019) Mapping status and conservation of global at-risk marine biodiversity. *Conservation Letters*, e12651.

O'Keefe, C.E., Cadrin, S.X. and Stokesbury, D.. (2014) Evaluating effectiveness of time/area closures, quotas/caps, and fleet communications to reduce fisheries bycatch. *ICES Journal of*

Marine Science **70**, 883–891.

O'Leary, B.C. and Roberts, C.M. (2018) Ecological connectivity across ocean depths: Implications for protected area design. *Global Ecology and Conservation* **15**, e00431.

OBIS (2020) Ocean Biogeographic Information System.

Oliver, S., Braccini, M., Newman, S.J. and Harvey, E.S. (2015) Global patterns in the bycatch of sharks and rays. *Marine Policy* **54**, 86–97.

Ortiz, N., Mangel, J.C., Wang, J., et al. (2016) Reducing green turtle bycatch in small-scale fisheries using illuminated gillnets: The cost of saving a sea turtle. *Marine Ecology Progress Series* **545**, 251–259.

Österblom, H. (2014) Catching up on fisheries crime. *Conservation Biology* **28**, 877–879.

Österblom, H., Jouffray, J.-B. and Spijkers, J. (2016) Where and how to prioritize fishery reform? *Proceedings of the National Academy of Sciences* **113**, E3473–E3474.

Osterblom, H., Jouffray, J.B., Folke, C., Crona, B., Troell, M., Merrie, A. and Rockström, J. (2015) Transnational corporations as "keystone actors" in marine ecosystems. *PLoS ONE* **10**, 1–15.

Österblom, H., Sissenwine, M., Symes, D., Kadin, M., Daw, T. and Folke, C. (2011) Incentives, social-ecological feedbacks and European fisheries. *Marine Policy* **35**, 568–574.

Palacios-Abrantes, J., Reygondeau, G., Wabnitz, C.C.C. and Cheung, W.W.L. (2020) The transboundary nature of the world's exploited marine species. *Scientific Reports* **10**, 1–12.

Parker, D., Winker, H., Silva, C., Mketsu, Q. and Kerwath, S. (2018) South African national report to the Scientific Committee of the Indian Ocean Tuna Commission, 2018. Cape Town.

Parsa, M., Emery, T.J., Williams, A.J. and Nicol, S. (2020) An empirical Bayesian approach for estimating fleet-and vessel-level bycatch rates in fisheries with effort heterogeneity and limited data: A prospective tool for measuring bycatch mitigation performance. *ICES Journal of Marine Science* **77**, 921–929.

Pascoe, S., Coglan, L., Punt, A.E. and Dichmont, C.M. (2012) Impacts of Vessel Capacity Reduction Programmes on Efficiency in Fisheries: The Case of Australia's Multispecies Northern Prawn Fishery. *Journal of Agricultural Economics* **63**, 425–443.

Pascoe, S.D., Plagányi, É.E. and Dichmont, C.M. (2017) Modelling multiple management objectives in fisheries: Australian experiences. *ICES Journal of Marine Science* **74**, 464–474.

Patrick, W.S., Spencer, P., Link, J., et al. (2010) Using productivity and susceptibility indices to assess the vulnerability of united states fish stocks to overfishing. *Fishery Bulletin* **108**, 305–

322.

Patterson, H., Noriega, R., Georgeson, L., Larcombe, J. and Curtotti, R. (2017) Fishery Status Reports 2017. Canberra.

Pauly (2013) Beyond Duplicity And Ignorance In Global Fisheries. In: *WIT Transactions on State of the Art in Science and Engineering*, Vol. 64. WIT Press, pp 121–132.

Pauly, D. (2017) A vision for marine fisheries in a global blue economy. *Marine Policy*, 0–1.

Pauly, D., Christensen, V., Guénette, S., et al. (2002) Towards sustainability in world fisheries. *Nature* **418**, 689–695.

Pauly, D. and Zeller, D. (2019) Agreeing with FAO: Comments on SOFIA 2018. *Marine Policy* **100**, 332–333.

Pauly, D., Zeller, D. and Palomares, M.L.D. (2020) Sea Around Us Concepts, Design and Data. Available at: seaaroundus.org.

Peel, D. and McLachlan, G.J. (2000) Robust mixture modelling using the t distribution. *Statistics and Computing* **10**, 339–348.

Petrossian, G.A. (2014) Preventing illegal, unreported and unregulated (IUU) fishing: A situational approach. *Biological Conservation* **189**, 39–48.

Phillips, K., Giannini, F., Lawrence, E. and Bensley, N. (2010) Cumulative assessment of the catch of non-target species in Commonwealth fisheries: a scoping study. Canberra.

Pikitch, E.K., Santora, C., Babcock, E.A., et al. (2004) Ecosystem-Based Fishery Management. *Science* **305**, 346–348.

Pinsky, M.L., Jensen, O.P., Ricard, D. and Palumbi, S.R. (2011) Unexpected patterns of fisheries collapse in the world's oceans. *Proceedings of the National Academy of Sciences* **108**, 8317–8322.

Pinsky, M.L., Worm, B., Fogarty, M.J., Sarmiento, J.L. and Levin, S.A. (2013) Marine Taxa Track Local Climate Velocities. *Science* **341**, 1239–1242.

Pitcher, T., Kalikoski, D., Pramod, G. and Short, K. (2009) Not honouring the code. *Nature* **457**, 658–659.

Pitcher, T.J. and Cheung, W.W.L. (2013) Fisheries: Hope or despair? *Marine Pollution Bulletin* **74**, 506–516.

Pons, M., Branch, T.A., Melnychuk, M.C., et al. (2017) Effects of biological, economic and management factors on tuna and billfish stock status. *Fish and Fisheries* **18**, 1–21.

Pons, M., Melnychuk, M.C. and Hilborn, R. (2018) Management effectiveness of large pelagic fisheries in the high seas. *Fish and Fisheries* **19**, 260–270.

Poonian, C.N.S. (2015) A first assessment of elasmobranch catch in Mauritian artisanal fisheries using interview surveys. *Phelsuma* **23**, 19–29.

Porszt, E.J., Peterman, R.M., Dulvy, N.K., Cooper, A.B. and Irvine, J.R. (2012) Reliability of Indicators of Decline in Abundance. *Conservation Biology* **26**, 894–904.

Powles, H., Bradford, M.J., Bradford, M.R.G., Doubleday, M.W.G., Innes, M.S. and Levings, C.D. (2000) Assessing and protecting endangered marine species. *ICES Journal of Marine Science* **57**, 669–676.

Probst, W.N. (2019) How emerging data technologies can increase trust and transparency in fisheries. *ICES Journal of Marine Science* **77**, 1286–1294.

Purvis, A., Gittleman, J.L., Cowlishaw, G. and Mace, G.M. (2000) Predicting extinction risk in declining species. *Proceedings of the Royal Society B: Biological Sciences* **267**, 1947–1952.

Putt, J. and Anderson, K. (2007) A national study of crime in the Australian fishing industry.

Van Putten, I.E., Kulmala, S., Thébaud, O., Dowling, N., Hamon, K.G., Hutton, T. and Pascoe, S. (2012) Theories and behavioural drivers underlying fleet dynamics models. *Fish and Fisheries* **13**, 216–235.

R Core Team (2019) R: A language and environment for statistical computing.

Ramesh, N., Rising, J.A. and Oremus, K.L. (2019) Consequences of larval dispersal. *Science* **1196**, 1192–1196.

Ramp, C., Gaspard, D., Gavrilchuk, K., et al. (2021) Up in the air: drone images reveal underestimation of entanglement rates in large rorqual whales. *Endangered Species Research* **44**, 33–44.

Rasmussen, A.R., Murphy, J.C., Ompi, M., Gibbons, J.W. and Uetz, P. (2011) Marine reptiles. *PLoS ONE* **6**.

Rassweiler, A., Costello, C. and Siegel, D.A. (2012) Marine protected areas and the value of spatially optimized fishery management. *PNAS* **109**, 11884–11889.

Read, A.J., Drinker, P. and Northridge, S. (2006) Bycatch of marine mammals in U.S. and global fisheries. *Conservation Biology* **20**, 163–169.

Ready, J., Kaschner, K., South, A.B., et al. (2010) Predicting the distributions of marine organisms at the global scale. *Ecological Modelling* **221**, 467–478 Contents.

Reddy, S.M.W., Montambault, J., Masuda, Y.J., et al. (2017) Advancing Conservation by Understanding and Influencing Human Behavior. *Conservation Letters* **10**, 248–256.

Reeves, R.R., McClellan, K. and Werner, T.B. (2013) Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. *Endangered Species Research* **20**, 71–97.

Regional seas programmes (2020) Available at: https://www.unenvironment.org/explore-topics/oceans-seas/what-we-do/working-regional-seas/regional-seas-programmes [Accessed July 1, 2020].

Reiss, H., Hoarau, G., Dickey-Collas, M. and Wolff, W.J. (2009) Genetic population structure of marine fish: Mismatch between biological and fisheries management units. *Fish and Fisheries* **10**, 361–395.

Reynolds, J.D., Dulvy, N.K., Goodwin, N.B. and Hutchings, J.A. (2005a) Biology of extinction risk in marine fishes. *Proceedings of the Royal Society B: Biological Sciences* **272**, 2337–2344.

Reynolds, J.D., Dulvy, N.K., Goodwin, N.B. and Hutchings, J.A. (2005b) Biology of extinction risk in marine fishes. *Proceedings of the Royal Society B: Biological Sciences* **272**, 2337–2344.

Ricard, D., Minto, C., Jensen, O.P. and Baum, J.K. (2012) Examining the knowledge base and status of commercially exploited marine species with the RAM Legacy Stock Assessment Database. *Fish and Fisheries* **13**, 380–398.

Ripple, W.J., Wolf, C., Newsome, T.M., et al. (2019) Are we eating the world's megafauna to extinction? *Conservation Letters* **12**, e12627.

Riskas, K.A., Fuentes, M.M.P.B. and Hamann, M. (2016) Justifying the need for collaborative management of fisheries bycatch: A lesson from marine turtles in Australia. *Biological Conservation* **196**, 40–47.

Roberson, L.A., Kiszka, J.J. and Watson, J.E.M. (2019) Need to address gaps in global fisheries observation. *Conservation Biology* **0**, cobi.13265.

Rogers, A., Galland, G., Nickson, A., Macfadyen, G., Huntington, T. and Defaux, V. (2016) Netting billions: a global valuation of tuna.

Roheim, C.A., Bush, S.R., Asche, F., Sanchirico, J.N. and Uchida, H. (2018) Evolution and future of the sustainable seafood market. *Nature Sustainability* **1**, 392–398.

Román-Verdesoto, M. and Orozco-Zöller, M. (2005) Bycatches of sharks in the tuna purse-seine fishery of the Eastern Pacific Ocean reported by observers of the Inter-American Tropical Tuna Commission, 1993-2004.

Romanov, E. V. (2002) Bycatch in the tuna purse-seine fisheries of the western Indian Ocean. *Fishery Bulletin* **100**, 90–105.

Roulin, A., Abu Rashid, M., Spiegel, B., Charter, M., Dreiss, A.N. and Leshem, Y. (2017) 'Nature Knows No Boundaries': The Role of Nature Conservation in Peacebuilding. *Trends in Ecology and Evolution* **32**, 305–310.

Rousseau, Y. (2020) Predicting the future of global seafood production.

Rousseau, Y., Watson, R.A., Blanchard, J.L. and Fulton, E.A. (2019) Evolution of global marine fishing fleets and the response of fished resources. *Proceedings of the National Academy of Sciences* **116**.

Runge, C.A., Martin, T.G., Possingham, H.P., Willis, S.G. and Fuller, R.A. (2014) Conserving mobile species. *Frontiers in Ecology and the Environment* **12**, 395–402.

Russell, S.D. and Alexander, R.T. (1996) The skipper effect debate: Views from a Philippine fishery. *Journal of Anthropological Research* **52**, 433–460.

Ruttan, L.M. and Tyedmers, P.H. (2007) Skippers, spotters and seiners: Analysis of the "skipper effect" in US menhaden (Brevoortia spp.) purse-seine fisheries. *Fisheries Research* **83**, 73–80.

Sala, E., Lubchenco, J., Grorud-Colvert, K., Novelli, C., Roberts, C. and Sumaila, U.R. (2018a) Assessing real progress towards effective ocean protection. *Marine Policy* **91**, 11–13.

Sala, E., Mayorga, J., Costello, C., et al. (2018b) The economics of fishing the high seas. *Science Advances*, 1–14.

Salomon, A.K., Gaichas, S.K., Jensen, O.P., et al. (2011) Bridging the divide between fisheries and marine conservation science. *Bulletin of Marine Science* **87**, 251–274.

Sampson, D.B. (2011) The accuracy of self-reported fisheries data: Oregon trawl logbook fishing locations and retained catches. *Fisheries Research* **112**, 59–76.

Savoca, M.S., Brodie, S., Welch, H., Hoover, A., Benaka, L.R., Bograd, S.J. and Hazen, E.L. (2020) Comprehensive bycatch assessment in US fisheries for prioritizing management. *Nature Sustainability* **3**, 472–480.

Scales, K.L., Hazen, E.L., Jacox, M.G., Castruccio, F., Maxwell, S.M., Lewison, R.L. and Bograd, S.J. (2018) Fisheries bycatch risk to marine megafauna is intensified in Lagrangian coherent structures. *Proceedings of the National Academy of Sciences of the United States of America* **115**, 7362–7367.

Scales, K.L., Hazen, E.L., Maxwell, S.M., et al. (2017) Fit to predict? Eco-informatics for predicting the catchability of a pelagic fish in near real time. *Ecological Applications* **27**, 2313–2329.

Scholtens, J. and Bavinck, M. (2014) Lessons for legal pluralism: Investigating the challenges of transboundary fisheries governance. *Current Opinion in Environmental Sustainability* **11**, 10–18.

Di Sciara, G.N., Hoyt, E., Reeves, R., Ardron, J., Marsh, H., Vongraven, D. and Barr, B. (2016) Place-based approaches to marine mammal conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems* **26**, 85–100.

Sekadende, B., Scott, L., Anderson, J., et al. (2020) The small pelagic fishery of the Pemba Channel, Tanzania: What we know and what we need to know for management under climate change. *Ocean and Coastal Management* **197**.

Selig, E.R., Kleisner, K.M., Ahoobim, O., et al. (2017) A typology of fisheries management tools: using experience to catalyse greater success. *Fish and Fisheries* **18**, 543–570.

Selig, E.R., Turner, W.R., Troëng, S., et al. (2014) Global priorities for marine biodiversity conservation. *PLoS ONE* **9**, 1–12.

Senko, J., White, E.R., Heppell, S.S. and Gerber, L.R. (2014) Comparing bycatch mitigation strategies for vulnerable marine megafauna. *Animal Conservation* **17**, 5–18.

Sequeira, A.M.M., Hays, G.C., Sims, D., et al. (2019) Overhauling Ocean Spatial Planning to Improve Marine Megafauna Conservation. *Frontiers in Ecology and the Environment* **6**.

Shahid, U., Khan, M.M., Nawaz, R., Abdul Razzaq, S. and Ayub, S. (2016) Bycatch analysis of tuna gillnet fisheries of Pakistan: An analysis of bycatch data from 2013-2015. IOTC-2016-WPEB12-INF11.

Shahid, U., Khan, M.M., Nawaz, R., Dimmlich, W. and Kiszka, J. (2015) A preliminary assessment of shark bycatch in tuna gillnet fisheries of Pakistan (Arabian Sea). IOTC-2015-WPEB!!-46.

Shiffman, D.S., Gallagher, A.J., Wester, J., Macdonald, C.C. and Thaler, A.D. (2014) Trophy fishing for species threatened with extinction: A way forward building on a history of conservation. *Marine Policy* **50**, 318–322.

Shono, H. (2008) Application of the Tweedie distribution to zero-catch data in CPUE analysis. *Fisheries Research* **93**, 154–162.

Simpfendorfer, C.A. and Dulvy, N.K. (2017) Bright spots of sustainable shark fishing. *Current Biology* **27**, R97–R98.

Sinan, H. and Bailey, M. (2020) Understanding Barriers in Indian Ocean Tuna Commission Allocation Negotiations on Fishing Opportunities. *Sustainability* **12**.

Sky, M.B. (2010) Getting On The list : Politics And Procedural Maneuvering In Cites Appendix I and II Decisions For Commercially Exploited Marine And Timber Species. *Sustainable Development Law & Policy* **10**, 35–40.

Slabbekoorn, H., Bouton, N., van Opzeeland, I., Coers, A., ten Cate, C. and Popper, A.N. (2010) A noisy spring: The impact of globally rising underwater sound levels on fish. *Trends in Ecology and Evolution* **25**, 419–427.

Smart, J.J., Chin, A., Baje, L., et al. (2016) Effects of including misidentified sharks in life history analyses: A case study on the Grey Reef shark Carcharhinus amblyrhynchos from papua new Guinea. *PLoS ONE* **11**, 1–18.

Smith, H. and Basurto, X. (2019) Defining Small-Scale Fisheries and Examining the Role of Science in Shaping Perceptions of Who and What Counts: A Systematic Review. *Frontiers in Marine Science* **6**.

Smith, M.D. (2012) The New Fisheries Economics: Incentives Across Many Margins. *Annual Review of Resource Economics* **4**, 379–402.

Song, A.M., Scholtens, J., Stephen, J., Bavinck, M. and Chuenpagdee, R. (2017) Transboundary research in fisheries. *Marine Policy* **76**, 8–18.

Song, L., Zhou, J., Zhou, Y., Nishida, T., Jiang, W. and Wang, J. (2009) Environmental preferences of bigeye tuna, Thunnus obesus, in the Indian Ocean: An application to a longline fishery. *Environmental Biology of Fishes* **85**, 153–171.

Soykan, C.U., Moore, J.E., Ždelis, R., Crowder, L.B., Safina, C. and Lewison, R.L. (2008) Why study bycatch? An introduction to the Theme Section on fisheries bycatch. *Endangered Species Research* **5**, 91–102.

Spalding, M.D., Fox, H.E., Allen, G.R., et al. (2007) Marine Ecoregions of the World : A Bioregionalization of Coastal and Shelf Areas. *Bioscience* **57**, 573–583.

Spijkers, J., Singh, G., Blasiak, R., Morrison, T.H., Le Billon, P. and Österblom, H. (2019) Global patterns of fisheries conflict: Forty years of data. *Global Environmental Change* **57**.

Squires, D. and Garcia, S. (2018) The least-cost biodiversity impact mitigation hierarchy with a focus on marine fisheries and bycatch issues. *Conservation Biology* **32**, 989–997.

Squires, D. and Kirkley, J. (2011) Skipper skill and panel data in fishing industries. *Canadian Journal of Fisheries and Aquatic Sciences* **56**, 2011–2018.

Squires, D., Restrepo, V., Garcia, S. and Dutton, P. (2018) Fisheries bycatch reduction within the least-cost biodiversity mitigation hierarchy: Conservatory offsets with an application to sea turtles. *Marine Policy* **93**, 55–61.

Stanley, R.D., Karim, T., Koolman, J. and Mcelderry, H. (2015) Design and implementation ofelectronic monitoring in the British Columbia groundfish hook and line fishery: a retrospective view of the ingredients of success. *ICES Journal of Marine Science* **72**, 1230–1236.

Stanley, R.D., Mcelderry, H., Mawani, T. and Koolman, J. (2011) The advantages of an audit over a census approach to the review of video imagery in fishery monitoring.

Steneck, R. (2009) Marine Conservation: Moving Beyond Malthus. *Current Biology* **19**, 117–119.

Stephenson, R.L., Paul, S., Pastoors, M.A., et al. (2016) Integrating fishers' knowledge research in science and management Robert. *ICES Journal of Marine Science* **73**, 1459–1465.

Stequert, B. and Marsac, F. (1989) Tropical tuna - surface fisheries in the Indian Ocean: FAO Technical Paper. Rome.

Stern, P.C. (2000) New Environmental Theories: Toward a Coherent Theory of Environmentally Significant Behavior. *Journal of Social Issues* **56**, 407–424.

Stobutzki, I.C., Miller, M.J., Jones, J.P. and Salini., J.P. (2001) Bycatch diversity and variation in penaeid fisheries; the implications for monitoring. *Fisheries Research* **53**, 283–301.

Studds, C.E., Kendall, B.E., Murray, N.J., et al. (2017) Rapid population decline in migratory shorebirds relying on Yellow Sea tidal mudflats as stopover sites. *Nature Communications* **8**, 1–7.

Sumaila, U.R., Cheung, W., Dyck, A., et al. (2012) Benefits of rebuilding global marine fisheries outweigh costs. *PLoS ONE* **7**, e40542.

Sumaila, U.R., Ebrahim, N., Schuhbauer, A., et al. (2019a) Updated estimates and analysis of global fisheries subsidies. *Marine Policy* **109**, 103695.

Sumaila, U.R., Ebrahim, N., Schuhbauer, A., et al. (2019b) Updated estimates and analysis of global fisheries subsidies. *Marine Policy* **109**, 103695.

Sumaila, U.R., Lam, V., Le Manach, F., Swartz, W. and Pauly, D. (2016) Global fisheries subsidies: An updated estimate. *Marine Policy* **69**, 189–193.

Sutinen, J.G. and Kuperan, K. (1999) A socio-economic theory of regulatory compliance. *International Journal of Social Economics* **26**, 174–193.

Suuronen, P. and Gilman, E. (2020) Monitoring and managing fisheries discards: New technologies and approaches. *Marine Policy* **116**, 103554.

Swartz, W., Sala, E., Tracey, S., Watson, R. and Pauly, D. (2010) The spatial expansion and ecological footprint of fisheries (1950 to present). *PLoS ONE* **5**, 3–8.

Swimmer, Y., Zollett, E.A. and Gutierrez, A. (2020) Bycatch mitigation of protected and threatened species in tuna purse seine and longline fisheries. *Endangered Species Research* **43**, 517–542.

Tai, T.C., Cashion, T., Lam, V.W.Y., Swartz, W. and Sumaila, U.R. (2017) Ex-vessel fish price database: Disaggregating prices for low-priced species from reduction fisheries. *Frontiers in Marine Science* **4**, 1–10.

Teh, L.C.L., Caddell, R., Allison, E.H., Kittinger, J.N., Nakamura, K. and Ota, Y. (2019) *The relevance of human rights to socially responsible seafood*. Elsevier Inc.

Teh, L.S.L., Teh, L.C.L., Hines, E., Junchompoo, C. and Lewison, R.L. (2015) Contextualising the coupled socio-ecological conditions of marine megafauna bycatch. *Ocean and Coastal Management* **116**, 449–465.

Telesetsky, A. (2017) U.S. Seafood Traceability as Food Law and the Future of Marine Fisheries. *Environmental Law* **47**, 765–795.

Temple, A.J., Kiszka, J.J., Stead, S.M., et al. (2018) Marine megafauna interactions with small-scale fisheries in the southwestern Indian Ocean: a review of status and challenges for research and management. *Reviews in Fish Biology and Fisheries*.

Temple, A.J., Wambiji, N., Poonian, C.N.S., Jiddawi, N., Stead, S.M., Kiszka, J.J. and Berggren, P. (2019) Marine megafauna catch in southwestern Indian Ocean small-scale fisheries from landings data. *Biological Conservation* **230**, 113–121.

Thomas, A.S., Milfont, T.L. and Gavin, M.C. (2016) New approach to identifying the drivers of regulation compliance using multivariate behavioural models. *PLoS ONE* **11**, 1–12.

Tickler, D., Meeuwig, J.J., Bryant, K., et al. (2018a) Modern slavery and the race to fish. *Nature Communications* **9**.

Tickler, D., Meeuwig, J.J., Palomares, M.L., Pauly, D. and Zeller, D. (2018b) Far from home: Distance patterns of global fishing fleets. *Science Advances* **4**, 4–10.

Tidd, A., Brouwer, S. and Pilling, G. (2017) Shooting fish in a barrel? Assessing fisher-driven changes in catchability within tropical tuna purse seine fleets. *Fish and Fisheries* **18**, 808–820.

Tittensor, D.P., Mora, C., Jetz, W., Lotze, H.K., Ricard, D., Berghe, E. Vanden and Worm, B.

(2010) Global patterns and predictors of marine biodiversity across taxa. *Nature* **466**, 1098–1101.

Toonen, H.M. and Bush, S.R. (2018) The digital frontiers of fisheries governance: fish attraction devices, drones and satellites. *Journal of Environmental Policy and Planning* **0**, 1–13.

Trebilco, R., Gales, R., Lawrence, E., Alderman, R., Robertson, G. and Baker, G.B. (2010) Characterizing seabird bycatch in the eastern Australian tuna and billfish pelagic longline fishery in relation to temporal, spatial and biological influences. *Aquatic Conservation: Marine and Freshwater Ecosystems* **20**, 531–542.

Tuck, G., Knuckey, I. and Klaer, N. (2013) Informing the review of the Commonwealth Policy on Fisheries Bycatch through assessing trends in bycatch of key Commonwealth fisheries. Canberra.

Tulloch, V., Pirotta, V., Grech, A., et al. (2020) *Long-term trends and a risk analysis of cetacean entanglements and bycatch in fisheries gear in Australian waters*, (Vol. 29).

UNEP-WCMC (2019) The checklist of CITES species website.

United Nations (2020) Intergovernmental Conference on an international legally binding instrument under the United Nations Convention on the Law of the Sea on the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction (General Ass. Available at: https://www.un.org/bbnj/ [Accessed March 13, 2020].

Varghese, S.P., Varghese, S. and Somvanshi, V.S. (2010) Impact of tuna longline fishery on the sea turtles of Indian seas. *Current Science* **98**, 1378–1384.

Vázquez-Rowe, I. and Tyedmers, P. (2013) Identifying the importance of the "skipper effect" within sources of measured inefficiency in fisheries through data envelopment analysis (DEA). *Marine Policy* **38**, 387–396.

Venturelli, P.A., Hyder, K. and Skov, C. (2017) Angler apps as a source of recreational fisheries data: opportunities, challenges and proposed standards. *Fish and Fisheries* **18**, 578–595.

Vierros, M.K. and Harden-Davies, H. (2020) Capacity building and technology transfer for improving governance of marine areas both beyond and within national jurisdiction. *Marine Policy*, 104158.

Vincent, A.C.J., Sadovy de Mitcheson, Y.J., Fowler, S.L. and Lieberman, S. (2014) The role of CITES in the conservation of marine fishes subject to international trade. *Fish and Fisheries* **15**, 563–592.

Viswanathan, K., Omar, I.H., Jeon, Y., Kirkley, J., Squires, D. and Susilowati, I. (2002) Fishing

Skill in Developing Country Fisheries: The Kedah, Malaysia Trawl Fishery. *Marine Resource Economics* **16**, 293–314.

Wakefield, C.B., Hesp, S.A., Blight, S., Molony, B.W., Newman, S.J. and Hall, N.G. (2018) Uncertainty associated with total bycatch estimates for rarely-encountered species varies substantially with observer coverage levels: Informing minimum requirements for statutory logbook validation. *Marine Policy* **95**, 273–282.

Wallace, B.P., Kot, C.Y., DiMatteo, A.D., Lee, T., Crowder, L.B. and Lewison, R.L. (2013) Impacts of fisheries bycatch on marine turtle populations worldwide: toward conservation and research priorities. *Ecosphere* **4**, art40.

Wallace, B.P., Lewison, R.L., Mcdonald, S.L., et al. (2010) Global patterns of marine turtle bycatch. *Conservation Letters* **3**, 131–142.

Warner, R.M. (2014) Conserving marine biodiversity in areas beyond national jurisdiction: Co-evolution and interaction with the law of the sea. *Frontiers in Marine Science* **1**, 1–11.

Watson, R. and Pauly, D. (2001) Systematic distortions in world fisheries catch trends. *Nature* **414**, 534–536.

Watson, R.A., Green, B.S., Tracey, S.R., Farmery, A. and Pitcher, T.J. (2016) Provenance of global seafood. *Fish and Fisheries* **17**, 585–595.

Watson, R.A. and Tidd, A. (2018) Mapping nearly a century and a half of global marine fishing: 1869–2015. *Marine Policy* **93**, 171–177.

Wayte, S., Dowdney, J., Williams, A., Bulman, C., Sporcic, M., Fuller, M. and Smith, T. (2007) Ecological Risk Assessment for Effects of Fishing: Report for the otter trawl sub-fishery of the Commonwealth trawl sector of the Southern and Eastern Scalefish and Shark Fishery. Canberra, Australia.

Webb, T.J. and Mindel, B.L. (2015) Global patterns of extinction risk in marine and non-marine systems. *Current Biology* **25**, 506–511.

White, C. and Costello, C. (2014) Close the High Seas to Fishing? *PLoS Biology* **12**, 1–5.

Wilcox, C. and Donlan, J. (2011) Compensatory fisheries conflicts mitigation as a solution conservation to. *Ecological Society of America* **5**, 325–331.

Wilen, J.E., Smith, M.D., Lockwood, D. and Botsford, L.W. (2002) Avoiding surprises: Incorporating fisherman behavior into management models. *Bulletin of Marine Science* **70**, 553–575.

Willette, D.A. and Cheng, S.H. (2018) Delivering on seafood traceability under the new U.S. import monitoring program. *Ambio* **47**, 25–30.

Williams, A.J., Georgeson, L., Summerson, R., et al. (2018) Assessment of the vulnerability of sea turtles to IOTC tuna fisheries. *Iotc-2018-Wpeb14-40*.

Williams, B.R., Burgess, M.G., Ashe, E., Gaines, S.D. and Randall, R. (2016) Marine mammal protections require increased global capacity. *Science* **354**, 1372–1374.

Wilson, S.M., Raby, G.D., Burnett, N.J., Hinch, S.G. and Cooke, S.J. (2014) Looking beyond the mortality of bycatch: Sublethal effects of incidental capture on marine animals. *Biological Conservation* **171**, 61–72.

Winter, S.T., Fahmi, Rudianto, D., Laglbauer, B.J.L., Ender, I. and Simpfendorfer, C.A. (2020) Immature individuals dominate elasmobranch fisheries of the Bali Strait. *Marine and Freshwater Research*, 1488–1500.

Wood, S. (2015) Package 'mgcv.' 29.

Woodhams, J., Ingram, B., Lyle, J., Braccini, M., Rogers, P.J. and Peddemors, V. (2020) School Shark (2018). Available at: http://www.fish.gov.au/report/182-School-Shark-2018 [Accessed January 28, 2021].

Worm, B., Barbier, E., Beaumont, N., et al. (2006) Impacts of biodiversity loss on ocean ecosystem services. *Science (New York, N.Y.)* **314**.

Worm, B., Hilborn, R., Baum, J.K., et al. (2009) Rebuilding global fisheries. *Science* **325**, 578–85.

WoRMS Editorial Board (2019) World Register of Marine Species. Available at: http://www.marinespecies.org [Accessed October 14, 2019].

WWF (2020) Unregulated fishing on the High Seas of the Indian Ocean: The impacts on, risks to, and challenges for sustainable fishing and ocean health.

WWF (2019) WWF-Australia and OpenSc. Available at: https://www.wwf.org.au/get-involved/panda-labs/opensc#gs.hi7rtb [Accessed November 19, 2019].

Ye, Y. and Gutierrez, N.L. (2017) Ending fishery overexploitation by expanding from local successes to globalized solutions. *Nature Ecology & Evolution* **1**, 0179.

Young, M.A. (2016) International trade law compatibility of market-related measures to combat illegal, unreported and unregulated (IUU) fishing. *Marine Policy* **69**, 209–219.

Young, O.R., Webster, D.G., Cox, M.E., et al. (2018) Moving beyond panaceas in fisheries governance. *Proceedings of the National Academy of Sciences* **115**, 9065–9073.

Yousuf, K.S.S.M., Anoop, A.K., Anoop, B., et al. (2009) Observations on incidental catch of cetaceans in three landing centres along the Indian coast. *Marine Biodiversity Records* **2**, 1–5.

Yulianto, I., Booth, H., Ningtias, P., et al. (2018) Practical measures for sustainable shark fisheries: Lessons learned from an Indonesian targeted shark fishery. *PLoS ONE* **13**, 1–18.

Zaman, K., Rangavajhala, S., McDonald, M.P. and Mahadevan, S. (2011) A probabilistic approach for representation of interval uncertainty. *Reliability Engineering and System Safety* **96**, 117–130.

Zeller, D., Cashion, T., Palomares, M. and Pauly, D. (2017) Global marine fisheries discards: A synthesis of reconstructed data. *Fish and Fisheries*, 1–10.

Zeller, D. and Pauly, D. (2019) Viewpoint: Back to the future for fisheries, where will we choose to go? *Global Sustainability* **2**.

Zhou, S., Campbell, R.A., Hoyle, S.D. and Anderson, E. (2019) Catch per unit effort standardization using spatio-temporal models for Australia's Eastern Tuna and Billfish Fishery. *ICES Journal of Marine Science* **76**, 1489–1504.

Zhou, S., Daley, R., Fuller, M., et al. (2013) ERA extension to assess cumulative effects of fishing on species.

Zhou, S., Hobday, A.J., Dichmont, C.M. and Smith, A.D.M. (2016) Ecological risk assessments for the effects of fishing: A comparison and validation of PSA and SAFE. *Fisheries Research* **183**, 518–529.

Zhou, S., Smith, A.D.M. and Fuller, M. (2011) Quantitative ecological risk assessment for fishing effects on diverse data-poor non-target species in a multi-sector and multi-gear fishery. *Fisheries Research* **112**, 168–178.

Zollett, E.A. and Swimmer, Y. (2019) Safe handling practices to increase post-capture survival of cetaceans, sea turtles, seabirds, sharks, and billfish in tuna fisheries. *Endangered Species Research* **38**, 115–125.

Appendix 1: Supplementary Materials for Chapter 2

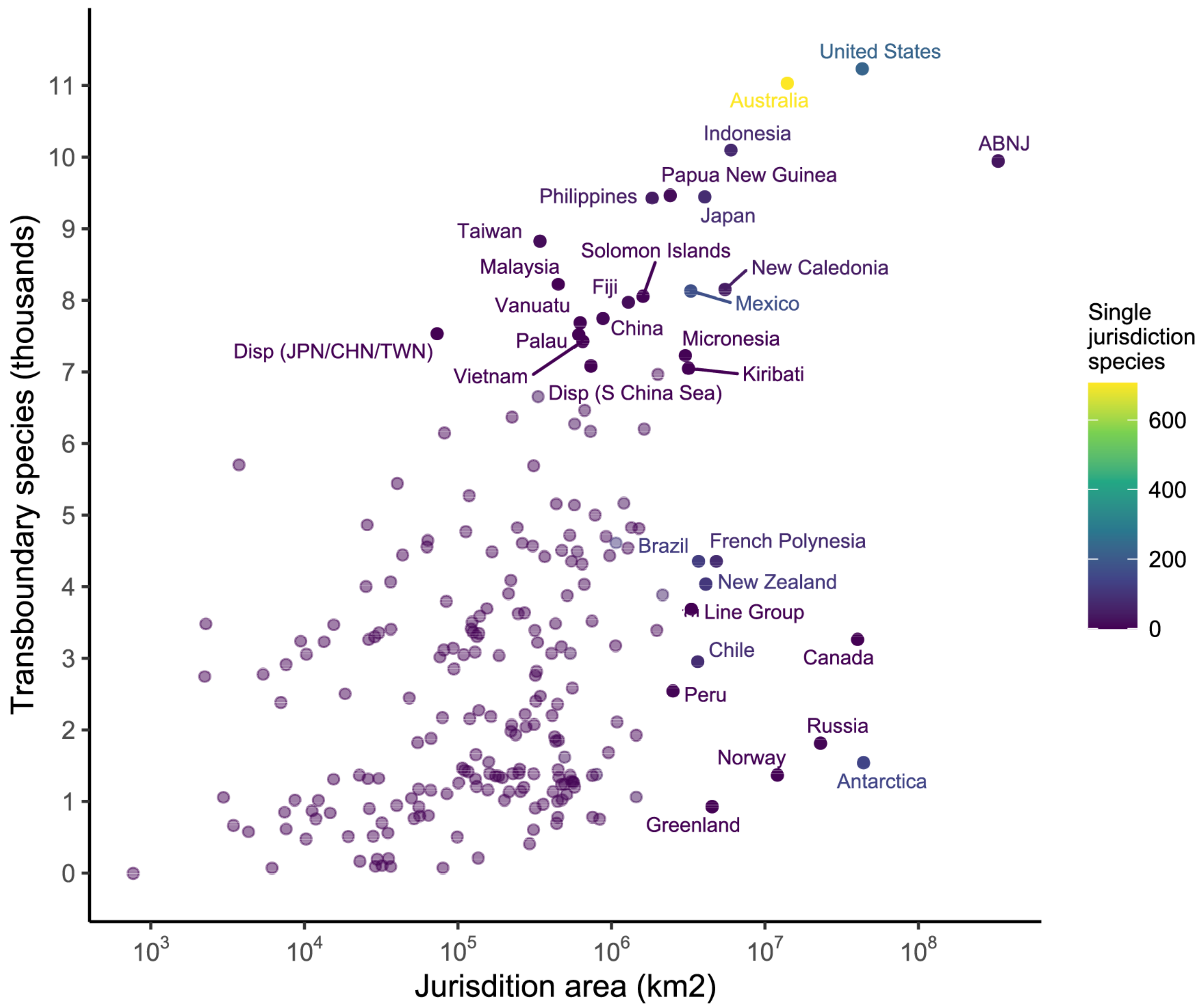

Figure S1.1: Transboundary species per area. Number of transboundary species compared to area of jurisdiction (km^2), shown on a log10 transformed scale. Labels show jurisdictions ranking in the top 20 for number of transboundary species or for area of jurisdiction. Disp = Disputed territory

Table S1.1: Taxonomic groupings for plant and animal species included in the analysis, and percent of species listed in the Ocean Biogeographic Information System (OBIS) database that have range maps in the IUCN or AquaMaps databases. Four kingdoms were excluded from the analysis. Groupings are not consistent across taxonomic levels (e.g., Mammals are a class of animals, whereas the group Lophophores contains multiple phyla)

Group	Subgroup 1	Subgroup 2	OBIS records	Records per Group	Maps per Group	Mapped (%)
Algae	Plants	Chlorophyta	872	3970	180	4.5
		Rhodophyta	3098			
Arthropods	Invertebrates	Arthropoda	32419	32419	3553	11.0
Cnidarians & Ctenophores	Invertebrates	Cnidarians	8324	8390	1532	18.3
		Ctenophores	66			
Echinoderms	Invertebrates	Echinodermata	5710	5710	1256	22.0
Fish (ray & lobe-finned)	Chordates	Actinopterygii	14530	14530	12848	88.4
Jawless fish & lancelets	Chordates	Agnatha	76	163	93	57.1
		Cephalochordata	22			
		Myxini	65			
Lophophores	Invertebrates	Brachiopoda	229	3186	224	7.0
		Bryzoa	2907			
		Entoprocta	39			
		Phoronida	11			
Mammals	Chordates	Mammalia	135	135	131	97.0
Mollusks	Invertebrates	Mollusca	26165	26165	5003	19.1
Reptiles	Chordates	Reptilia	89	89	74	83.1
*Seabirds	Chordates	Aves	600	600	359	59.8
*Sharks, Rays, Chimaeras	Chordates	Chondrichthyans	1053	1096	1199	109.4
		Holocephali	43			
Sponges	Invertebrates	Porifera	7688	7688	440	5.7
Tunicates	Chordates	Tunicata	1843	1843	665	36.1
Vascular plants	Plants	Mangroves	*68	138	136	98.6
		Seagrasses	*70			
Worms & microscopic animals	Invertebrates	Acanthocephala	110	17425	560	3.2
		Annelida	9100			
		Chaetognatha	69			
		Echiura	117			
		Gastrotricha	191			
		Gnathostomulida	39			
		Hemichordata	57			
		Mesozoa	52			
		Myxozoa	48			
		Nematoda	3292			
		Nemertea	362			
		Placozoa	2			
		Platyhelminthes	3186			

	Rotifera	183			
	Tardigrada	202			
	Xenacoelomorpha	415			
Excluded	Bacteria	624	624	5	0.8
Excluded	Chromista	10784	10784	24	0.2
Excluded	Fungi	231	231	0	0.0
Excluded	Protozoa	267	267	18	6.7

*The low proportion of Seabird maps compared to species listed in OBIS is due to different designations of species as marine. We use an expert-reviewed list of seabirds from BirdLife International, which uses a stricter definition of a "seabird" compared to OBIS
*The proportion of mapped Sharks, Rays, Chimaeras is greater than 100% due to recent changes in taxonomies

Table S1.2: Species conservation status and taxonomic information. The top 100 species are shown, ranked by number of jurisdictions (Jur.) they occur in. Red List categories (Cat.) are CR = Critically Endangered, EN = Endangered, VU = Vulnerable, NT = Near Threatened, LC = Least Concern, DD = Data Deficient, None = Not assessed.

Rank	Species name	Jur.	Cat.	Species group	Class
1	*Orcinus orca*	220	DD	Mammals	Mammalia
2	*Balaenoptera acutorostrata*	211	LC	Mammals	Mammalia
3	*Tursiops truncatus*	211	LC	Mammals	Mammalia
4	*Physeter macrocephalus*	210	VU	Mammals	Mammalia
5	*Alopias vulpinus*	205	VU	Sharks, Rays, Chimaeras	Chondrichthyes
6	*Ziphius cavirostris*	204	LC	Mammals	Mammalia
7	*Eretmochelys imbricata*	203	CR	Reptiles	Reptilia
8	*Grampus griseus*	202	LC	Mammals	Mammalia
9	*Megaptera novaeangliae*	201	LC	Mammals	Mammalia
10	*Xiphias gladius*	201	LC	Fish (ray & lobe-finned)	Actinopterygii
11	*Pseudorca crassidens*	200	NT	Mammals	Mammalia
12	*Microlophichthys microlophus*	200	LC	Fish (ray & lobe-finned)	Actinopterygii
13	*Pyroteuthis margaritifera*	199	None	Mollusks	Cephalopoda
14	*Argyropelecus hemigymnus*	198	LC	Fish (ray & lobe-finned)	Actinopterygii
15	*Carcharodon carcharias*	197	VU	Sharks, Rays, Chimaeras	Chondrichthyes
16	*Pteroplatytrygon violacea*	197	LC	Sharks, Rays, Chimaeras	Chondrichthyes
17	*Remora remora*	197	LC	Fish (ray & lobe-finned)	Actinopterygii
18	*Balaenoptera musculus*	196	EN	Mammals	Mammalia
19	*Prionace glauca*	196	NT	Sharks, Rays, Chimaeras	Chondrichthyes
20	*Isurus oxyrinchus*	195	EN	Sharks, Rays, Chimaeras	Chondrichthyes
21	*Katsuwonus pelamis*	195	LC	Fish (ray & lobe-finned)	Actinopterygii
22	*Istiophorus platypterus*	195	LC	Fish (ray & lobe-finned)	Actinopterygii
23	*Stenella coeruleoalba*	194	LC	Mammals	Mammalia
24	*Steno bredanensis*	192	LC	Mammals	Mammalia
25	*Cyclothone braueri*	192	LC	Fish (ray & lobe-finned)	Actinopterygii
26	*Chtenopteryx sicula*	190	None	Mollusks	Cephalopoda
27	*Haliphron atlanticus*	189	None	Mollusks	Cephalopoda
28	*Walvisteuthis virilis*	189	None	Mollusks	Cephalopoda
29	*Lucifer typus*	187	None	Arthropods	Malacostraca
30	*Ulva lactuca*	187	None	Algae	Ulvophyceae
31	*Vitreledonella richardi*	187	None	Mollusks	Cephalopoda
32	*Anoplogaster cornuta*	187	LC	Fish (ray & lobe-finned)	Actinopterygii
33	*Cyclothone pseudopallida*	187	LC	Fish (ray & lobe-finned)	Actinopterygii

34	*Onychoteuthis banksii*	186	None	Mollusks	Cephalopoda
35	*Chauliodus sloani*	186	LC	Fish (ray & lobe-finned)	Actinopterygii
36	*Cranchia scabra*	185	None	Mollusks	Cephalopoda
37	*Lagocephalus lagocephalus*	185	LC	Fish (ray & lobe-finned)	Actinopterygii
38	*Melanocetus johnsonii*	185	LC	Fish (ray & lobe-finned)	Actinopterygii
39	*Sigmops elongatus*	184	None	Fish (ray & lobe-finned)	Actinopterygii
40	*Cryptopsaras couesii*	184	LC	Fish (ray & lobe-finned)	Actinopterygii
41	*Balaenoptera borealis*	183	EN	Mammals	Mammalia
42	*Ulva clathrata*	182	None	Algae	Ulvophyceae
43	*Octopoteuthis sicula*	182	None	Mollusks	Cephalopoda
44	*Kogia breviceps*	182	DD	Mammals	Mammalia
45	*Carcharhinus longimanus*	182	VU	Sharks, Rays, Chimaeras	Chondrichthyes
46	*Cyclothone pallida*	182	LC	Fish (ray & lobe-finned)	Actinopterygii
47	*Chaenophryne ramifera*	182	LC	Fish (ray & lobe-finned)	Actinopterygii
48	*Vampyroteuthis infernalis*	181	None	Mollusks	Cephalopoda
49	*Sternoptyx diaphana*	180	None	Fish (ray & lobe-finned)	Actinopterygii
50	*Scopeloberyx opisthopterus*	179	None	Fish (ray & lobe-finned)	Actinopterygii
51	*Melamphaes polylepis*	179	None	Fish (ray & lobe-finned)	Actinopterygii
52	*Bolitaena pygmaea*	179	None	Mollusks	Cephalopoda
53	*Kogia sima*	179	DD	Mammals	Mammalia
54	*Chaenophryne longiceps*	179	LC	Fish (ray & lobe-finned)	Actinopterygii
55	*Ceratias holboelli*	179	LC	Fish (ray & lobe-finned)	Actinopterygii
56	*Liguriella podophthalma*	178	None	Mollusks	Cephalopoda
57	*Cunina octonaria*	178	None	Cnidarians & Ctenophores	Hydrozoa
58	*Coryphaena hippurus*	178	LC	Fish (ray & lobe-finned)	Actinopterygii
59	*Alopias superciliosus*	178	VU	Sharks, Rays, Chimaeras	Chondrichthyes
60	*Notolychnus valdiviae*	178	LC	Fish (ray & lobe-finned)	Actinopterygii
61	*Melanostomias niger*	177	None	Fish (ray & lobe-finned)	Actinopterygii
62	*Phyllodoce madeirensis*	177	None	Worms & microscopic animals	Polychaeta
63	*Mesoplodon densirostris*	177	DD	Mammals	Mammalia
64	*Cyclothone acclinidens*	177	LC	Fish (ray & lobe-finned)	Actinopterygii
65	*Valenciennellus tripunctulatus*	176	None	Fish (ray & lobe-finned)	Actinopterygii
66	*Gennadas scutatus*	176	None	Arthropods	Malacostraca
67	*Thysanoteuthis rhombus*	176	None	Mollusks	Cephalopoda
68	*Liocranchia reinhardti*	176	None	Mollusks	Cephalopoda
69	*Polycheles typhlops*	176	LC	Arthropods	Malacostraca
70	*Eurypharynx pelecanoides*	176	LC	Fish (ray & lobe-finned)	Actinopterygii
71	*Eustomias dendriticus*	175	None	Fish (ray & lobe-finned)	Actinopterygii
72	*Echeneis naucrates*	175	LC	Fish (ray & lobe-finned)	Actinopterygii
73	*Bentheogennema intermedia*	174	None	Arthropods	Malacostraca

74	*Gelidium pusillum*	174	None	Algae	Florideophyceae
75	*Didemnum candidum*	174	None	Tunicates	Ascidiacea
76	*Ommastrephes bartramii*	174	None	Mollusks	Cephalopoda
77	*Glycera tesselata*	174	None	Worms & microscopic animals	Polychaeta
78	*Cyclothone alba*	174	LC	Fish (ray & lobe-finned)	Actinopterygii
79	*Lobianchia gemellarii*	174	LC	Fish (ray & lobe-finned)	Actinopterygii
80	*Nemichthys scolopaceus*	173	None	Fish (ray & lobe-finned)	Actinopterygii
81	*Systellaspis debilis*	173	None	Arthropods	Malacostraca
82	*Japetella diaphana*	173	None	Mollusks	Cephalopoda
83	*Remora osteochir*	173	LC	Fish (ray & lobe-finned)	Actinopterygii
84	*Sergia japonica*	172	None	Arthropods	Malacostraca
85	*Sandalops melancholicus*	172	None	Mollusks	Cephalopoda
86	*Lysidice collaris*	172	None	Worms & microscopic animals	Polychaeta
87	*Globicephala macrorhynchus*	172	LC	Mammals	Mammalia
88	*Euprotomicrus bispinatus*	172	LC	Sharks, Rays, Chimaeras	Chondrichthyes
89	*Taaningichthys bathyphilus*	172	LC	Fish (ray & lobe-finned)	Actinopterygii
90	*Scopelarchus analis*	172	LC	Fish (ray & lobe-finned)	Actinopterygii
91	*Ranzania laevis*	171	None	Fish (ray & lobe-finned)	Actinopterygii
92	*Gnathophausia zoea*	171	None	Arthropods	Malacostraca
93	*Stenella attenuata*	171	LC	Mammals	Mammalia
94	*Mobula birostris*	171	VU	Sharks, Rays, Chimaeras	Chondrichthyes
95	*Malacosteus niger*	170	None	Fish (ray & lobe-finned)	Actinopterygii
96	*Bathothauma lyromma*	170	None	Mollusks	Cephalopoda
97	*Pterygioteuthis giardi*	170	None	Mollusks	Cephalopoda
98	*Manta birostris*	169	None	Sharks, Rays, Chimaeras	Elasmobranchii
99	*Systellaspis pellucida*	169	None	Arthropods	Malacostraca
100	*Balaenoptera brydei*	169	None	Mammals	Mammalia

Table S1.3: Species totals for 228 jurisdictions ranked by number of transboundary (TB) species. TB Thr = Threatened (Critically Endangered, Endangered, Vulnerable) species, One Jur. = one (single) jurisdiction species, TB/area = rank for number of transboundary species per km2. Composite World Governance Indicator score is scaled 0-1 (1 = best governance score)

Jurisdiction	Number of species			Rank			WGI score
	TB	TB Thr	One Jur	TB	TB/area	Area	
United States	11234	141	231	1	222	3	0.75
Australia	11033	222	706	2	220	6	0.82
Indonesia	10099	305	75	3	204	8	0.47
ABNJ	9946	125	31	4	228	1	NA
Papua New Guinea	9469	237	17	5	166	21	0.38
Japan	9450	207	82	6	188	13	0.77
Philippines	9431	276	45	7	151	25	0.43
Taiwan	8827	193	17	8	60	94	0.72
Malaysia	8226	274	1	9	75	77	0.59
New Caledonia	8154	142	45	10	207	9	0.72
Mexico	8133	107	174	11	185	17	0.43
Solomon Isls	8058	189	5	12	154	27	0.46
Fiji	7974	135	18	13	140	32	0.54
China	7750	106	11	14	115	41	0.44
Vanuatu	7689	123	4	15	97	54	0.52
Disp (JPN/CHN/TWN)	7538	151	0	16	24	167	NA
Palau	7524	116	1	17	96	55	0.56
Vietnam	7429	173	1	18	101	52	0.43
Micronesia	7229	153	3	19	186	19	0.57
Disp (S China Sea)	7084	120	1	20	108	48	NA
Kiribati	7051	114	0	21	192	18	0.57
Marshall Isls	6969	110	4	22	170	23	0.47
Panama	6655	80	28	23	69	95	0.52
Tonga	6465	78	10	24	107	50	0.55
Nicaragua	6369	68	0	25	50	120	0.32
Costa Rica	6274	82	17	26	102	57	0.62
India	6204	144	12	27	167	26	0.48
Colombia	6172	89	20	28	119	49	0.46
Disp (AUS/IND/TLS)	6148	123	0	29	31	162	NA

Disp (AUS/PNG)	5702	145	0	30	2	223	NA
Nauru	5689	31	0	31	74	103	0.49
East Timor	5444	227	0	32	16	181	0.41
Guatemala	5276	66	0	33	41	150	0.38
Madagascar	5169	122	20	34	163	34	0.35
Howland Isl & Baker Isl	5160	33	0	35	100	84	0.75
Mozambique	5144	130	8	36	113	58	0.34
Somalia	5003	102	1	37	139	44	0.08
Brunei	4866	230	0	38	13	199	0.62
Seychelles	4827	105	5	39	168	31	0.57
Tanzania	4826	94	2	40	70	117	0.39
Tuvalu	4815	115	0	41	175	28	0.56
Kenya	4771	95	2	42	43	152	0.39
Sri Lanka	4720	133	2	43	114	67	0.47
Maldives	4701	84	5	44	152	40	0.40
Mayotte	4648	93	0	45	32	171	0.72
South Africa	4613	98	111	46	162	36	0.53
Wallis & Futuna	4609	72	1	47	77	112	0.72
Thailand	4571	226	4	48	83	106	0.44
Juan de Nova Isl	4558	89	0	49	33	172	0.72
Mauritius	4542	110	8	50	169	33	0.65
Venezuela	4509	68	9	51	109	73	0.15
Bahamas	4491	69	13	52	127	56	0.62
Comoro Isls	4488	95	1	53	56	131	NA
Glorioso Isls	4443	87	0	54	25	180	0.72
Northern Mariana Isls & Guam	4435	90	7	55	157	38	0.75
Cuba	4421	62	2	56	99	91	0.41
Yemen	4357	120	2	57	123	64	0.10
Brazil	4356	83	123	58	211	14	0.45
French Polynesia	4355	77	81	59	217	10	0.72
British Indian Ocean Territory	4319	102	3	60	136	53	0.77
Honduras	4091	56	1	61	72	123	0.37
Belize	4069	57	9	62	22	185	0.44
New Zealand	4039	64	111	63	215	12	0.86
Andaman & Nicobar	4035	102	4	64	142	51	0.48
Aruba	4005	60	1	65	14	200	0.74

Puerto Rico & Virgin Isls (USA)	3906	60	2	66	73	125	0.75
Ecuador	3889	58	65	67	199	22	0.42
Myanmar	3880	136	4	68	126	68	0.31
Disp (JPN/KOR)	3798	49	0	69	40	161	NA
Turks & Caicos Isls	3699	57	0	70	64	136	0.77
Line Group	3690	58	0	71	212	16	0.57
Dominican Republic	3640	55	0	72	88	110	0.45
Jamaica	3627	52	0	73	85	116	0.55
Guadeloupe & Martinique	3593	56	3	74	59	137	0.72
Phoenix Group	3522	71	1	75	155	47	0.57
Haiti	3510	54	4	76	51	147	0.27
Norfolk Isl	3488	52	4	77	122	86	0.82
Oecussi Ambeno	3485	28	0	78	1	226	0.41
Saint Lucia	3473	50	0	79	11	205	0.61
Bassas da India	3414	26	0	80	54	148	0.72
Saint Vincent & the Grenadines	3410	51	0	81	26	183	0.61
Cook Isls	3395	65	3	82	203	24	0.52
Reunion	3394	94	4	83	103	101	0.69
Ile Europa	3369	28	0	84	57	146	0.72
Curacao	3359	32	2	85	23	190	0.66
Guyana	3351	40	0	86	62	138	0.46
Samoa	3307	92	0	87	61	141	0.63
Dominica	3303	53	0	88	20	194	0.60
Canada	3266	60	0	89	225	4	0.82
Grenada	3265	54	0	90	18	197	0.56
Saba	3243	52	0	91	7	214	0.66
Bonaire	3235	31	0	92	10	208	0.66
Christmas Isl	3225	28	1	93	106	96	0.82
Palmyra Atoll	3179	41	0	94	176	37	0.75
Cocos Isls	3166	49	1	95	135	75	0.82
Anguilla	3145	56	0	96	47	159	0.67
British Virgin Isls	3120	59	0	97	45	163	0.77
Suriname	3089	42	0	98	63	145	0.46
Oman	3072	73	21	99	145	65	0.53
American Samoa	3071	93	0	100	125	90	0.68
Saint Kitts & Nevis	3059	50	0	101	9	212	0.62

Antigua & Barbuda	3052	54	0	102	53	154	0.58
Barbados	3041	50	2	103	81	128	0.66
Trinidad & Tobago	3021	60	3	104	44	166	0.52
Chile	2955	58	86	105	219	15	0.70
Montserrat	2917	50	0	106	5	217	0.77
El Salvador	2855	33	0	107	49	158	0.43
South Korea	2820	55	1	108	116	97	0.68
Northern Saint-Martin	2780	51	0	109	4	221	0.72
Jarvis Isl	2763	33	0	110	118	99	0.75
Sint Eustatius	2749	27	0	111	3	227	0.66
Spain	2587	63	3	112	156	62	0.66
Peru	2546	51	6	113	214	20	0.47
Disp (COL/JAM)	2508	22	0	114	15	204	NA
France	2475	59	1	115	133	93	0.72
Cambodia	2450	120	0	116	37	179	0.35
Tokelau	2406	30	0	117	128	98	0.86
Djibouti	2386	74	0	118	8	219	0.34
Johnston Atoll	2362	34	0	119	149	81	NA
French Guiana	2277	41	0	120	79	139	0.72
Ile Tromelin	2221	21	0	121	121	109	0.72
Wake Isl	2206	33	0	122	147	89	0.75
Iran	2192	59	0	123	89	132	0.30
Eritrea	2176	100	1	124	55	165	0.18
Cayman Isls	2161	50	3	125	76	149	0.67
Argentina	2113	57	11	126	197	35	0.50
Portugal	2078	56	0	127	137	102	0.71
Pakistan	2072	69	2	128	110	121	0.31
Morocco	2046	62	0	129	131	108	0.44
Saudi Arabia	1983	115	3	130	112	122	0.45
Egypt	1934	123	5	131	120	118	0.34
United Kingdom	1929	44	0	132	209	29	0.77
Ireland	1906	42	1	133	160	87	0.78
Sudan	1884	94	0	134	52	168	0.18
Bermuda	1856	30	8	135	165	78	0.72
Clipperton Isl	1844	34	8	136	164	85	NA
United Arab Emirates	1824	53	0	137	48	176	0.63

Russia	1815	52	7	138	226	5	0.37
Azores	1689	39	1	139	200	39	0.71
Uruguay	1658	61	0	140	92	143	0.67
Angola	1622	57	24	141	173	71	0.32
Senegal	1556	66	9	142	105	134	0.48
Antarctica	1541	29	145	143	227	2	NA
Guinea Bissau	1470	56	0	144	87	155	0.27
Western Sahara	1454	56	0	145	144	114	NA
Canary Isls	1447	56	1	146	174	79	0.66
Guinea	1438	54	1	147	91	153	0.31
North Korea	1419	39	0	148	98	151	0.18
Liberia	1398	54	1	149	146	115	0.35
Sierra Leone	1391	52	1	150	117	119	0.51
Ghana	1391	59	0	150	141	133	0.38
Equatorial Guinea	1389	58	0	151	159	104	0.24
Cape Verde	1381	43	45	152	202	43	0.60
Norway	1372	32	2	153	224	7	0.85
Gambia	1370	61	0	154	35	202	0.42
Italy	1367	57	2	155	183	66	0.60
Iceland	1365	23	2	156	198	46	0.81
Ivory Coast	1359	56	0	157	124	130	0.40
Nigeria	1358	55	2	158	130	129	0.29
Madeira	1344	54	3	159	177	76	0.71
Gabon	1335	53	1	160	134	127	0.35
Benin	1327	52	0	161	42	191	0.44
Algeria	1322	47	0	162	104	144	0.34
Disp (EGY/SDN)	1322	94	0	162	38	198	NA
Togo	1315	52	0	163	27	206	0.34
Falkland Isls	1287	26	2	164	187	63	0.77
Namibia	1278	54	1	165	189	61	0.56
Kerguelen Isls	1270	18	21	166	191	60	0.72
Tunisia	1263	50	0	167	94	156	0.46
Greece	1244	45	2	168	184	70	0.56
Prince Edward Isls	1242	19	0	169	181	74	0.53
Sao Tome & Principe	1213	40	3	170	111	142	0.46
Crozet Isls	1206	20	2	171	195	59	0.72

Faeroe Isls	1201	24	0	172	158	111	0.77
Croatia	1176	44	1	173	67	175	0.59
Mauritania	1165	56	0	174	129	135	0.35
Disp (JPN/KOR 2)	1160	17	0	175	78	169	NA
Turkey	1143	43	0	176	161	113	0.40
Heard & McDonald Isls	1139	16	0	177	179	88	0.82
Disp (JPN/RUS)	1138	31	0	178	148	124	NA
Bangladesh	1111	56	0	179	90	160	0.34
Amsterdam Isl & Saint Paul Isl	1104	25	6	180	193	69	0.72
South Georgia & South Sandwich Isls	1066	17	26	181	221	30	0.77
Jersey	1061	28	0	182	6	225	0.75
Disp (ESH/MAR)	1049	47	0	183	68	178	NA
Macquarie Isl	1029	21	3	184	194	72	0.82
Guernsey	1023	29	0	185	19	215	0.75
Denmark	1022	23	0	186	153	126	0.84
Saint Pierre & Miquelon	1020	28	0	187	28	209	0.72
Ascension	1008	27	9	188	190	82	NA
Libya	965	44	0	189	180	92	0.12
Republique du Congo	948	52	1	190	65	182	0.17
Greenland	931	16	0	191	223	11	0.78
Malta	926	38	0	192	80	174	0.71
Niue	910	32	2	193	178	100	0.42
Democratic Republic of the Congo	904	49	0	194	46	196	0.17
Albania	873	41	0	195	30	211	0.50
Montenegro	855	41	0	196	21	218	0.53
Cameroon	845	53	0	197	36	207	0.29
Netherlands	808	22	0	198	93	170	0.84
Germany	800	21	0	199	86	173	0.80
Tristan da Cunha	786	28	1	200	213	45	0.77
Saint Helena	785	25	7	201	201	80	0.77
Disp (KEN/SOM)	766	28	0	202	84	177	NA
Kuwait	761	44	0	203	34	210	0.48
Pitcairn	758	29	1	204	218	42	0.77
Qatar	704	44	0	205	66	188	0.57
Bouvet Isl	697	12	0	206	206	83	0.85
Belgium	668	22	0	207	12	224	0.74

Bahrain	621	44	0	208	29	216	0.45
Sweden	610	23	0	209	196	105	0.84
Disp (TTO/VEN/GUY)	580	20	0	210	17	222	NA
Disp (NGA/STP)	564	29	0	211	82	187	NA
Israel	519	34	1	212	71	195	0.63
Lebanon	515	29	0	213	58	203	0.33
Cyprus	509	32	0	214	150	157	0.66
Syria	480	28	0	215	39	213	0.11
Jan Mayen	412	11	0	216	208	107	0.85
Ukraine	214	13	1	217	205	140	0.36
Bulgaria	205	13	0	218	143	186	0.55
Romania	197	13	0	219	138	192	0.53
Georgia	167	10	0	220	132	201	0.59
Poland	109	7	0	221	171	189	0.63
Latvia	97	7	0	222	172	193	0.66
Estonia	95	7	0	223	182	184	0.74
Finland	77	5	0	224	216	164	0.85
Lithuania	76	6	0	225	95	220	0.68
Singapore	1	0	0	226	210	228	0.83

Appendix 2: Supplementary Materials for Chapter 3

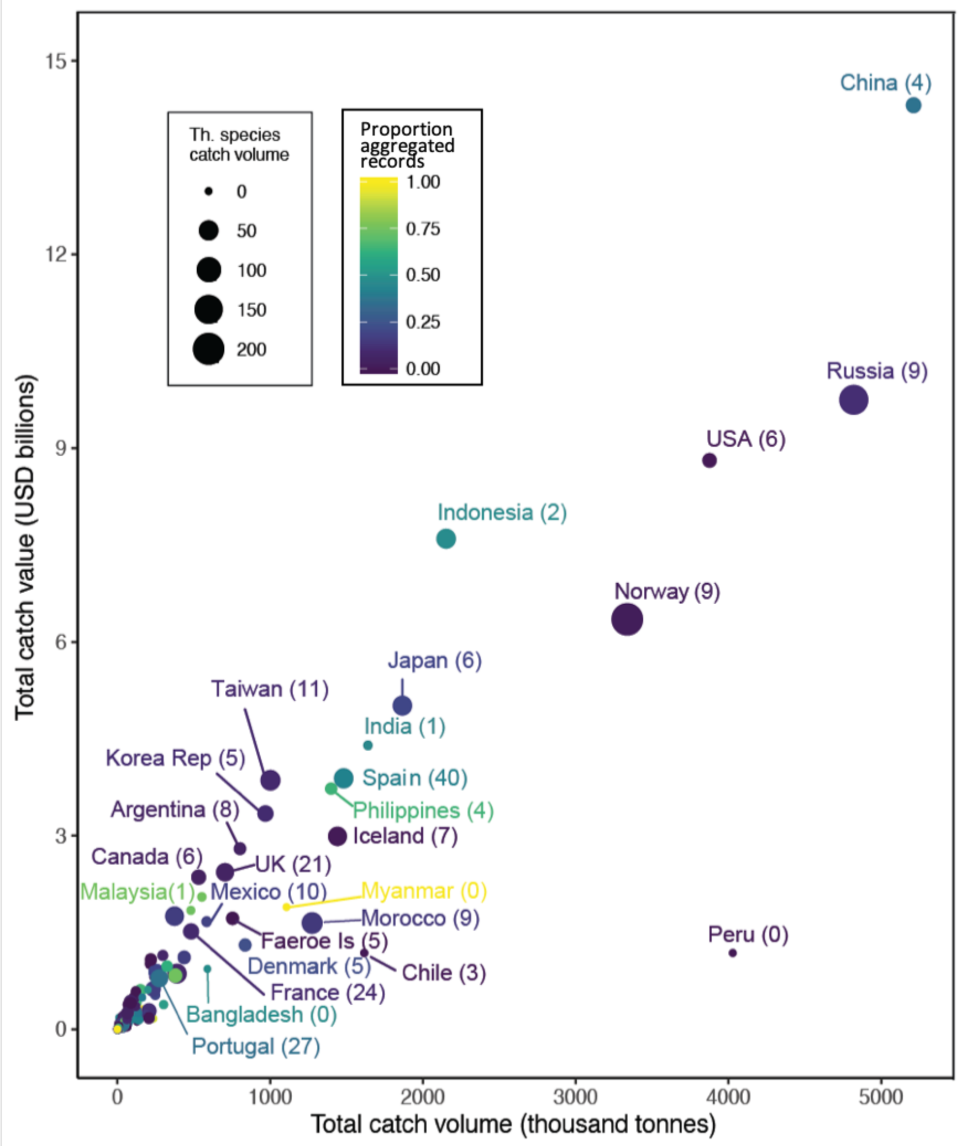

Figure S2.1: Catch volume and estimated value for 181 fishing countries in the global catch database described in Watson & Tidd (2018). Bubble size corresponds to volume of threatened species catch. Number of threatened species each country catches is in parentheses. Colour shows the ratio of volume of aggregated records to volume of species-level records (i.e., yellow indicates catch volumes mostly recorded in aggregated and purple indicates catch volumes mostly recorded to the species-level). Volumes and values are 5-year weighted moving averages for 2010

Table S2.1: Red List assessment and fishing information for the threatened species appearing in global catch data. Chond = chondrichthyan, Invert = invertebrate. Cat = Category, CR = Critically Endangered, EN = Endangered, VU = Vulnerable). Threats were coded as 1 = Targeted industrial fishing, 2 = Incidental industrial fishing, 3 = Targeted non-industrial fishing, 4 = Incidental non-industrial fishing, 5 = Unspecified fishing, 6 = Other. Price is mean ex-vessel price over the time period (2006 - 2014). Species in bold are listed in the RAM Stock Legacy Database. Species highlighted in grey were last assessed before 2010. *Gadus morhua* was excluded from the final analysis of threatened species

Species	Taxon group	Red List Assessments				Appendices		Price (USD/kg)	Countries (num.)	
		Cat.	Year	Pop. trend	Threats	CITES	CMS		Fishers	Importers
Acipenser gueldenstaedtii	Teleost	CR	2009	Decreasing	1,2,3,4,6	II	II	1.1	1	--
Acipenser stellatus	Teleost	CR	2009	Decreasing	1,3,6	II	II	1.1	1	--
Acipenser sturio	Teleost	CR	2009	Decreasing	2,4,6	I	I/II	1.4	1	--
Alopias superciliosus	Chond.	VU	2007	Decreasing	1,2,3,4,6	II	II	0.4	19	--
Alopias vulpinus	Chond.	VU	2007	Decreasing	1,2,3,4,6	II	II	0.8	20	--
Alosa aestivalis	Teleost	VU	2011	Decreasing	6	--	--	1.2	1	--
Alosa immaculata	Teleost	VU	2008	Decreasing	1,3,6	--	--	0.9	4	--
Anguilla anguilla	Teleost	CR	2013	Decreasing	1,3,5,6	II	II	9.5	18	--
Anguilla rostrata	Teleost	EN	2013	Decreasing	1,6	--	--	7.7	2	--
Apostichopus japonicus	Invert.	EN	2010	Decreasing	1,3	--	--	2.1	3	88
Atlantoraja cyclophora	Chond.	VU	2006	Decreasing	2,3	--	--	2.3	1	--
Balistes capriscus	**Teleost**	**VU**	**2011**	**Decreasing**	**1,3**	**--**	**--**	**1.7**	**8**	**--**
Bolbometopon muricatum	Teleost	VU	2007	Decreasing	3,6	--	--	4.2	1	--
Carcharhinus albimarginatus	Chond.	VU	2015	Decreasing	1,2,3,4	--	--	0.5	1	--
Carcharhinus dussumieri	Chond.	EN	2018	Decreasing	2,3,4	--	--	1.5	2	--
Carcharhinus falciformis	Chond.	VU	2017	Decreasing	2,4	II	II	0.8	31	--
Carcharhinus longimanus	Chond.	VU	2006	Decreasing	1,2,3,4	II	--	0.8	31	--

Carcharhinus obscurus	**Chond.**	**VU**	**2007**	**Decreasing**	**1,2,3,4,6**	**--**	**II**	**1.5**	**10**	**--**
Carcharhinus plumbeus	**Chond.**	**VU**	**2007**	**Decreasing**	**1,2,3,4,6**	**--**	**--**	**0.9**	**7**	**--**
Carcharias taurus	Chond.	VU	2005	Unknown	1,2,4,6	--	--	4.9	6	--
Carcharodon carcharias	Chond.	VU	2005	Unknown	2,3,4,5,6	II	I/II	2.1	11	--
Centrophorus lusitanicus	Chond.	VU	2008	Unknown	2,3,4	--	--	3.3	2	--
Centrophorus squamosus	**Chond.**	**VU**	**2003**	**Decreasing**	**2,4**	**--**	**--**	**1.5**	**11**	**--**
Cetorhinus maximus	**Chond.**	**VU**	**2005**	**Decreasing**	**2,4,5,6**	**II**	**I/II**	**2.1**	**10**	**--**
Coryphaenoides rupestris	**Teleost**	**CR**	**2012**	**Unknown**	**1**	**--**	**--**	**1.4**	**14**	**--**
Cymatoceps nasutus	Teleost	VU	2009	Decreasing	1,3	--	--	4.3	1	--
Dalatias licha	**Chond.**	**VU**	**2017**	**Decreasing**	**1,2,3,4**	**--**	**--**	**1**	**11**	**--**
Dentex dentex	Teleost	VU	2009	Unknown	1,3,6	--	--	17.1	12	--
Dipturus batis	Chond.	CR	2006	Decreasing	2,4	--	--	1.7	6	--
Epinephelus itajara	**Teleost**	**VU**	**2016**	**Decreasing**	**1,3,4,6**	**--**	**--**	**11.9**	**3**	**--**
Epinephelus marginatus	Teleost	VU	2016	Decreasing	1,3	--	--	11.4	13	--
Epinephelus morio	**Teleost**	**VU**	**2016**	**Decreasing**	**1,3,6**	**--**	**--**	**4.4**	**3**	**--**
Epinephelus striatus	Teleost	CR	2016	Decreasing	1,3,6	--	--	8.4	2	11
****Gadus morhua***	**Teleost**	***VU**	**1996**	**Unknown**	**--**	**--**	**--**	**2.9**	**24**	**179**
Galeorhinus galeus	**Chond.**	**VU**	**2006**	**Decreasing**	**1,2,3,4**	**--**	**--**	**1.6**	**20**	**--**
Gymnura altavela	Chond.	VU	2007	Decreasing	2,3,4	--	--	2.3	2	--
Hippoglossus hippoglossus	**Teleost**	**EN**	**1996**	**Unknown**	**--**	**--**	**--**	**9**	**20**	**153**
Hyporthodus flavolimbatus	Teleost	VU	2016	Decreasing	1,2,3	--	--	5.8	3	--
Hyporthodus niveatus	Teleost	VU	2016	Decreasing	1,3	--	--	6.1	2	--
Isurus oxyrinchus	**Chond.**	**EN**	**2018**	**Decreasing**	**1,2,3,4,5,6**	**II**	**II**	**2.9**	**45**	**--**
Isurus paucus	Chond.	EN	2018	Decreasing	1,2,3,4	II	II	1.1	10	--
Kajikia albida	Teleost	VU	2010	Decreasing	1,2,3,4	--	--	2.9	22	19
Lamna nasus	**Chond.**	**VU**	**2006**	**Decreasing**	**1,2,3,4**	**II**	**II**	**3.4**	**33**	**--**

Lethrinus mahsena	Teleost	EN	2018	Decreasing	1	--	--	4.4	3	--
Leucoraja circularis	**Chond.**	**EN**	**2014**	**Decreasing**	**2,4**	**--**	**--**	**2.4**	**5**	**--**
Leucoraja fullonica	**Chond.**	**VU**	**2014**	**Decreasing**	**2,4**	**--**	**--**	**2.1**	**5**	**--**
Limulus polyphemus	Invert.	VU	2016	Decreasing	1,3,6	--	--	1.3	1	--
Lopholatilus chamaeleonticeps	**Teleost**	**EN**	**2013**	**Decreasing**	**1,6**	**--**	**--**	**4.9**	**1**	**--**
Lutjanus campechanus	**Teleost**	**VU**	**2015**	**Decreasing**	**1,3**	**--**	**--**	**4.8**	**2**	**--**
Makaira nigricans	**Teleost**	**VU**	**2010**	**Decreasing**	**1,2,3,4**	**--**	**--**	**1.9**	**30**	**51**
Megalops atlanticus	Teleost	VU	2018	Decreasing	2,3,4,6	--	--	0.8	4	--
Melanogrammus aeglefinus	**Teleost**	**VU**	**1996**	**Unknown**	**--**	**--**	**--**	**1.7**	**18**	**181**
Merluccius senegalensis	Teleost	EN	2012	Decreasing	1,3,6	--	--	2.2	6	70
Mobula mobular	Chond.	EN	2014	Decreasing	1,2,3,4	II	I/II	0.7	1	--
Mola mola	Teleost	VU	2011	Decreasing	1,2	--	--	1.9	12	--
Mustelus mustelus	Chond.	VU	2004	Decreasing	2,4	--	--	2.3	16	--
Mustelus schmitti	Chond.	EN	2006	Decreasing	1,2,4	--	--	2.6	2	--
Mycteroperca interstitialis	Teleost	VU	2016	Decreasing	1,3	--	--	3	1	--
Mycteroperca microlepis	**Teleost**	**VU**	**2016**	**Decreasing**	**1,3**	**--**	**--**	**8.2**	**2**	**--**
Nebrius ferrugineus	Chond.	VU	2003	Decreasing	2,4	--	**--**	0.6	1	--
Nemipterus virgatus	Teleost	VU	2009	Decreasing	1	--	--	2.1	3	--
Oxynotus centrina	Chond.	VU	2007	Unknown	2	--	--	1.5	2	--
Palinurus elephas	Invert.	VU	2013	Decreasing	1	--	--	15.5	9	--
Pentanemus quinquarius	Teleost	VU	2014	Decreasing	2,3,4	--	--	10.5	7	--
Plectropomus areolatus	Teleost	VU	2016	Decreasing	1,3,6	--	--	3.9	1	--
Plectropomus pessuliferus	Teleost	VU	2016	Decreasing	1,3	--	--	7.5	1	--
Pomatomus saltatrix	**Teleost**	**VU**	**2014**	**Decreasing**	**1,2,3,4**	**--**	**--**	**3.6**	**23**	**--**
Pseudotolithus senegalensis	Teleost	EN	2009	Decreasing	1,3,6	--	--	1.4	9	--
Pseudotolithus senegallus	Teleost	VU	2014	Decreasing	1,3,6	--	--	1.1	2	--

Pseudupeneus prayensis	Teleost	VU	2013	Decreasing	1,2,3	--	--	1.9	13	--
Raja undulata	**Chond.**	**EN**	**2003**	**Decreasing**	**2,4**	**--**	**--**	**2.7**	**4**	**--**
Rhincodon typus	Chond.	EN	2016	Decreasing	1,2,3,4,6	II	I/II	0.6	1	--
Rhomboplites aurorubens	**Teleost**	**VU**	**2015**	**Decreasing**	**1,3,6**	**--**	**--**	**3.9**	**4**	**--**
Rhynchobatus djiddensis	Chond.	CR	2018	Decreasing	1,2,3,4,6	II	--	0.8	2	--
Rostroraja alba	Chond.	EN	2006	Decreasing	2	--	--	2.5	2	--
Sardinella maderensis	Teleost	VU	2014	Unknown	1,3,6	--	--	0.6	13	--
Sciades parkeri	Teleost	VU	2011	Decreasing	1,3	--	--	1.3	1	--
Sebastolobus alascanus	**Teleost**	**EN**	**2000**	**Unknown**	**--**	**--**	**--**	**2.4**	**2**	**--**
Sphyrna lewini	**Chond.**	**EN**	**2007**	**Unknown**	**1,2,3,4,5,6**	**II**	**II**	**0.7**	**18**	**--**
Sphyrna mokarran	Chond.	EN	2007	Decreasing	1,2,3,4,6	II	II	0.5	5	--
Sphyrna zygaena	Chond.	VU	2005	Decreasing	1,2,3,4,6	II	--	0.7	20	--
Squalus acanthias	**Chond.**	**VU**	**2016**	**Decreasing**	**1,2,3,4**	**--**	**II**	**1.6**	**36**	**173**
Squatina argentina	Chond.	CR	2017	Decreasing	2,4	--	--	1.3	1	--
Squatina squatina	**Chond.**	**CR**	**2017**	**Decreasing**	**1,2,3,4,6**	**--**	**I/II**	**1.4**	**7**	**--**
Tautoga onitis	Teleost	VU	2008	Decreasing	3,6	--	--	5.8	1	--
Thunnus maccoyii	**Teleost**	**CR**	**2009**	**Decreasing**	**1**	**--**	**--**	**5.8**	**11**	**57**
Thunnus obesus	**Teleost**	**VU**	**2011**	**Decreasing**	**1,2**	**--**	**--**	**3**	**76**	**193**
Thunnus orientalis	**Teleost**	**VU**	**2014**	**Decreasing**	**1**	**--**	**--**	**7.3**	**23**	**5**
Thunnus thynnus	**Teleost**	**EN**	**2014**	**Decreasing**	**1**	**--**	**--**	**8**	**32**	**127**
Trachurus mediterraneus	**Teleost**	**VU**	**2017**	**Decreasing**	**1,3**	**--**	**--**	**2.4**	**13**	**--**
Trachurus trachurus	**Teleost**	**VU**	**2013**	**Decreasing**	**1,3**	**--**	**--**	**1**	**29**	**133**
Zearaja chilensis	**Chond.**	**VU**	**2007**	**Decreasing**	**1,2,3,4**	**--**	**--**	**2**	**8**	**--**

Table S2.2: IUCN threat codes listed for the 92 threatened species found in the catch data. Codes are categorized into 6 groups ("Code"). Spp = number of species with that threat listed. Threat codes numbered >100 and described as "OLD" are for species last assessed before the updated IUCN threat codes

Code	Description	Spp.	IUCN threat code and description
1	Targeted industrial fishing	65	5.4.2 Intentional use: (large scale) [harvest]
1	Targeted industrial fishing	65	101.4 OLD 3.1.3 Harvesting (hunting/gathering)->Food->Regional/international trade
1	Targeted industrial fishing	65	101.16 OLD 3.4.3 Harvesting (hunting/gathering)->Materials->Regional/international trade
1	Targeted industrial fishing	65	5.3.2 Intentional use: (large scale) [harvest]
2	Incidental industrial fishing	50	5.4.4 Unintentional effects: (large scale) [harvest]
3	Targeted non-industrial fishing	61	5.4.1 Intentional use: (subsistence/small scale) [harvest]
3	Targeted non-industrial fishing	61	101.2 OLD 3.1.1 Harvesting (hunting/gathering)->Food->Subsistence use/local trade
3	Targeted non-industrial fishing	61	101.3 OLD 3.1.2 Harvesting (hunting/gathering)->Food->Sub-national/national trade
4	Incidental non-industrial fishing	44	5.4.3 Unintentional effects: (subsistence/small scale) [harvest]
5	Unspecified fishing	5	101.17 OLD 3.5 Harvesting (hunting/gathering)->Cultural/scientific/leisure activities
5	Unspecified fishing	5	101.1 OLD 3.1 Harvesting (hunting/gathering)->Food
5	Unspecified fishing	5	5.1.1 Intentional use (species is the target)
6	Other	36	5.4.5 Persecution/control
6	Other	36	9.3.4 Type Unknown/Unrecorded
6	Other	36	6.1 Recreational activities
6	Other	36	9.1.1 Sewage
6	Other	36	9.1.3 Type Unknown/Unrecorded
6	Other	36	8.1.2 Named species
6	Other	36	101.35 OLD 9.5 Intrinsic factors->Low densities
6	Other	36	101.32 OLD 9.2 Intrinsic factors->Poor recruitment/reproduction/regeneration
6	Other	36	3.1 Oil & gas drilling
6	Other	36	4.3 Shipping lanes
6	Other	36	7.2.10 Large dams
6	Other	36	8.2 Problematic native species/diseases
6	Other	36	7.2.9 Small dams
6	Other	36	9.2.3 Type Unknown/Unrecorded
6	Other	36	100.18 OLD 4.1.1 Accidental mortality->Bycatch->Fisheries related

6	Other	36	101.37 OLD 9.7 Intrinsic factors->Slow growth rates
6	Other	36	101.13 OLD 3.4 Harvesting (hunting/gathering)->Materials
6	Other	36	1.1 Housing & urban areas
6	Other	36	7.2.11 Dams (size unknown)
6	Other	36	8.1.1 Unspecified species
6	Other	36	12.1 Other threat
6	Other	36	3.2 Mining & quarrying
6	Other	36	9.2.1 Oil spills
6	Other	36	11.1 Habitat shifting & alteration
6	Other	36	11.3 Temperature extremes
6	Other	36	9.1.2 Run-off
6	Other	36	6.3 Work & other activities
6	Other	36	5.4.6 Motivation Unknown/Unrecorded
6	Other	36	101.40 OLD 9.10 Intrinsic factors->Other
6	Other	36	101.39 OLD 9.9 Intrinsic factors->Restricted range
6	Other	36	9.4 Garbage & solid waste
6	Other	36	9.6.3 Noise pollution
6	Other	36	7.3 Other ecosystem modifications
6	Other	36	1.3 Tourism & recreation areas
6	Other	36	1.2 Commercial & industrial areas
6	Other	36	2.4.3 Scale Unknown/Unrecorded
6	Other	36	11.5 Other impacts
6	Other	36	9.3.3 Herbicides and pesticides
6	Other	36	9.2.2 Seepage from mining
6	Other	36	11.4 Storms & flooding
6	Other	36	2.2.2 Agro-industry plantations
6	Other	36	2.3.3 Agro-industry grazing, ranching or farming
6	Other	36	2.1.3 Agro-industry farming
6	Other	36	9.6.2 Thermal pollution
6	Other	36	3.3 Renewable energy
6	Other	36	9.5.1 Acid rain
6	Other	36	100.29 OLD 4.2.3 Accidental mortality->Collision->Other

6	Other	36	100.55 OLD 12 Unknown
6	Other	36	4.1 Roads & railroads
6	Other	36	7.2.1 Abstraction of surface water (domestic use)
6	Other	36	7.2.5 Abstraction of ground water (domestic use)
6	Other	36	7.2.6 Abstraction of ground water (commercial use)
6	Other	36	7.2.7 Abstraction of ground water (agricultural use)
6	Other	36	11.2 Droughts
6	Other	36	7.2.2 Abstraction of surface water (commercial use)
6	Other	36	7.2.3 Abstraction of surface water (agricultural use)
6	Other	36	9.3.2 Soil erosion, sedimentation
6	Other	36	9.3.1 Nutrient loads

Table S2.3: Volume and value of country catch. Catch ranks and proportions are based off the 8-year weighted moving average for 2014. The 50 countries catching the largest volumes of threatened (Th.) species between 2006 - 2014 are shown. Ranks are for all 163 fishing countries. Aggregated = not species-level commodity record.

Fishing country	Th. species volume		Th. species value		Commodities		Aggregated records (%)
	Rank	% of total	Rank	% of total	Th.	All	
Norway	1	5.9	1	8.6	8	37	6.1
Russia	2	2.9	3	3.9	12	225	11.2
Netherlands	3	18.4	11	7.9	11	157	6.6
Morocco	4	5.2	18	3.6	12	86	15.5
Ireland	5	25.9	5	23.5	12	162	12.3
Iceland	6	4.5	4	6.0	8	68	20.9
Belize	7	14.0	8	34.9	5	70	67.0
Mauritania	8	26.2	31	13.1	5	75	50.6
UK	9	5.9	9	4.8	27	275	9.1
Unknown	10	4.8	10	4.8	29	115	5.3
Spain	11	4.3	7	5.4	43	410	33.3
Turkey	12	5.5	16	9.0	8	85	3.7
Japan	13	0.8	2	2.1	21	267	42.7
Philippines	14	1.7	15	3.1	3	100	53.6
Portugal	15	13.0	12	17.2	39	308	38.1
France	16	4.7	14	4.0	32	333	18.8
USA	17	0.4	6	1.9	33	271	7.4
Canada	18	1.6	13	1.8	11	99	11.9
Tunisia	19	17.6	34	7.9	5	61	44.2
Senegal	20	6.4	27	3.5	16	113	41.0
Taiwan	21	2.7	19	4.1	15	123	11.2
Ukraine	22	6.2	24	15.0	19	227	52.6
India	23	0.5	39	0.3	13	95	69.9
Denmark	24	1.3	21	1.8	8	97	33.1
China	25	0.1	23	0.1	13	195	51.1
Latvia	26	5.9	30	12.0	5	63	42.3
South Korea	27	0.7	20	1.1	24	339	25.1
Mexico	28	0.7	22	3.6	8	53	2.9
Ghana	29	10.0	32	13.3	7	83	11.7
Faeroe Islands	30	1.5	25	2.0	6	59	44.0
Lithuania	31	5.4	35	7.5	5	86	34.6
Indonesia	32	0.2	29	0.2	2	68	57.8

Papua New Guinea	33	3.0	38	4.5	3	17	0.2
Nigeria	34	4.4	41	1.8	1	27	73.4
Brazil	35	1.5	26	2.7	31	225	21.4
Ecuador	36	1.7	33	4.5	8	46	13.9
New Zealand	37	2.4	37	2.0	15	191	27.1
Marshall Islands	38	6.3	42	10.0	3	13	0.1
Australia	39	2.6	28	2.6	15	246	69.5
Kiribati	40	4.5	48	7.1	2	11	0.0
Sri Lanka	41	2.8	56	1.2	7	52	81.1
Italy	42	1.3	17	4.3	11	137	28.7
Micronesia	43	6.9	43	10.7	3	12	0.5
Germany	44	0.9	51	1.1	8	110	18.3
Greenland	45	0.8	45	0.7	3	31	14.2
Costa Rica	46	11.5	61	6.1	4	25	58.7
Congo Republic	47	17.2	46	21.6	2	38	34.6
Seychelles	48	8.9	60	11.5	1	14	3.5
Libya	49	34.9	40	48.3	6	34	35.9
Namibia	50	0.2	50	0.6	7	62	32.8

Table S2.4: Volume and value of country imports. Import ranks and proportions are based off the 8-year weighted moving average for 2014. The 50 countries catching the largest volumes of threatened (Th.) species between 2006 - 2014 are shown. Ranks are for all 204 importing countries. Aggregated = not species-level commodity record.

Importing country	Th. species volume		Th. species value		Commodities		Aggregated records (%)
	Rank	% of total	Rank	% of total	Th.	All	
UK	1	8.7	2	6.5	10	338	28.1
Germany	2	5.3	4	4.6	9	327	25.2
Nigeria	3	4.7	6	5.1	11	308	45.8
Belgium	4	9.8	9	5.5	8	292	33.2
Spain	5	2.2	1	3.3	11	368	49.4
Denmark	6	2.2	8	2.5	11	333	35.1
USA	7	1.5	3	2.1	12	347	50.3
Netherlands	8	1.9	13	1.7	10	341	35.0
China	9	1.1	7	1.5	11	365	51.2
New Zealand	10	13.8	11	10.7	8	256	61.9
Thailand	11	1.3	5	2.6	12	359	45.0
Sweden	12	1.7	15	2.0	10	347	33.6
France	13	1.6	12	1.9	12	351	39.5
Italy	14	2.0	10	2.5	10	353	41.2
Canada	15	2.6	17	2.9	7	271	29.1
Mauritius	16	4.8	14	7.4	10	297	37.8
Portugal	17	2.3	16	3.4	11	325	35.5
Taiwan	18	1.7	18	1.9	9	337	51.3
Turkey	19	7.2	24	6.9	9	264	22.8
Ukraine	20	2.1	21	3.3	7	228	19.8
Hong Kong	21	1.3	19	2.2	8	253	56.3
Cote d'Ivoire	22	1.7	22	3.7	7	262	30.3
Angola	23	5.1	31	5.5	10	236	49.2
Japan	24	0.3	20	0.5	9	247	74.9
Belarus	25	2.9	32	3.9	6	194	17.8
Poland	26	0.9	35	1.1	7	263	42.2
Peru	27	3.1	33	4.6	9	255	39.1
Namibia	28	1.1	23	1.6	11	300	47.6
Norway	29	0.5	37	0.9	9	293	52.6
Ecuador	30	0.8	26	1.8	10	252	24.8
South Africa	31	2.0	34	2.4	10	310	68.6
Australia	32	1.7	30	2.2	8	169	47.2
Fiji	33	2.7	28	4.5	7	181	24.4

Greece	34	2.3	27	3.7	11	304	40.3
Cameroon	35	2.0	39	3.1	8	222	35.1
Korea Rep	36	0.3	38	0.5	9	291	72.4
Falkland Is	37	3.7	29	6.0	10	171	14.0
Ghana	38	1.0	40	1.8	9	269	35.4
UAE	39	1.3	36	2.1	3	134	59.0
Egypt	40	0.7	25	2.4	9	259	47.2
Chile	41	1.1	42	1.5	7	209	40.0
Gabon	42	21.6	53	21.0	7	156	29.6
Russia	43	0.2	44	0.4	8	244	57.7
Seychelles	44	2.7	41	5.2	7	216	64.4
Mexico	45	1.1	43	1.7	10	260	52.8
Iceland	46	1.8	50	1.6	8	277	35.7
Brazil	47	0.8	54	1.2	8	266	37.7
Latvia	48	1.2	51	1.5	5	197	16.4
Benin	49	2.8	48	5.1	8	187	49.4
Switzerland	50	2.2	47	3.0	7	251	39.7

Table S2.5: Best model of countries' threatened species catch volumes (two-way ANOVA). All catch volumes are 2014 weighted moving averages (8-year window). GDP is 2014 per capita GDP (USD). CI = confidence interval, LL = lower limit, UL = upper limit, Sig = Significance

Predictor variable	Estimate	Std. Error	*t*	95% CI		Pr(>*t*)	Sig.
				LL	UL		
(Intercept)	904	1.72E+03	0.526	-2494	4302	0.5998	
Total catch volume	0.0100	0.0021	4.656	0.000574	0.0142	4.40E-07	***
Aggregated records volume	-0.0151	0.0047	-3.232	-0.0244	-0.00587	0.00153	**
GDP	0.2355	0.0737	3.197	0.0898	0.3811	0.00172	**
Residual standard error: 15330 on 139 degrees of freedom							
Adjusted R-squared: 0.2117							
F-statistic: 13.71, p = 6.919e-08							

Table S2.6: Best model of countries' threatened species import volumes (two-way ANOVA). All import volumes are 2015 weighted moving averages (9-year window). GDP is 2014 per capita GDP (USD). CI = confidence interval, LL = lower limit, UL = upper limit, Sig = Significance

Predictor variable	Estimate	Std. Error	*t*	95% CI		Pr(>*t*)	Sig.
				LL	UL		
(Intercept)	42.8409	2.40E+02	0.178	-430.6	516.3	0.859	
Total imports volume	0.0522	0.0035	14.921	0.0452	0.0590	<2e-16	***
Aggregated records volume	-0.0708	0.0069	-10.241	-0.0843	-0.0571	<2e-16	***
Residual standard error: 3175 on 206 degrees of freedom							
Adjusted R-squared: 0.6644							
F-statistic: 206.9, p < 2.2e-16							

Table S2.7**:** Names and Red List categories of 61 threatened species found in the global catch database described in Watson & Tidd (2018). CR = Critically Endangered, EN = Endangered, VU = Vulnerable

Species	**Common name**	**Taxon group**	**Red List Category**
Alopias superciliosus	Bigeye thresher	Chondrichthyan	VU
Alopias vulpinus	Thintail thresher	Chondrichthyan	VU
Alosa immaculata	Pontic shad	Teleost	VU
Apostichopus japonicus	Japanese sea cucumber	Invertebrate	EN
Argyrosomus hololepidotus	Southern meagre	Teleost	EN
Balistes capriscus	Grey triggerfish	Teleost	VU
Carcharhinus falciformis	Silky shark	Chondrichthyan	VU
Carcharhinus longimanus	Oceanic whitetip shark	Chondrichthyan	VU
Carcharhinus plumbeus	Sandbar shark	Chondrichthyan	VU
Carcharias taurus	Sand tiger shark	Chondrichthyan	VU
Carcharodon carcharias	Great white shark	Chondrichthyan	VU
Centrophorus lusitanicus	Lowfin gulper shark	Chondrichthyan	VU
Centrophorus squamosus	Leafscale gulper shark	Chondrichthyan	VU
Cetorhinus maximus	Basking shark	Chondrichthyan	VU
Dalatias licha	Kitefin shark	Chondrichthyan	VU
Dentex dentex	Common dentex	Teleost	VU
Dipturus batis	Blue skate	Chondrichthyan	CR
Epinephelus marginatus	Dusky grouper	Teleost	VU
Epinephelus striatus	Nassau grouper	Teleost	CR
Gadus morhua	Atlantic cod	Teleost	VU
Galeorhinus galeus	Tope shark	Chondrichthyan	VU
Gymnura altavela	Spiny butterfly ray	Chondrichthyan	VU
Hippoglossus hippoglossus	Atlantic halibut	Teleost	EN
Isurus oxyrinchus	Shortfin mako	Chondrichthyan	EN
Isurus paucus	Longfin mako	Chondrichthyan	EN
Kajikia albida/Tetrapturus albidus	Atlantic white marlin	Teleost	VU
Lamna nasus	Porbeagle	Chondrichthyan	VU
Lethrinus mahsena	Sky emperor	Teleost	EN
Leucoraja circularis	Sandy ray	Chondrichthyan	EN
Leucoraja fullonica	Shagreen ray	Chondrichthyan	VU
Lutjanus campechanus	Northern red snapper	Teleost	VU
Makaira nigricans	Atlantic blue marlin	Teleost	VU
Megalops atlanticus	Tarpon	Teleost	VU
Melanogrammus aeglefinus	Haddock	Teleost	VU
Merluccius senegalensis	Senegalese hake	Teleost	EN

Mobula mobular	Devil fish	Chondrichthyan	EN
Mola mola	Ocean sunfish	Teleost	VU
Mustelus mustelus	Smooth-hound	Chondrichthyan	VU
Mustelus schmitti	Narrownose smoothhound	Chondrichthyan	EN
Nemipterus virgatus	Golden threadfin bream	Teleost	VU
Oxynotus centrina	Angular roughshark	Chondrichthyan	VU
Palinurus elephas	Common spiny lobster	Invertebrate	VU
Pentanemus quinquarius	Royal threadfin	Teleost	VU
Pomatomus saltatrix	Bluefish	Teleost	VU
Pseudotolithus senegalensis	Cassava croaker	Teleost	EN
Pseudotolithus senegallus	Law croaker	Teleost	VU
Pseudupeneus prayensis	West African goatfish	Teleost	VU
Raja undulata	Undulate ray	Chondrichthyan	EN
Rhomboplites aurorubens	Vermilion snapper	Teleost	VU
Sardinella maderensis	Madeiran sardinella	Teleost	VU
Sebastolobus alascanus	Shortspine thornyhead	Teleost	EN
Sphyrna lewini	Scalloped hammerhead	Chondrichthyan	EN
Sphyrna zygaena	Smooth hammerhead	Chondrichthyan	VU
Squalus acanthias	Piked dogfish	Chondrichthyan	VU
Squatina argentina	Argentine angelshark	Chondrichthyan	CR
Squatina squatina	Angelshark	Chondrichthyan	CR
Thunnus maccoyii	Southern bluefin tuna	Teleost	CR
Thunnus obesus	Bigeye tuna	Teleost	VU
Thunnus orientalis	Pacific bluefin tuna	Teleost	VU
Thunnus thynnus	Atlantic bluefin tuna	Teleost	VU
Trachurus trachurus	Atlantic horse mackerel	Teleost	VU

Appendix 3: Supplementary Materials for Chapter 5

Table S3.1: Results for the 49 cetacean species comparing the catchability probability (rank and mean) from the rank-probability approach to the catch susceptibility score from the categorical scores approach. Mean catchability is across all cells where the species overlaps with fishing. Horizontal overlap= percent of species' range that overlaps with driftnet fishing. Depth overlap = percent of depth range that overlaps with driftnet fishing. Select = selectivity.

Species	Group	Catchability		Horz. overlap (%)	Depth overlap (%)	Select. rank	Lethality interval	Catch Susc. score
		Rank	Mean prob.					
Neophocaena phocaenoides	Shallow nearshore dolphins porpoises	1	8.29E-05	98.5	100	2	Lethal	3.00
Sousa chinensis	Shallow nearshore dolphins porpoises	2	6.83E-05	97.4	100	2	Lethal	3.00
Orcaella brevirostris	Shallow nearshore dolphins porpoises	3	5.24E-05	99.4	100	2	Lethal	3.00
Tursiops aduncus	Pelagic & semipelagic dolphins	4	4.37E-05	91.7	100	1	Potentially lethal	3.00
Tursiops truncatus	Pelagic & semipelagic dolphins	5	3.26E-05	67.0	25	1	Potentially lethal	2.71
Steno bredanensis	Pelagic & semipelagic dolphins	6	1.75E-05	72.0	50	1	Potentially lethal	3.00
Delphinus capensis	Pelagic & semipelagic dolphins	7	1.32E-05	91.9	50	1	Potentially lethal	3.00
Stenella attenuata	Pelagic & semipelagic dolphins	8	1.21E-05	71.8	50	1	Potentially lethal	3.00
Stenella longirostris	Pelagic & semipelagic dolphins	9	5.65E-06	72.9	20	1	Potentially lethal	2.71
Feresa attenuata	Large pelagic dolphins	10	2.53E-06	71.7	12.5	3	Potentially lethal	2.21
Stenella coeruleoalba	Pelagic & semipelagic dolphins	11	1.93E-06	69.0	7.1	1	Potentially lethal	2.28
Peponocephala electra	Large pelagic dolphins	12	1.73E-06	73.4	10	3	Potentially lethal	2.21
Delphinus delphis	Pelagic & semipelagic dolphins	13	1.46E-06	63.9	25	1	Potentially lethal	2.45
Orcinus orca	Large pelagic dolphins	14	1.37E-06	64.6	10	3	Potentially lethal	2.00
Grampus griseus	Large pelagic dolphins	15	1.07E-06	66.9	5	3	Potentially lethal	1.86
Globicephala macrorhynchus	Large pelagic dolphins	16	9.99E-07	70.4	6.2	3	Potentially lethal	1.86

Megaptera novaeangliae	Large whales	17	7.00E-07	64.5	25	5	Potentially lethal	1.68
Lagenodelphis hosei	Pelagic & semipelagic dolphins	18	6.32E-07	71.4	8.3	1	Potentially lethal	2.06
Mesoplodon ginkgodens	Beaked & small sperm whales	19	5.75E-07	72.8	2.5	4	Potentially lethal	1.86
Kogia breviceps	Beaked & small sperm whales	20	5.53E-07	68.9	12.5	4	Potentially lethal	2.00
Indopacetus pacificus	Beaked & small sperm whales	21	4.15E-07	72.7	3.3	4	Potentially lethal	1.86
Orcaella heinsohni	Shallow nearshore dolphins porpoises	22	3.51E-07	99.4	100	2	Lethal	3.00
Balaenoptera musculus	Large whales	23	3.11E-07	64.2	20	5	Sublethal	1.41
Pseudorca crassidens	Large pelagic dolphins	24	3.01E-07	68.9	2.5	3	Potentially lethal	1.86
Kogia sima	Beaked & small sperm whales	25	2.69E-07	71.7	2.5	4	Potentially lethal	1.86
Mesoplodon densirostris	Beaked & small sperm whales	26	2.02E-07	68.9	2.5	4	Potentially lethal	1.86
Balaenoptera physalus	Large whales	27	1.55E-07	64.5	20	5	Potentially lethal	1.68
Ziphius cavirostris	Beaked & small sperm whales	28	1.31E-07	65.7	1.7	4	Potentially lethal	1.86
Balaenoptera brydei	Large whales	29	1.27E-07	69.0	2.5	5	Potentially lethal	1.57
Balaenoptera edeni	Large whales	30	8.06E-08	68.0	2.5	5	Potentially lethal	1.57
Lissodelphis peronii	Pelagic & semipelagic dolphins	31	7.67E-08	56.6	25	1	Potentially lethal	2.45
Lagenorhynchus obscurus	Pelagic & semipelagic dolphins	32	3.80E-08	54.9	25	1	Potentially lethal	2.45
Lagenorhynchus cruciger	Pelagic & semipelagic dolphins	33	3.65E-08	40.9	25	1	Potentially lethal	2.45
Physeter macrocephalus	Large whales	34	3.35E-08	64.5	2	5	Potentially lethal	1.41
Balaenoptera acutorostrata	Large whales	35	1.47E-08	64.6	2.5	5	Potentially lethal	1.41
Globicephala melas	Large pelagic dolphins	36	1.28E-08	59.1	12.5	3	Potentially lethal	2.00
Caperea marginata	Large whales	37	1.15E-08	57.5	50	5	Potentially lethal	1.86
Balaenoptera borealis	Large whales	38	1.12E-08	61.5	16.7	5	Potentially lethal	1.68
Eubalaena australis	Large whales	39	9.73E-09	59.1	28.6	5	Potentially lethal	1.68
Balaenoptera bonaerensis	Large whales	40	6.46E-09	59.1	50	5	Potentially lethal	1.86
Berardius arnuxii	Beaked & small sperm whales	41	4.52E-09	59.1	5	4	Potentially lethal	1.68
Mesoplodon layardii	Beaked & small sperm whales	42	4.38E-09	59.1	5	4	Potentially lethal	1.68

Phocoena dioptrica	Shallow nearshore dolphins porpoises	43	2.81E-09	44.9	2.5	2	Lethal	2.06
Mesoplodon bowdoini	Beaked & small sperm whales	44	2.74E-09	59.1	2.5	4	Potentially lethal	1.68
Mesoplodon grayi	Beaked & small sperm whales	45	2.45E-09	59.1	2.5	4	Potentially lethal	1.68
Mesoplodon mirus	Beaked & small sperm whales	46	2.27E-09	65.7	2.5	4	Potentially lethal	1.86
Hyperoodon planifrons	Beaked & small sperm whales	47	2.11E-09	59.1	2.5	4	Potentially lethal	1.68
Mesoplodon hectori	Beaked & small sperm whales	48	1.40E-09	57.4	2	4	Potentially lethal	1.68
Tasmacetus shepherdi	Beaked & small sperm whales	49	3.78E-10	62.4	5	4	Potentially lethal	1.68

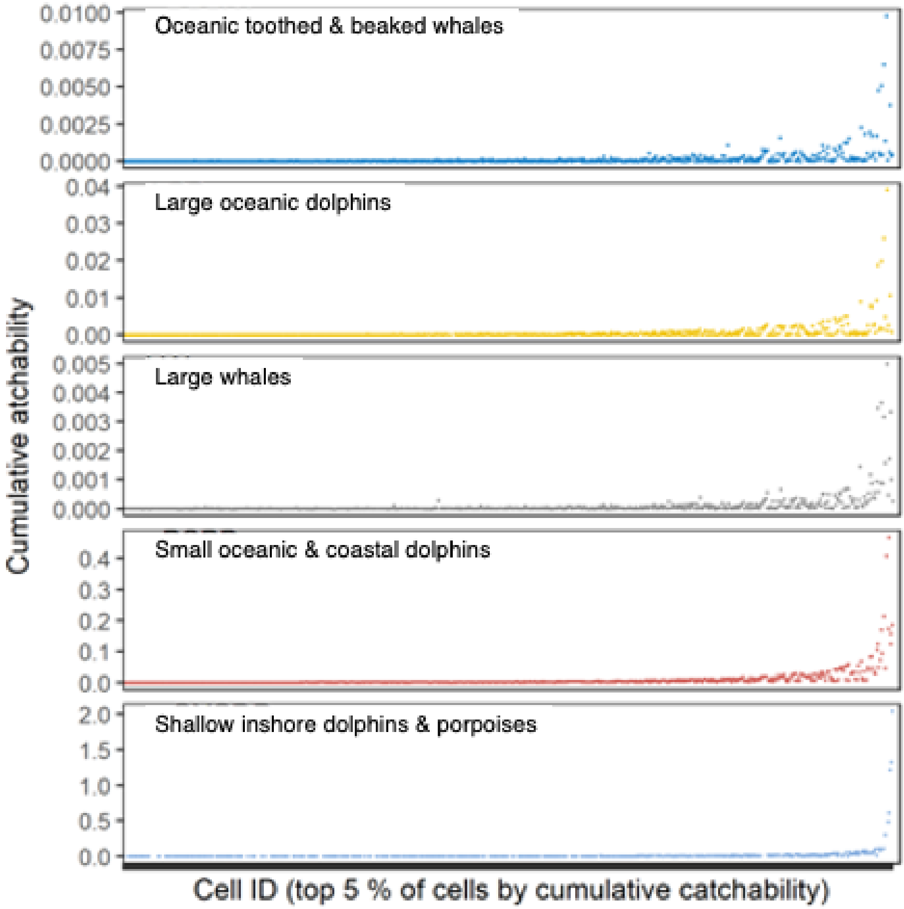

Figure S3.2: Expanded scales show a similar pattern for each species group, with the highest cumulative catchability scores occurring in a small cluster of the cells

Appendix 4: Supplementary Materials for Chapter 6

Appendix 4.1 Supplementary Information

Appendix 4.1.1 Supplementary Info 1: Fishing effort

The model of fishing effort uses data from FAO and country-specific reports to divide each country's effort into ten power classes based on gross tonnage, length overall, and engine power and associate effort with a corresponding catch (Rousseau *et al.* 2019; Rousseau 2020). The effort was mapped in 0.5 degree cells using a ratio to the total catch, and limiting the distance from the coast that boats of certain size classes could operate (e.g. limiting artisanal boats to the EEZ of the country and unmotorised boats to 12nm from the coast) (Rousseau 2020). Incompatibilities between effort and catch were resolved by comparing broader families of gears (e.g., lines instead of longlines, bottom nets instead of bottom trawls, etc.). For countries where there was no information on the link between tonnage, length, and engine power, characteristics are assumed to be similar to neighbouring countries. This approach fills missing data with information from neighbouring countries, which improves upon earlier approaches where missing data were replaced with global averages derived from the larger industrial fleets (Rousseau *et al.* 2019). This approach can generate errors for countries with missing information that are anomalous to their neighbours. We removed South Africa's large gillnet effort in the P4 and P5 power categories (50-200 kW). South Africa does not have a fleet targeting tuna and tuna-likes with gillnets in the IOTC area (Parker *et al.* 2018), and this error likely arises because of the characteristics of neighbouring countries that do have substantial gillnet effort in the low and medium power classes.

We also conducted a review of the peer-reviewed and grey literature, including IOTC reports for each country, to identify which countries have a gillnet sector targeting tuna or tuna likes in the Indian Ocean. For countries where there is no available information about whether their gillnets are small inshore bottom set nets versus larger drift nets, we errored on the conservative side and included effort from these countries in the final analysis. The model maps effort to particular grid cells. Where information on catch is missing, effort is attributed to grid cells based on the characteristics of that country's fleet, including assumptions about major ports and the distance that vessels in different power classes can travel from the coast.

Despite these assumptions, the lack of spatial information in the catch data (especially for gillnets) results in extremely skewed effort in a small number of cells typically clustered near ports along certain coasts. Assuming that effort from one fishing country and gear type will not vary dramatically between neighbouring cells, we first smoothed the predicted fishing effort across each country and gear type using a custom smoothing method in R based on functions in the GDAL library. Next, we made separate rasters for each country and fishing gear, then smoothed the fishing effort values by first summing each cell's value with its 8 neighbouring cells, then dividing the sum by the sea surface area within the 9 cells. The rasters from all countries were summed to obtain a global raster for each gear type. Next, we examined the spread of fishing effort and adjusted outlier values based on quantile thresholds for each gear type. For gillnets, we replaced values greater than the 90th percentile with a value one greater than that percentile (replacing all the very high values with one number). For purse seine and longline effort, which is less skewed, we replaced the values above the 95th percentile value. Finally, we log transformed and scaled the effort from 0 to 1 across all gear types, to get a relative probability that fishing occurs for each gear type in each cell. The resulting effort remains heavily skewed, but we assume the skewedness derives from real patterns in fishing effort. For example, smaller gillnet vessels are clustered near certain ports and population centres, and in some areas are known to concentrate near Fish Aggregating Devices.

Appendix 4.1.2 Supplementary Info 2: Species information

The AquaMaps model gives four depth limits (minimum, preferred minimum, maximum, and preferred maximum). For air-breathing species (sea turtles and marine mammals), we used the minimum depth (0m) and maximum preferred depth. For the majority of the air-breathing species, the maximum preferred depth predicted by AquaMaps extends beyond the deepest published dive records. For these 43 sea turtles and cetaceans we used information from IUCN, OBIS, and WoRMS to adjust the depth maxima. Where depth information was not available for a species (e.g., many beaked whales), we adjusted the maximum depth to the genus or family average. For elasmobranchs, we selected the minimum preferred depth and the maximum depth because overall, these limits corresponded best to information from published global databases (WoRMS Editorial Board 2019; Froese and Pauly 2019; OBIS 2020). Modelled depth limits aligned better with empirical data for elasmobranchs compared to air-breathing taxa, and we only adjusted depths for two requiem shark species (silky shark, *Carcharhinus falciformis* and Human's whaler shark, *C. humani,* Carcharhinidae).

Appendix 4.2 Supplementary Figures

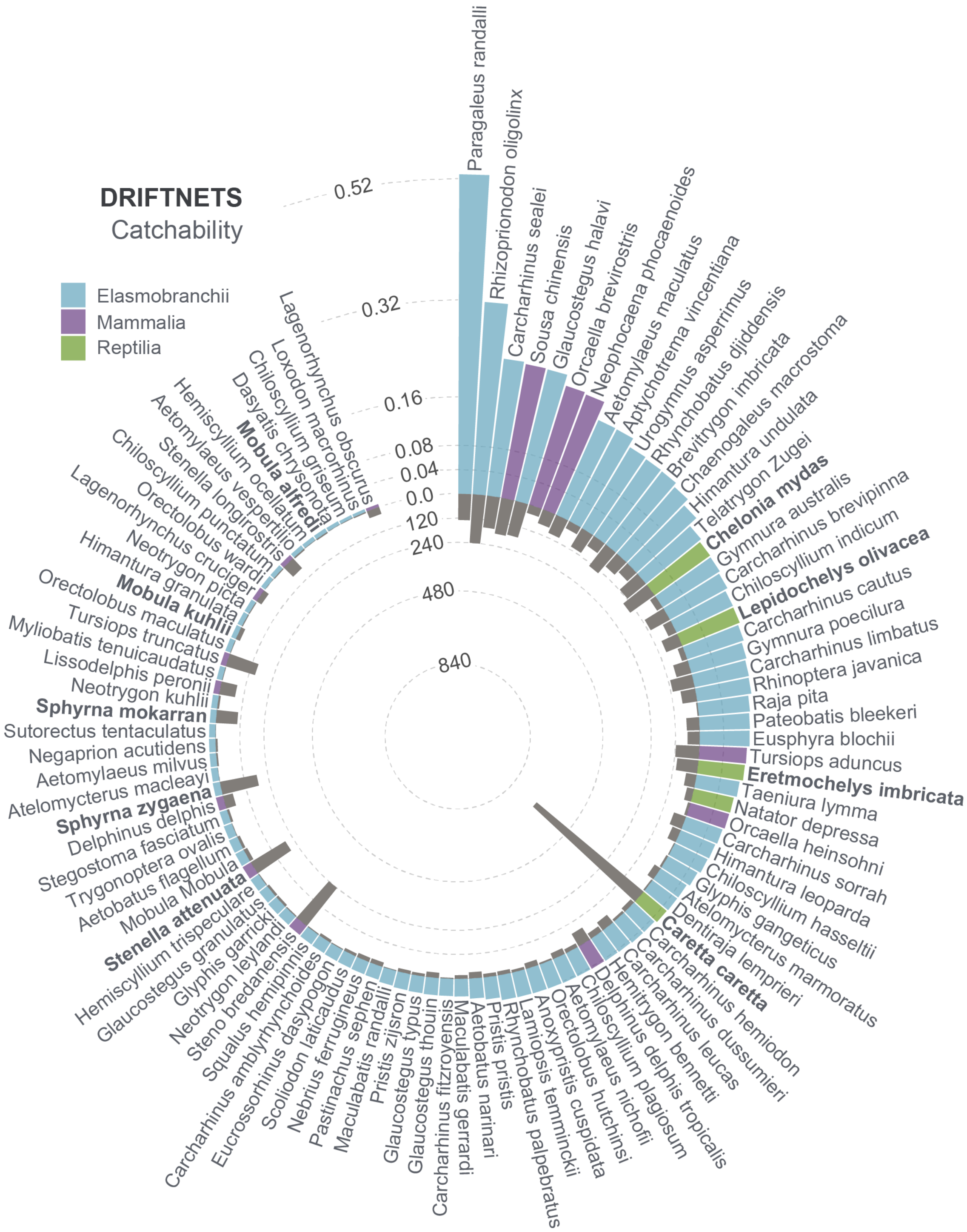

Figure S4.2.1: Mean (outer ring) and cumulative (inner ring) catchability scores for driftnets. Species are ordered clockwise by descending mean catchability score and the top 100 species are shown. Bars are colored by taxonomic group (elasmobranchs, cetaceans, and sea turtles). Species names are in bold if that species is listed in catch records for that gear type in the Indian Ocean (peer reviewed literature or IOTC reports).

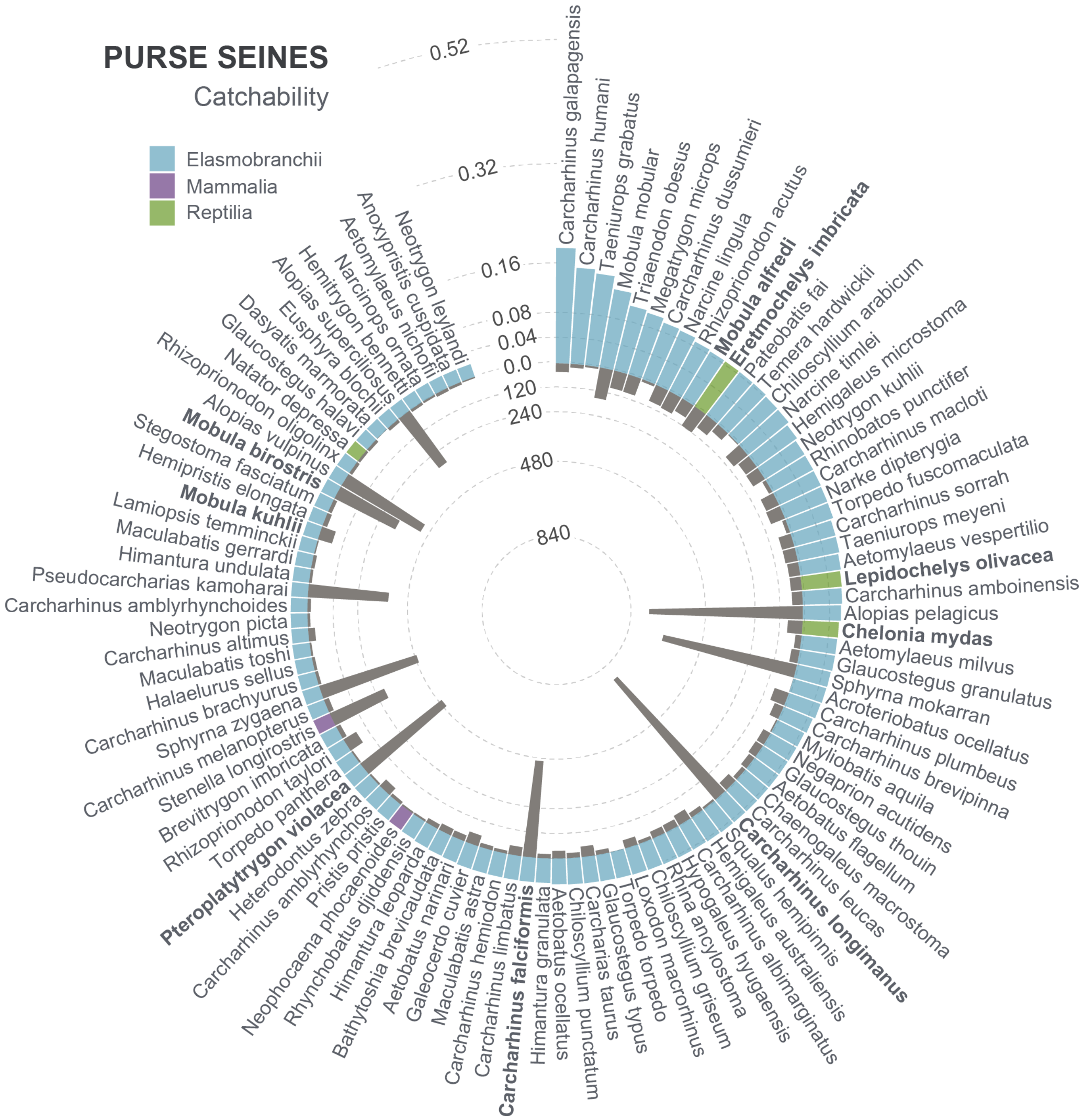

Figure S4.2.2: Mean (outer ring) and cumulative (inner ring) catchability scores for purse seines. Species are ordered clockwise by descending mean catchability score and the top 100 species are shown. Bars are colored by taxonomic group (elasmobranchs, cetaceans, and sea turtles). Species names are in bold if that species is listed in catch records for that gear type in the Indian Ocean (peer reviewed literature or IOTC reports).

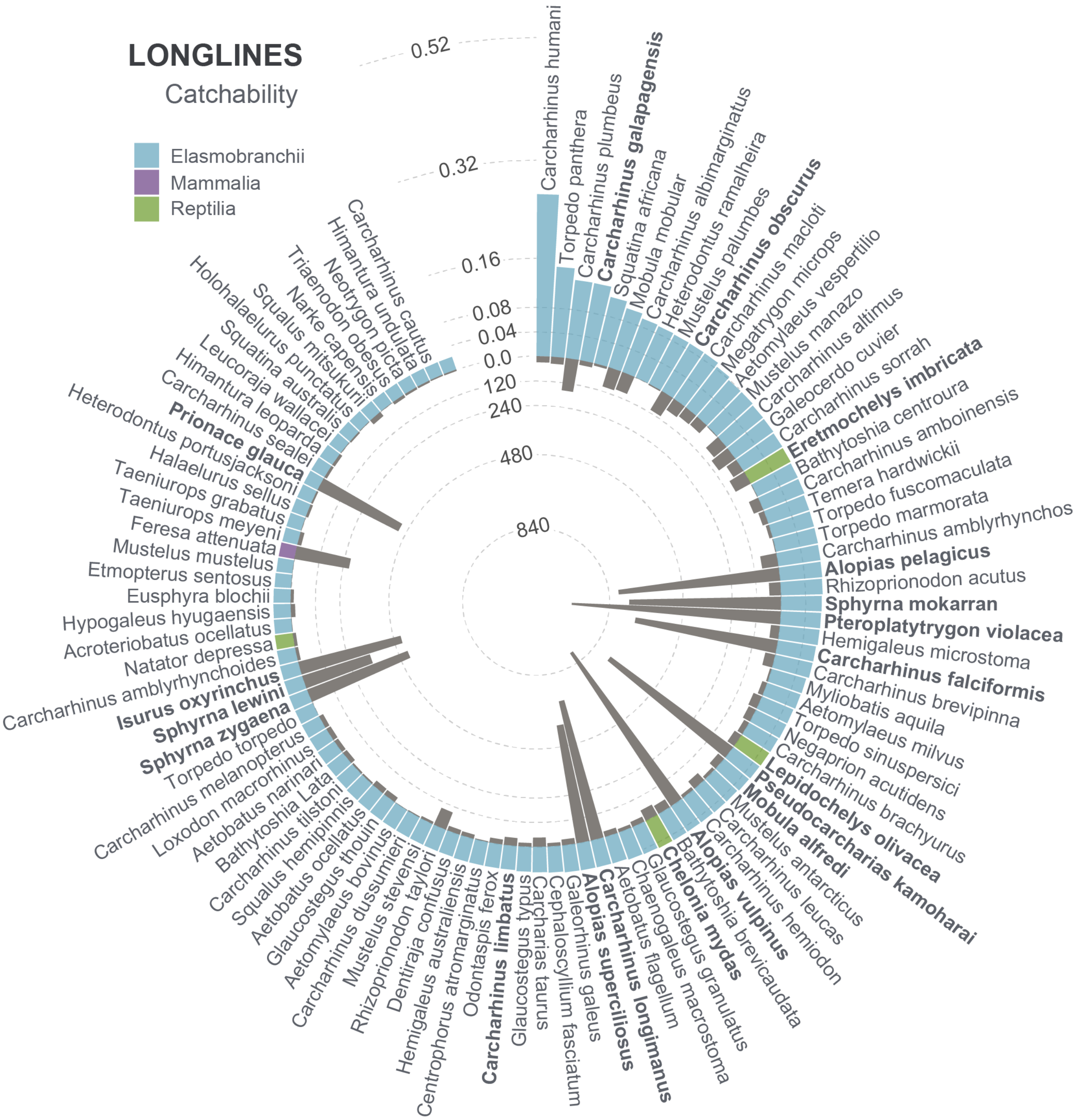

Figure S4.2.3: Mean (outer ring) and cumulative (inner ring) catchability scores for longlines. Species are ordered clockwise by descending mean catchability score and the top 100 species are shown. Bars are colored by taxonomic group (elasmobranchs, cetaceans, and sea turtles).. Species names are in bold if that species is listed in catch records for that gear type in the Indian Ocean (peer reviewed literature or IOTC reports).

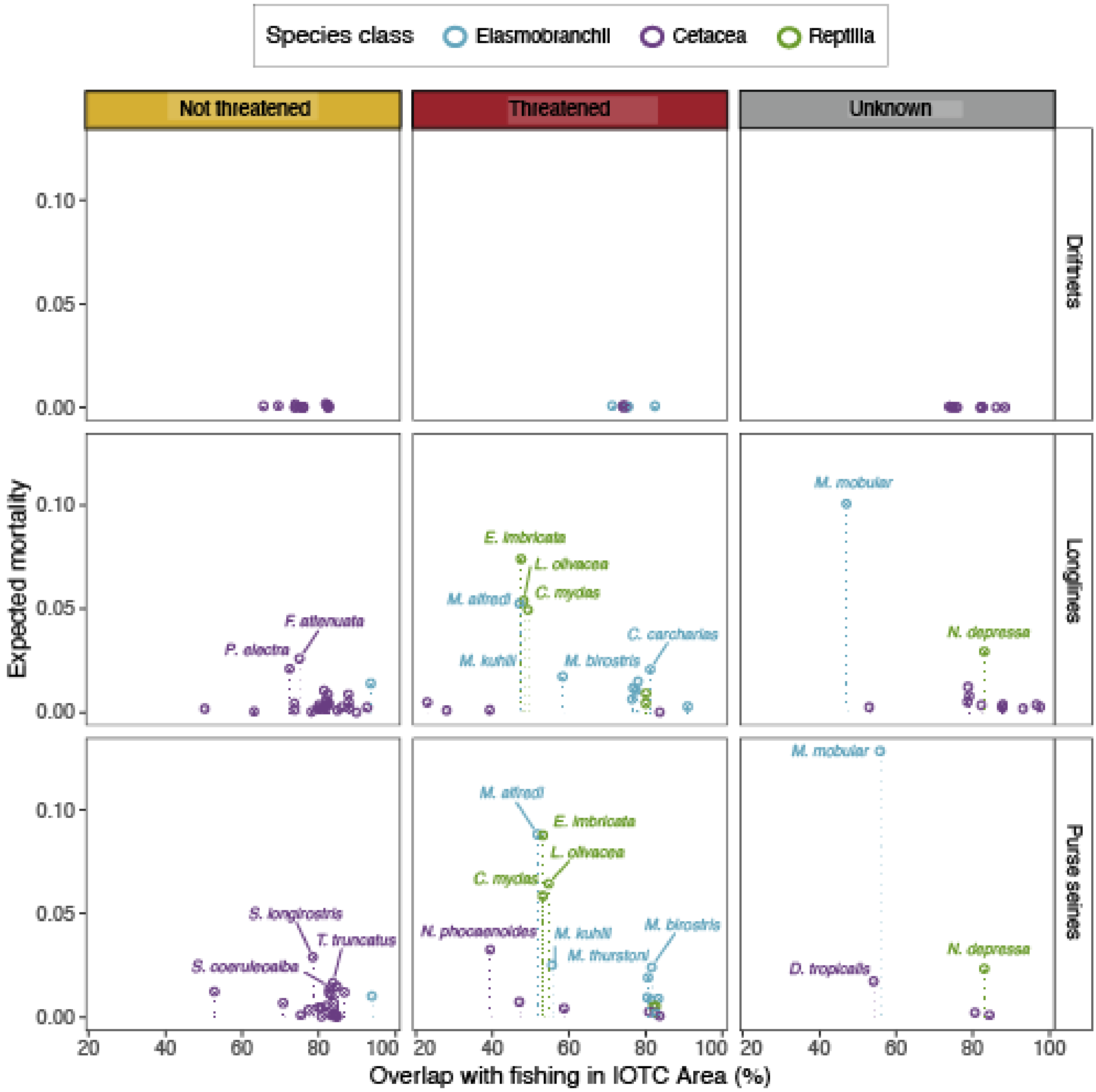

Figure S4.2.4: Mean expected mortality across all cells and percent range overlap with drift gillnets, longlines, and purse seines for the 67 species that were not in the "lethal" category for at least one of the three gears. Species are grouped by conservation status (Threatened = Critically Endangered, Endangered, Vulnerable, Not threatened = Least Concern or Near Threatened, Unknown= Data Deficient or Not Assessed). Species with the 25 highest mean catchability scores overall are labeled.

Appendix 4.3 Supplementary Tables

Table S4.3.1: Fishing countries known to use gillnets targeting tuna or tuna-like species in the Indian Ocean

Country	**Reference**
Australia	Hobsbawn, P.I., Patterson, H.M. and Williams, A.J. (2018) Australian National Report To the Scientific Committee of the Indian Ocean Tuna Commission for 2018.
Bahrain	FAO (2012) Bahrain Skiffs gillnets small pelagics and Spanish mackerel fishery - Gulf Bahraini waters (1-20/40m). Available at: http://firms.fao.org/firms/fishery/670/en#VesseltypeOverview.
Bangladesh	Barua, S., Akter, M.R. and Roy, B. (2018) Bangladesh National Report to the Scientific Committee of the Indian Ocean Tuna Commission, 2018.
Brunei Darussalam	No specific reference for tuna gillnets in the IOTC Area
China	Zhu, J., Wu, F. and Yang, X. (2018) China National Report to the Scientific Committee of the Indian Ocean Tuna Commission, 2018.
Eritrea	Anderson, R., Herrera, M., Ilangakoon, A., Koya, K., Moazzam, M., Mustika, P. and Sutaria, D. (2020) Cetacean bycatch in Indian Ocean tuna gillnet fisheries. *Endangered Species Research* **41**, 39–53.
India	Ramalingam, L., Tiburtius, A., Siva, A., Das, A., Sanadi, R.B. and Kumar Tailor, R.B. (2015) India's National Report to the Scientific Committee of the Indian Ocean Tuna Commission 2015.
Indonesia	Ruchimat, T., Fahmi, Z., Setyadji, B. and Yunanda, T. (2018) Indonesia National Report to the Scientific Committee of the Indian Ocean Tuna Commission, 2018.
Iran	IOTC (2018) I.R.Iran National Report For IOTC-2018-SC21-R10 The 21nd Scientific Committee of the IOTC, 2018.
Kenya	Ndegwa, S. and Okemwa, G. (2017) Kenya National Report to the Scientific Committee of the Indian Ocean Tuna Commission, 2017.
Kuwait	Ye, Y., Al-Husaini, M. and Al-Baz, A. (2001) Use of generalized linear models to analyze catch rates having zero values: The Kuwait driftnet fishery. *Fisheries Research* **53**, 151–168.
Madagascar	Ye, Y., Al-Husaini, M. and Al-Baz, A. (2001) Use of generalized linear models to analyze catch rates having zero values: The Kuwait driftnet fishery. *Fisheries Research* **53**, 151–168.
Malaysia	Samsudin, B., Sallehudin, J., Tengku Balkis, T.. and Nor Azlin, M. (2018) Malaysia National Report to the Scientific Committee of the Indian Ocean Tuna Commission, 2018.
Mauritius	Poonian, C.N.S. (2015) A first assessment of elasmobranch catch in Mauritian artisanal fisheries using interview surveys. *Phelsuma* **23**, 19–29.

Mozambique	Anderson, R., Herrera, M., Ilangakoon, A., Koya, K., Moazzam, M., Mustika, P. and Sutaria, D. (2020) Cetacean bycatch in Indian Ocean tuna gillnet fisheries. *Endangered Species Research* **41**, 39–53.
Myanmar	Alessi, M. De (2017) Fishery Performance Indicators and Coastal Fisheries Management in Southern Rakhine.
Oman	Al-Zaabi, I.A.A. (2015) Sultanate of Oman National Report to the Scientific Committee of the Indian Ocean Tuna Commission, 2015.
Pakistan	Khan, M.W. (2017) Pakistan's National Report to the Scientific Committee of the Indian Ocean Tuna Commission, 2017: IOTC-2017-SC20-NR20 Rev_1.
Qatar	Grandcourt, E.M. (2013) A review of the fisheries, biology, status and management of the narrow-barred Spanish mackerel (Scomberomorus commerson) in the Gulf Cooperation Council countries (Bahrain, Kuwait, Oman, Qatar, Saudi Arabia and the United Arab Emirates).
Saudi Arabia	Abdulqader, E.A.A., Miller, J., Al-Mansi, A., Al-Abdulkader, K., Fita, N., Al-Nadhiri, H. and Rabaoui, L. (2017) Turtles and other marine megafauna bycatch in artisanal fisheries in the Saudi waters of the Arabian Gulf. *Fisheries Research* **196**, 75–84.
Somalia	Breuil, C. and Grima, D. (2014) Country Review Smartfish Programme Somalia. Ebene, Mauritius.
Sri Lanka	Aranda, M. (2017) Description of tuna gillnet capacity and bycatch in the IOTC Convention Area.
Tanzania	Amir, O.A. and Hamid, Z.A. (2016) Tanzania National Report to the Scientific Committee of the Indian Ocean Tuna Commission, 2016. 1–9.
Thailand	Anderson, R., Herrera, M., Ilangakoon, A., Koya, K., Moazzam, M., Mustika, P. and Sutaria, D. (2020) Cetacean bycatch in Indian Ocean tuna gillnet fisheries. *Endangered Species Research* **41**, 39–53.
Timor-Leste	Anderson, R., Herrera, M., Ilangakoon, A., Koya, K., Moazzam, M., Mustika, P. and Sutaria, D. (2020) Cetacean bycatch in Indian Ocean tuna gillnet fisheries. *Endangered Species Research* **41**, 39–53.
UAE	Anderson, R., Herrera, M., Ilangakoon, A., Koya, K., Moazzam, M., Mustika, P. and Sutaria, D. (2020) Cetacean bycatch in Indian Ocean tuna gillnet fisheries. *Endangered Species Research* **41**, 39–53.
Viet Nam	No specific reference for tuna gillnets in the IOTC Area

Table S4.3.2: Taxonomic information for 367 species scoring as catchable in at least one gear type, with selectivity rank and lethality interval for the three gear types (GND=driftnets, PST=purse seines, LLT=longlines). Pot lethal=potentially lethal. Min and max depths are from the AquaMaps model except for 46 species with adjusted depths.

AquaMaps ID	Tax group	Subgroup code	Name	New name	Selectivity rank			Lethality interval			Depth (m)	
					GND	PST	LLT	GND	PST	LLT	Min	Max
ITS-Mam-180524	Cetaceans	BW	*Balaenoptera acutorostrata*	No	9	9	--	Pot.lethal	Sublethal	No damage	0	2000
ITS-Mam-612592	Cetaceans	BW	*Balaenoptera bonaerensis*	No	9	9	--	Pot.lethal	Sublethal	No damage	0	100
ITS-Mam-180526	Cetaceans	BW	*Balaenoptera borealis*	No	9	9	--	Pot.lethal	Sublethal	No damage	0	300
ITS-Mam-612597	Cetaceans	BW	*Balaenoptera brydei*	No	9	9	--	Pot.lethal	Sublethal	No damage	0	2000
ITS-Mam-180525	Cetaceans	BW	*Balaenoptera edeni*	No	9	9	--	Pot.lethal	Sublethal	No damage	0	2000
ITS-Mam-180528	Cetaceans	BW	*Balaenoptera musculus*	No	9	9	--	Pot.lethal	No damage	No damage	0	250
ITS-Mam-180527	Cetaceans	BW	*Balaenoptera physalus*	No	9	9	--	Pot.lethal	Sublethal	No damage	0	250
ITS-Mam-180535	Cetaceans	BW	*Caperea marginata*	No	9	9	--	Pot.lethal	Sublethal	No damage	0	100
ITS-Mam-552771	Cetaceans	BW	*Eubalaena australis*	No	8	9	--	Pot.lethal	Sublethal	No damage	0	175
ITS-Mam-180530	Cetaceans	BW	*Megaptera novaeangliae*	No	8	9	--	Pot.lethal	Sublethal	No damage	0	200
ITS-Mam-180461	Cetaceans	LOD	*Feresa attenuata*	No	7	9	9	Pot.lethal	Sublethal	Sublethal	0	400
ITS-Mam-180466	Cetaceans	LOD	*Globicephala macrorhynchus*	No	7	9	9	Pot.lethal	Sublethal	Sublethal	0	800
ITS-Mam-552461	Cetaceans	LOD	*Globicephala melas*	No	7	9	9	Pot.lethal	Sublethal	Sublethal	0	400
ITS-Mam-180457	Cetaceans	LOD	*Grampus griseus*	No	7	9	9	Pot.lethal	Sublethal	Sublethal	0	1000
ITS-Mam-180469	Cetaceans	LOD	*Orcinus orca*	No	7	9	9	Pot.lethal	Sublethal	Sublethal	0	500
ITS-Mam-180459	Cetaceans	LOD	*Peponocephala electra*	No	7	9	9	Pot.lethal	Sublethal	Sublethal	0	500
ITS-Mam-180463	Cetaceans	LOD	*Pseudorca crassidens*	No	7	9	9	Pot.lethal	Sublethal	Sublethal	0	2000
ITS-Mam-180495	Cetaceans	OCTBW	*Berardius arnuxii*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	1000
ITS-Mam-180505	Cetaceans	OCTBW	*Hyperoodon planifrons*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	2000
ITS-Mam-180502	Cetaceans	OCTBW	*Indopacetus pacificus*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	1500
ITS-Mam-180491	Cetaceans	OCTBW	*Kogia breviceps*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	400

ITS-Mam-180492	Cetaceans	OCTBW	*Kogia sima*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	2000
ITS-Mam-180513	Cetaceans	OCTBW	*Mesoplodon bowdoini*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	2000
ITS-Mam-180517	Cetaceans	OCTBW	*Mesoplodon densirostris*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	2000
ITS-Mam-180510	Cetaceans	OCTBW	*Mesoplodon ginkgodens*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	2000
ITS-Mam-180511	Cetaceans	OCTBW	*Mesoplodon grayi*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	2000
ITS-Mam-180507	Cetaceans	OCTBW	*Mesoplodon hectori*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	2500
ITS-Mam-180516	Cetaceans	OCTBW	*Mesoplodon layardii*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	1000
ITS-Mam-180508	Cetaceans	OCTBW	*Mesoplodon mirus*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	2000
ITS-Mam-180488	Cetaceans	OCTBW	*Physeter macrocephalus*	No	9	--	11	Pot.lethal	No damage	No damage	0	2500
ITS-Mam-180500	Cetaceans	OCTBW	*Tasmacetus shepherdi*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	1000
ITS-Mam-180498	Cetaceans	OCTBW	*Ziphius cavirostris*	No	8	--	11	Pot.lethal	Sublethal	Sublethal	0	3000
ITS-Mam-180451	Cetaceans	SINDP	*Cephalorhynchus heavisidii*	No	3	8	12	Lethal	Pot.lethal	Pot.lethal	0	50
ITS-Mam-180478	Cetaceans	SINDP	*Neophocaena phocaenoides*	No	3	8	12	Lethal	Pot.lethal	Pot.lethal	0	50
ITS-Mam-180471	Cetaceans	SINDP	*Orcaella brevirostris*	No	3	8	12	Lethal	Pot.lethal	Pot.lethal	0	10
ITS-Mam-771132	Cetaceans	SINDP	*Orcaella heinsohni*	No	3	8	12	Lethal	Pot.lethal	Pot.lethal	0	10
ITS-Mam-180475	Cetaceans	SINDP	*Phocoena dioptrica*	No	3	8	12	Lethal	Pot.lethal	Pot.lethal	0	2000
ITS-Mam-180419	Cetaceans	SINDP	*Sousa chinensis*	No	3	8	12	Lethal	Pot.lethal	Pot.lethal	0	25
ITS-Mam-612596	Cetaceans	SINDP	*Tursiops aduncus*	No	3	8	12	Lethal	Pot.lethal	Pot.lethal	0	50
ITS-Mam-180449	Cetaceans	SOCCOD	*Cephalorhynchus commersonii*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	50
ITS-Mam-180438	Cetaceans	SOCCOD	*Delphinus delphis*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	200
ITS-Mam-555654	Cetaceans	SOCCOD	*Delphinus delphis tropicalis*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	100
ITS-Mam-180440	Cetaceans	SOCCOD	*Lagenodelphis hosei*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	600
ITS-Mam-180447	Cetaceans	SOCCOD	*Lagenorhynchus cruciger*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	200
ITS-Mam-180445	Cetaceans	SOCCOD	*Lagenorhynchus obscurus*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	200
ITS-Mam-180455	Cetaceans	SOCCOD	*Lissodelphis peronii*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	200
ITS-Mam-180430	Cetaceans	SOCCOD	*Stenella attenuata*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	100
ITS-Mam-180434	Cetaceans	SOCCOD	*Stenella coeruleoalba*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	700

ITS-Mam-180429	Cetaceans	SOCCOD	*Stenella longirostris*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	250
ITS-Mam-180417	Cetaceans	SOCCOD	*Steno bredanensis*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	100
ITS-Mam-180426	Cetaceans	SOCCOD	*Tursiops truncatus*	No	2	8	12	Lethal	Pot.lethal	Pot.lethal	0	200
Fis-170784	Elasmobranchs	DGE	*Aetomylaeus bovinus*	No	6	9	5	Lethal	Lethal	Lethal	25	150
Fis-140641	Elasmobranchs	DGE	*Asymbolus occiduus*	No	6	9	12	Lethal	Lethal	Lethal	132	400
Fis-140692	Elasmobranchs	DGE	*Asymbolus rubiginosus*	No	6	9	12	Lethal	Lethal	Lethal	86	540
Fis-25886	Elasmobranchs	DGE	*Brachaelurus waddi*	No	6	9	12	Lethal	Lethal	Lethal	15	140
Fis-161440	Elasmobranchs	DGE	*Cephaloscyllium albipinnum*	No	6	9	8	Lethal	Lethal	Lethal	176	554
Fis-161438	Elasmobranchs	DGE	*Cephaloscyllium speccum*	No	6	9	8	Lethal	Lethal	Lethal	184	455
Fis-23084	Elasmobranchs	DGE	*Cephaloscyllium sufflans*	No	6	9	8	Lethal	Lethal	Lethal	107	600
Fis-160851	Elasmobranchs	DGE	*Dentiraja cerva*	Yes	6	9	12	Lethal	Lethal	Lethal	73	470
Fis-161213	Elasmobranchs	DGE	*Dentiraja falloarga*	Yes	6	9	12	Lethal	Lethal	Lethal	81	256
Fis-131829	Elasmobranchs	DGE	*Dipturus pullopunctatus*	No	6	9	12	Lethal	Lethal	Lethal	97	457
Fis-23113	Elasmobranchs	DGE	*Echinorhinus brucus*	No	4	9	12	Lethal	Lethal	Lethal	350	900
Fis-29406	Elasmobranchs	DGE	*Halaelurus boesemani*	No	6	9	12	Lethal	Lethal	Lethal	60	250
Fis-29409	Elasmobranchs	DGE	*Halaelurus lineatus*	No	6	9	12	Lethal	Lethal	Lethal	32	290
Fis-23144	Elasmobranchs	DGE	*Halaelurus natalensis*	No	6	9	12	Lethal	Lethal	Lethal	18	172
Fis-161473	Elasmobranchs	DGE	*Hemitrygon parvonigra*	Yes	6	9	6	Lethal	Lethal	Lethal	130	183
Fis-23149	Elasmobranchs	DGE	*Heptranchias perlo*	No	6	9	7	Lethal	Lethal	Lethal	180	1000
Fis-23153	Elasmobranchs	DGE	*Heterodontus portusjacksoni*	No	6	9	7	Lethal	Lethal	Lethal	31	275
Fis-23155	Elasmobranchs	DGE	*Heterodontus ramalheira*	No	6	9	7	Lethal	Lethal	Lethal	100	275
Fis-33775	Elasmobranchs	DGE	*Heteronarce garmani*	No	6	9	12	Lethal	Lethal	Lethal	101	329
Fis-23157	Elasmobranchs	DGE	*Hexanchus griseus*	No	6	9	6	Lethal	Lethal	Lethal	180	2500
Fis-29416	Elasmobranchs	DGE	*Hexanchus nakamurai*	No	6	9	12	Lethal	Lethal	Lethal	90	600
Fis-132550	Elasmobranchs	DGE	*Leucoraja wallacei*	No	6	9	8	Lethal	Lethal	Lethal	114	450
Fis-31600	Elasmobranchs	DGE	*Mustelus manazo*	No	6	9	4	Lethal	Lethal	Lethal	41	360
Fis-58409	Elasmobranchs	DGE	*Narcine rierai*	No	6	9	12	Lethal	Lethal	Lethal	173	214

Fis-54855	Elasmobranchs	DGE	*Narcinops tasmaniensis*	Yes	6	9	12	Lethal	Lethal	Lethal	82	640
Fis-25888	Elasmobranchs	DGE	*Nebrius ferrugineus*	No	6	9	12	Lethal	Lethal	Lethal	5	70
Fis-58285	Elasmobranchs	DGE	*Neoraja stehmanni*	No	6	9	12	Lethal	Lethal	Lethal	382	1025
Fis-161493	Elasmobranchs	DGE	*Neotrygon annotata*	No	6	9	12	Lethal	Lethal	Lethal	15	62
Fis-24153	Elasmobranchs	DGE	*Notorynchus cepedianus*	No	6	9	12	Lethal	Lethal	Lethal	68	570
Fis-131815	Elasmobranchs	DGE	*Okamejei powelli*	No	6	9	12	Lethal	Lethal	Lethal	135	244
Fis-23202	Elasmobranchs	DGE	*Orectolobus ornatus*	No	6	9	12	Lethal	Lethal	Lethal	10	100
Fis-31583	Elasmobranchs	DGE	*Parascyllium ferrugineum*	No	6	9	12	Lethal	Lethal	Lethal	20	150
Fis-25895	Elasmobranchs	DGE	*Parascyllium variolatum*	No	6	9	12	Lethal	Lethal	Lethal	20	180
Fis-21801	Elasmobranchs	DGE	*Pastinachus sephen*	No	6	9	12	Lethal	Lethal	Lethal	6	60
Fis-35265	Elasmobranchs	DGE	*Pateobatis jenkinsii*	Yes	6	9	12	Lethal	Lethal	Lethal	34	50
Fis-54800	Elasmobranchs	DGE	*Pavoraja nitida*	No	6	9	12	Lethal	Lethal	Lethal	71	390
Fis-29481	Elasmobranchs	DGE	*Pliotrema warreni*	No	6	9	12	Lethal	Lethal	Lethal	60	430
Fis-23220	Elasmobranchs	DGE	*Poroderma africanum*	No	6	9	12	Lethal	Lethal	Lethal	10	100
Fis-23221	Elasmobranchs	DGE	*Poroderma pantherinum*	No	6	9	12	Lethal	Lethal	Lethal	28	256
Fis-23223	Elasmobranchs	DGE	*Pristiophorus cirratus*	No	6	9	12	Lethal	Lethal	Lethal	37	310
Fis-29485	Elasmobranchs	DGE	*Pristiophorus nudipinnis*	No	6	9	12	Lethal	Lethal	Lethal	50	165
Fis-31187	Elasmobranchs	DGE	*Raja miraletus*	No	6	9	12	Lethal	Lethal	Lethal	50	462
Fis-131805	Elasmobranchs	DGE	*Rajella caudaspinosa*	No	6	9	12	Lethal	Lethal	Lethal	357	718
Fis-33777	Elasmobranchs	DGE	*Rhinobatos holcorhynchus*	No	6	9	12	Lethal	Lethal	Lethal	94	253
Fis-32609	Elasmobranchs	DGE	*Rhinobatos schlegelii*	No	6	9	12	Lethal	Lethal	Lethal	66	200
Fis-131821	Elasmobranchs	DGE	*Rostroraja alba*	No	6	9	12	Lethal	Lethal	Lethal	50	600
Fis-23252	Elasmobranchs	DGE	*Scyliorhinus capensis*	No	6	9	12	Lethal	Lethal	Lethal	81	495
Fis-23253	Elasmobranchs	DGE	*Scyliorhinus garmani*	No	6	9	12	Lethal	Lethal	Lethal	116	800
Fis-160854	Elasmobranchs	DGE	*Spiniraja whitleyi*	Yes	6	9	12	Lethal	Lethal	Lethal	21	170
Fis-29539	Elasmobranchs	DGE	*Squatina africana*	No	6	9	6	Lethal	Lethal	Lethal	60	494
Fis-29540	Elasmobranchs	DGE	*Squatina australis*	No	6	9	6	Lethal	Lethal	Lethal	41	256

Fis-160862	Elasmobranchs	DGE	*Squatina pseudocellata*	No	6	9	6	Lethal	Lethal	Lethal	167	312
Fis-29547	Elasmobranchs	DGE	*Squatina tergocellata*	No	6	9	6	Lethal	Lethal	Lethal	250	400
Fis-31247	Elasmobranchs	DGE	*Torpedo marmorata*	No	6	9	7	Lethal	Lethal	Lethal	44	370
Fis-61240	Elasmobranchs	DGE	*Torpedo panthera*	No	6	9	7	Lethal	Lethal	Lethal	41	350
Fis-32610	Elasmobranchs	DGE	*Torpedo sinuspersici*	No	6	9	7	Lethal	Lethal	Lethal	22	200
Fis-53171	Elasmobranchs	DGE	*Urolophus cruciatus*	No	6	9	7	Lethal	Lethal	Lethal	18	160
Fis-54647	Elasmobranchs	DGE	*Urolophus expansus*	No	6	9	7	Lethal	Lethal	Lethal	200	420
Fis-34717	Elasmobranchs	DGE	*Urolophus flavomosaicus*	No	6	9	7	Lethal	Lethal	Lethal	86	300
Fis-47425	Elasmobranchs	DGE	*Urolophus viridis*	No	6	9	7	Lethal	Lethal	Lethal	80	200
Fis-61410	Elasmobranchs	DGE	*Urolophus westraliensis*	No	6	9	7	Lethal	Lethal	Lethal	76	210
Fis-131852	Elasmobranchs	DSE	*Amblyraja hyperborea*	No	--	--	12	Lethal	Lethal	Lethal	300	2500
Fis-140639	Elasmobranchs	DSE	*Asymbolus parvus*	No	--	--	12	Lethal	Lethal	Lethal	170	260
Fis-32598	Elasmobranchs	DSE	*Bathytoshia Lata*	Yes	--	--	7	Lethal	Lethal	Lethal	51	440
Fis-154010	Elasmobranchs	DSE	*Bythaelurus hispidus*	No	--	--	12	Lethal	Lethal	Lethal	222	403
Fis-154012	Elasmobranchs	DSE	*Bythaelurus lutarius*	No	--	--	12	Lethal	Lethal	Lethal	388	766
Fis-131127	Elasmobranchs	DSE	*Centrophorus atromarginatus*	No	--	--	7	Lethal	Lethal	Lethal	213	450
Fis-29321	Elasmobranchs	DSE	*Centrophorus moluccensis*	No	--	--	7	Lethal	Lethal	Lethal	210	823
Fis-23077	Elasmobranchs	DSE	*Centrophorus uyato*	No	--	--	7	Lethal	Lethal	Lethal	200	1400
Fis-23074	Elasmobranchs	DSE	*Centroscymnus crepidater*	No	--	--	7	Lethal	Lethal	Lethal	394	1500
Fis-29332	Elasmobranchs	DSE	*Cephaloscyllium fasciatum*	No	--	--	8	Lethal	Lethal	Lethal	232	450
Fis-161448	Elasmobranchs	DSE	*Cephaloscyllium hiscosellum*	No	--	--	8	Lethal	Lethal	Lethal	307	420
Fis-29338	Elasmobranchs	DSE	*Chlamydoselachus anguineus*	No	--	--	12	Lethal	Lethal	Lethal	120	1570
Fis-131465	Elasmobranchs	DSE	*Cruriraja andamanica*	No	--	--	12	Lethal	Lethal	Lethal	300	511
Fis-164699	Elasmobranchs	DSE	*Cruriraja hulleyi*	No	--	--	12	Lethal	Lethal	Lethal	200	545
Fis-27678	Elasmobranchs	DSE	*Cruriraja parcomaculata*	No	--	--	12	Lethal	Lethal	Lethal	205	620
Fis-23101	Elasmobranchs	DSE	*Dalatias licha*	No	--	--	12	Lethal	Lethal	Lethal	200	1800
Fis-161159	Elasmobranchs	DSE	*Dentiraja healdi*	Yes	--	--	12	Lethal	Lethal	Lethal	327	520

Fis-161218	Elasmobranchs	DSE	*Dentiraja oculata*	Yes	--	--	12	Lethal	Lethal	Lethal	220	389
Fis-132518	Elasmobranchs	DSE	*Dipturus campbelli*	No	--	--	12	Lethal	Lethal	Lethal	167	403
Fis-131844	Elasmobranchs	DSE	*Dipturus stenorhynchus*	No	--	--	12	Lethal	Lethal	Lethal	313	761
Fis-31585	Elasmobranchs	DSE	*Eridacnis radcliffei*	No	--	--	12	Lethal	Lethal	Lethal	156	766
Fis-25897	Elasmobranchs	DSE	*Eridacnis sinuans*	No	--	--	12	Lethal	Lethal	Lethal	214	480
Fis-58162	Elasmobranchs	DSE	*Etmopterus bigelowi*	No	--	--	8	Lethal	Lethal	Lethal	267	1000
Fis-166044	Elasmobranchs	DSE	*Etmopterus sculptus*	No	--	--	8	Lethal	Lethal	Lethal	320	900
Fis-29385	Elasmobranchs	DSE	*Etmopterus sentosus*	No	--	--	8	Lethal	Lethal	Lethal	234	500
Fis-23127	Elasmobranchs	DSE	*Euprotomicrus bispinatus*	No	--	--	12	Lethal	Lethal	Lethal	241	1800
Fis-6652	Elasmobranchs	DSE	*Figaro boardmani*	No	--	--	12	Lethal	Lethal	Lethal	213	823
Fis-125906	Elasmobranchs	DSE	*Galeus gracilis*	No	--	--	12	Lethal	Lethal	Lethal	309	470
Fis-30995	Elasmobranchs	DSE	*Hexatrygon bickelli*	No	--	--	12	Lethal	Lethal	Lethal	362	1120
Fis-31589	Elasmobranchs	DSE	*Iago garricki*	No	5	--	12	Lethal	Lethal	Lethal	275	475
Fis-25903	Elasmobranchs	DSE	*Iago omanensis*	No	5	--	12	Lethal	Lethal	Lethal	394	2195
Fis-161235	Elasmobranchs	DSE	*Irolita westraliensis*	No	--	--	12	Lethal	Lethal	Lethal	148	209
Fis-161233	Elasmobranchs	DSE	*Leucoraja pristispina*	No	--	--	8	Lethal	Lethal	Lethal	236	504
Fis-31578	Elasmobranchs	DSE	*Mitsukurina owstoni*	No	--	--	12	Lethal	Lethal	Lethal	270	1300
Fis-149485	Elasmobranchs	DSE	*Narcinops lasti*	Yes	--	--	12	Lethal	Lethal	Lethal	196	350
Fis-23198	Elasmobranchs	DSE	*Odontaspis ferox*	No	--	--	6	Lethal	Lethal	Lethal	72	530
Fis-161225	Elasmobranchs	DSE	*Okamejei arafurensis*	No	--	--	12	Lethal	Lethal	Lethal	191	298
Fis-132528	Elasmobranchs	DSE	*Okamejei heemstrai*	No	--	--	12	Lethal	Lethal	Lethal	286	500
Fis-161228	Elasmobranchs	DSE	*Okamejei leptoura*	No	--	--	12	Lethal	Lethal	Lethal	265	735
Fis-144985	Elasmobranchs	DSE	*Parascyllium sparsimaculatum*	No	--	--	12	Lethal	Lethal	Lethal	208	245
Fis-54790	Elasmobranchs	DSE	*Pavoraja alleni*	No	--	--	12	Lethal	Lethal	Lethal	320	458
Fis-161638	Elasmobranchs	DSE	*Pavoraja arenaria*	No	--	--	12	Lethal	Lethal	Lethal	300	712
Fis-26519	Elasmobranchs	DSE	*Plesiobatis daviesi*	No	--	--	12	Lethal	Lethal	Lethal	275	780
Fis-165849	Elasmobranchs	DSE	*Pristiophorus nancyae*	No	--	--	12	Lethal	Lethal	Lethal	318	570

Fis-132559	Elasmobranchs	DSE	*Rajella barnardi*	No	--	--	12	Lethal	Lethal	Lethal	372	1700
Fis-160879	Elasmobranchs	DSE	*Sinobatis bulbicauda*	No	--	--	12	Lethal	Lethal	Lethal	273	1125
Fis-29531	Elasmobranchs	DSE	*Squaliolus laticaudus*	No	--	--	7	Lethal	Lethal	Lethal	326	1200
Fis-160439	Elasmobranchs	DSE	*Squalus edmundsi*	No	--	--	7	Lethal	Lethal	Lethal	300	850
Fis-160378	Elasmobranchs	DSE	*Squalus montalbani*	No	--	--	7	Lethal	Lethal	Lethal	383	1370
Fis-160444	Elasmobranchs	DSE	*Squalus nasutus*	No	--	--	7	Lethal	Lethal	Lethal	300	850
Fis-23075	Elasmobranchs	DSPE	*Centrophorus granulosus*	No	1--	--	7	Lethal	Lethal	Lethal	200	1200
Fis-29319	Elasmobranchs	DSPE	*Centrophorus harrissoni*	No	1--	--	7	Lethal	Lethal	Lethal	314	790
Fis-23278	Elasmobranchs	DSPE	*Cirrhigaleus asper*	No	1--	--	7	Lethal	Lethal	Lethal	253	650
Fis-29380	Elasmobranchs	DSPE	*Etmopterus gracilispinis*	No	1--	--	7	Lethal	Lethal	Lethal	187	1000
Fis-23124	Elasmobranchs	DSPE	*Etmopterus spinax*	No	1--	--	7	Lethal	Lethal	Lethal	200	2490
Fis-23204	Elasmobranchs	DSPE	*Oxynotus bruniensis*	No	1--	--	7	Lethal	Lethal	Lethal	350	1070
Fis-61614	Elasmobranchs	DSPE	*Scymnodalatias albicauda*	No	1--	--	7	Lethal	Lethal	Lethal	191	510
Fis-23260	Elasmobranchs	DSPE	*Somniosus rostratus*	No	1--	--	7	Lethal	Lethal	Lethal	345	1330
Fis-29532	Elasmobranchs	DSPE	*Squalus acanthias*	No	1--	--	7	Lethal	Lethal	Lethal	50	1460
Fis-159586	Elasmobranchs	DSPE	*Squalus crassispinus*	No	1--	--	7	Lethal	Lethal	Lethal	194	262
Fis-29536	Elasmobranchs	DSPE	*Squalus mitsukurii*	No	1--	--	7	Lethal	Lethal	Lethal	48	600
Fis-31408	Elasmobranchs	INE	*Acroteriobatus annulatus*	Yes	6	8	12	Lethal	Lethal	Lethal	7	73
Fis-32608	Elasmobranchs	INE	*Acroteriobatus blochii*	Yes	5	8	12	Lethal	Lethal	Lethal	3	30
Fis-27240	Elasmobranchs	INE	*Aetobatus flagellum*	No	5	7	5	Lethal	Lethal	Lethal	9	80
Fis-28560	Elasmobranchs	INE	*Aetomylaeus maculatus*	No	5	7	5	Lethal	Lethal	Lethal	2	18
Fis-28561	Elasmobranchs	INE	*Aetomylaeus milvus*	No	5	7	5	Lethal	Lethal	Lethal	10	100
Fis-26906	Elasmobranchs	INE	*Anoxypristis cuspidata*	No	6	8	12	Lethal	Lethal	Lethal	4	40
Fis-131407	Elasmobranchs	INE	*Atelomycterus fasciatus*	No	6	8	12	Lethal	Lethal	Lethal	37	122
Fis-29298	Elasmobranchs	INE	*Atelomycterus macleayi*	No	6	8	12	Lethal	Lethal	Lethal	0	4
Fis-26085	Elasmobranchs	INE	*Bathytoshia brevicaudata*	Yes	5	8	7	Lethal	Lethal	Lethal	0	476
Fis-23993	Elasmobranchs	INE	*Bathytoshia centroura*	Yes	5	8	7	Lethal	Lethal	Lethal	15	270

Fis-23055	Elasmobranchs	INE	*Carcharhinus fitzroyensis*	No	6	8	4	Lethal	Lethal	Lethal	4	40
Fis-23082	Elasmobranchs	INE	*Cephaloscyllium laticeps*	No	6	8	8	Lethal	Lethal	Lethal	25	220
Fis-58398	Elasmobranchs	INE	*Dasyatis chrysonota*	No	4	8	7	Lethal	Lethal	Lethal	11	100
Fis-33107	Elasmobranchs	INE	*Dasyatis marmorata*	No	4	8	7	Lethal	Lethal	Lethal	17	65
Fis-60598	Elasmobranchs	INE	*Fontitrygon margaritella*	Yes	6	8	12	Lethal	Lethal	Lethal	6	50
Fis-23138	Elasmobranchs	INE	*Glyphis gangeticus*	No	6	8	12	Lethal	Lethal	Lethal	3	20
Fis-161453	Elasmobranchs	INE	*Glyphis garricki*	No	6	8	12	Lethal	Lethal	Lethal	1	11
Fis-24044	Elasmobranchs	INE	*Gymnura altavela*	No	6	8	12	Lethal	Lethal	Lethal	15	100
Fis-24046	Elasmobranchs	INE	*Gymnura micrura*	No	6	8	12	Lethal	Lethal	Lethal	6	55
Fis-15849	Elasmobranchs	INE	*Gymnura natalensis*	No	6	8	12	Lethal	Lethal	Lethal	28	100
Fis-26932	Elasmobranchs	INE	*Gymnura poecilura*	No	6	8	12	Lethal	Lethal	Lethal	3	25
Fis-28559	Elasmobranchs	INE	*Gymnura zonura*	No	6	8	12	Lethal	Lethal	Lethal	29	37
Fis-154456	Elasmobranchs	INE	*Hemitrygon bennetti*	Yes	6	8	7	Lethal	Lethal	Lethal	5	40
Fis-28555	Elasmobranchs	INE	*Himantura granulata*	No	6	8	12	Lethal	Lethal	Lethal	9	85
Fis-148497	Elasmobranchs	INE	*Lamiopsis temminckii*	No	6	8	12	Lethal	Lethal	Lethal	5	50
Fis-161488	Elasmobranchs	INE	*Maculabatis astra*	Yes	6	8	12	Lethal	Lethal	Lethal	16	141
Fis-47488	Elasmobranchs	INE	*Maculabatis gerrardi*	Yes	6	8	12	Lethal	Lethal	Lethal	6	50
Fis-166946	Elasmobranchs	INE	*Maculabatis randalli*	Yes	6	8	12	Lethal	Lethal	Lethal	5	40
Fis-47495	Elasmobranchs	INE	*Maculabatis toshi*	Yes	6	8	12	Lethal	Lethal	Lethal	23	140
Fis-47352	Elasmobranchs	INE	*Megatrygon microps*	Yes	5	8	7	Lethal	Lethal	Lethal	22	200
Fis-25062	Elasmobranchs	INE	*Myliobatis aquila*	No	5	8	5	Lethal	Lethal	Lethal	35	300
Fis-47427	Elasmobranchs	INE	*Narcine lingula*	No	6	8	12	Lethal	Lethal	Lethal	22	200
Fis-26903	Elasmobranchs	INE	*Narcine timlei*	No	6	8	12	Lethal	Lethal	Lethal	22	200
Fis-28947	Elasmobranchs	INE	*Narke dipterygia*	No	6	8	12	Lethal	Lethal	Lethal	39	200
Fis-28785	Elasmobranchs	INE	*Pateobatis bleekeri*	Yes	6	8	12	Lethal	Lethal	Lethal	3	30
Fis-35264	Elasmobranchs	INE	*Pateobatis fai*	Yes	6	8	12	Lethal	Lethal	Lethal	22	200
Fis-27224	Elasmobranchs	INE	*Pristis pristis*	No	6	8	12	Lethal	Lethal	Lethal	5	50

Fis-32599	Elasmobranchs	INE	*Pristis zijsron*	No	6	8	12	Lethal	Lethal	Lethal	0	5
Fis-22814	Elasmobranchs	INE	*Pseudobatus percellens*	Yes	6	8	12	Lethal	Lethal	Lethal	11	110
Fis-64122	Elasmobranchs	INE	*Raja pita*	No	6	8	12	Lethal	Lethal	Lethal	1	15
Fis-57444	Elasmobranchs	INE	*Rhinobatos punctifer*	No	6	8	12	Lethal	Lethal	Lethal	16	150
Fis-29505	Elasmobranchs	INE	*Scoliodon laticaudus*	No	6	8	12	Lethal	Lethal	Lethal	10	13
Fis-27236	Elasmobranchs	INE	*Taeniurops grabatus*	Yes	6	8	12	Lethal	Lethal	Lethal	42	300
Fis-166734	Elasmobranchs	INE	*Taeniurops meyeni*	No	6	8	12	Lethal	Lethal	Lethal	20	500
Fis-32811	Elasmobranchs	INE	*Temera hardwickii*	No	6	8	7	Lethal	Lethal	Lethal	39	200
Fis-31223	Elasmobranchs	INE	*Torpedo fuscomaculata*	No	6	8	7	Lethal	Lethal	Lethal	51	439
Fis-25905	Elasmobranchs	INE	*Triakis megalopterus*	No	6	8	12	Lethal	Lethal	Lethal	6	50
Fis-161466	Elasmobranchs	INE	*Trygonoptera imitata*	No	6	8	12	Lethal	Lethal	Lethal	13	120
Fis-61250	Elasmobranchs	INE	*Trygonoptera ovalis*	No	6	8	12	Lethal	Lethal	Lethal	8	43
Fis-6035	Elasmobranchs	INE	*Urogymnus asperrimus*	No	5	8	12	Lethal	Lethal	Lethal	3	20
Fis-31568	Elasmobranchs	OCE	*Alopias pelagicus*	No	5	4	1	Lethal	Lethal	Lethal	0	300
Fis-23898	Elasmobranchs	OCE	*Alopias superciliosus*	No	5	4	1	Lethal	Lethal	Lethal	0	730
Fis-23899	Elasmobranchs	OCE	*Alopias vulpinus*	No	5	4	1	Lethal	Lethal	Lethal	0	650
Fis-23061	Elasmobranchs	OCE	*Carcharhinus longimanus*	No	5	2	1	Lethal	Lethal	Lethal	0	230
Fis-58485	Elasmobranchs	OCE	*Isurus oxyrinchus*	No	5	4	1	Lethal	Lethal	Lethal	100	750
Fis-29423	Elasmobranchs	OCE	*Isurus paucus*	No	5	4	1	Lethal	Lethal	Lethal	234	1752
Fis-25899	Elasmobranchs	OCE	*Pseudocarcharias kamoharai*	No	5	4	1	Lethal	Lethal	Lethal	0	590
Fis-22747	Elasmobranchs	PFFE	*Cetorhinus maximus*	No	4	5	7	Pot.lethal	Pot.lethal	Pot.lethal	0	2000
Fis-31577	Elasmobranchs	PFFE	*Megachasma pelagios*	No	4	5	1--	Pot.lethal	Pot.lethal	Pot.lethal	120	600
Fis-163295	Elasmobranchs	PFFE	*Mobula alfredi*	Yes	4	5	6	Lethal	Pot.lethal	Pot.lethal	13	120
Fis-24098	Elasmobranchs	PFFE	*Mobula birostris*	Yes	4	5	6	Lethal	Pot.lethal	Pot.lethal	12	120
Fis-61508	Elasmobranchs	PFFE	*Mobula kuhlii*	No	4	5	6	Lethal	Pot.lethal	Pot.lethal	10	100
Fis-21798	Elasmobranchs	PFFE	*Mobula Mobula*	Yes	4	5	6	Lethal	Pot.lethal	Pot.lethal	0	300
Fis-35514	Elasmobranchs	PFFE	*Mobula tarapacana*	No	4	5	6	Lethal	Pot.lethal	Pot.lethal	0	1896

Fis-24127	Elasmobranchs	PFFE	*Mobula thurstoni*	No	4	5	6	Lethal	Pot.lethal	Pot.lethal	10	100
Fis-30583	Elasmobranchs	PFFE	*Rhincodon typus*	No	4	5	1--	Pot.lethal	Pot.lethal	Pot.lethal	0	1928
Fis-23322	Elasmobranchs	PGE	*Aetobatus narinari*	No	5	6	5	Lethal	Lethal	Lethal	1	80
Fis-28563	Elasmobranchs	PGE	*Aetobatus ocellatus*	No	5	6	5	Lethal	Lethal	Lethal	20	100
Fis-28562	Elasmobranchs	PGE	*Aetomylaeus vespertilio*	No	5	6	2	Lethal	Lethal	Lethal	11	110
Fis-23044	Elasmobranchs	PGE	*Carcharhinus albimarginatus*	No	5	6	4	Lethal	Lethal	Lethal	20	800
Fis-23054	Elasmobranchs	PGE	*Carcharhinus falciformis*	No	5	1	1	Lethal	Lethal	Lethal	0	500
Fis-23056	Elasmobranchs	PGE	*Carcharhinus galapagensis*	No	5	6	2	Lethal	Lethal	Lethal	30	286
Fis-23057	Elasmobranchs	PGE	*Carcharhinus hemiodon*	No	5	6	2	Lethal	Lethal	Lethal	6	50
Fis-169677	Elasmobranchs	PGE	*Carcharhinus humani*	No	5	6	2	Lethal	Lethal	Lethal	22.5	408
Fis-23064	Elasmobranchs	PGE	*Carcharhinus obscurus*	No	5	6	2	Lethal	Lethal	Lethal	200	400
Fis-23066	Elasmobranchs	PGE	*Carcharhinus plumbeus*	No	5	6	2	Lethal	Lethal	Lethal	20	500
Fis-23071	Elasmobranchs	PGE	*Carcharodon carcharias*	No	6	6	2	Pot.lethal	Pot.lethal	Sublethal	0	1200
Fis-29367	Elasmobranchs	PGE	*Echinorhinus cookei*	No	5	6	2	Lethal	Lethal	Lethal	70	1100
Fis-23129	Elasmobranchs	PGE	*Galeocerdo cuvier*	No	5	6	2	Lethal	Lethal	Lethal	0	800
Fis-25233	Elasmobranchs	PGE	*Galeorhinus galeus*	No	5	6	2	Lethal	Lethal	Lethal	2	1100
Fis-22768	Elasmobranchs	PGE	*Lamna nasus*	No	5	6	2	Lethal	Lethal	Lethal	87	715
Fis-25412	Elasmobranchs	PGE	*Mustelus mustelus*	No	5	6	4	Lethal	Lethal	Lethal	5	624
Fis-31594	Elasmobranchs	PGE	*Mustelus palumbes*	No	5	6	4	Lethal	Lethal	Lethal	52	443
Fis-161402	Elasmobranchs	PGE	*Mustelus stevensi*	No	5	6	4	Lethal	Lethal	Lethal	152	402
Fis-32960	Elasmobranchs	PGE	*Myliobatis tenuicaudatus*	Yes	5	6	5	Lethal	Lethal	Lethal	9	85
Fis-23193	Elasmobranchs	PGE	*Negaprion acutidens*	No	5	6	2	Lethal	Lethal	Lethal	9	92
Fis-23222	Elasmobranchs	PGE	*Prionace glauca*	No	5	6	1	Lethal	Lethal	Lethal	1	1000
Fis-20033	Elasmobranchs	PGE	*Pteroplatytrygon violacea*	No	5	6	2	Lethal	Lethal	Lethal	1	381
Fis-32611	Elasmobranchs	PGE	*Rhinoptera javanica*	No	5	6	2	Lethal	Lethal	Lethal	3	30
Fis-23280	Elasmobranchs	PGE	*Squalus megalops*	No	5	6	2	Lethal	Lethal	Lethal	118	750
Fis-23028	Elasmobranchs	RE	*Asymbolus analis*	No	8	8	12	Lethal	Lethal	Lethal	10	180

Fis-140634	Elasmobranchs	RE	*Asymbolus submaculatus*	No	8	8	12	Lethal	Lethal	Lethal	48	200
Fis-23029	Elasmobranchs	RE	*Asymbolus vincenti*	No	8	8	12	Lethal	Lethal	Lethal	102	650
Fis-24448	Elasmobranchs	RE	*Atelomycterus marmoratus*	No	8	8	12	Lethal	Lethal	Lethal	5	25
Fis-23046	Elasmobranchs	RE	*Carcharhinus amblyrhynchoides*	No	8	8	4	Lethal	Lethal	Lethal	5	50
Fis-23047	Elasmobranchs	RE	*Carcharhinus amblyrhynchos*	No	8	8	4	Lethal	Lethal	Lethal	0	1000
Fis-23063	Elasmobranchs	RE	*Carcharhinus melanopterus*	No	8	8	4	Lethal	Lethal	Lethal	25	75
Fis-31571	Elasmobranchs	RE	*Chiloscyllium arabicum*	No	8	8	12	Lethal	Lethal	Lethal	13	100
Fis-30780	Elasmobranchs	RE	*Chiloscyllium griseum*	No	8	8	12	Lethal	Lethal	Lethal	12	80
Fis-132130	Elasmobranchs	RE	*Chiloscyllium hasseltii*	No	8	8	12	Lethal	Lethal	Lethal	1	12
Fis-25892	Elasmobranchs	RE	*Chiloscyllium indicum*	No	8	8	12	Lethal	Lethal	Lethal	2	20
Fis-25470	Elasmobranchs	RE	*Chiloscyllium plagiosum*	No	8	8	12	Lethal	Lethal	Lethal	7	25
Fis-31573	Elasmobranchs	RE	*Chiloscyllium punctatum*	No	8	8	12	Lethal	Lethal	Lethal	8	85
Fis-23126	Elasmobranchs	RE	*Eucrossorhinus dasypogon*	No	8	8	12	Lethal	Lethal	Lethal	5	40
Fis-25894	Elasmobranchs	RE	*Hemiscyllium ocellatum*	No	8	8	12	Lethal	Lethal	Lethal	5	50
Fis-31576	Elasmobranchs	RE	*Hemiscyllium trispeculare*	No	8	8	12	Lethal	Lethal	Lethal	5	50
Fis-23156	Elasmobranchs	RE	*Heterodontus zebra*	No	8	8	12	Lethal	Lethal	Lethal	66	200
Fis-161494	Elasmobranchs	RE	*Neotrygon kuhlii*	No	4	8	12	Lethal	Lethal	Lethal	9	170
Fis-23201	Elasmobranchs	RE	*Orectolobus maculatus*	No	8	8	12	Lethal	Lethal	Lethal	0	110
Fis-29459	Elasmobranchs	RE	*Orectolobus wardi*	No	8	8	12	Lethal	Lethal	Lethal	1	3
Fis-32975	Elasmobranchs	RE	*Rhina ancylostoma*	No	8	8	12	Lethal	Lethal	Lethal	12	90
Fis-8339	Elasmobranchs	RE	*Stegostoma fasciatum*	No	8	8	7	Lethal	Lethal	Lethal	5	63
Fis-23292	Elasmobranchs	RE	*Sutorectus tentaculatus*	No	8	8	12	Lethal	Lethal	Lethal	5	50
Fis-25603	Elasmobranchs	RE	*Taeniura lymma*	No	8	8	12	Lethal	Lethal	Lethal	3	20
Fis-23311	Elasmobranchs	RE	*Triaenodon obesus*	No	8	8	12	Lethal	Lethal	Lethal	8	330
Fis-47714	Elasmobranchs	SSE	*Acroteriobatus ocellatus*	Yes	1	8	7	Lethal	Lethal	Lethal	73	185
Fis-27676	Elasmobranchs	SSE	*Aetomylaeus nichofii*	No	1	7	5	Lethal	Lethal	Lethal	8	70

Fis-54720	Elasmobranchs	SSE	*Aptychotrema vincentiana*	No	1	8	7	Lethal	Lethal	Lethal	4	32
Fis-28787	Elasmobranchs	SSE	*Brevitrygon imbricata*	Yes	1	8	7	Lethal	Lethal	Lethal	6	50
Fis-23045	Elasmobranchs	SSE	*Carcharhinus altimus*	No	1	8	4	Lethal	Lethal	Lethal	80	810
Fis-23048	Elasmobranchs	SSE	*Carcharhinus amboinensis*	No	1	8	4	Lethal	Lethal	Lethal	16	150
Fis-23050	Elasmobranchs	SSE	*Carcharhinus brachyurus*	No	1	8	4	Lethal	Lethal	Lethal	41	360
Fis-23051	Elasmobranchs	SSE	*Carcharhinus brevipinna*	No	1	8	2	Lethal	Lethal	Lethal	0	100
Fis-23052	Elasmobranchs	SSE	*Carcharhinus cautus*	No	1	8	4	Lethal	Lethal	Lethal	6	50
Fis-23053	Elasmobranchs	SSE	*Carcharhinus dussumieri*	No	1	8	4	Lethal	Lethal	Lethal	10	100
Fis-23059	Elasmobranchs	SSE	*Carcharhinus leucas*	No	1	8	4	Lethal	Lethal	Lethal	1	152
Fis-23060	Elasmobranchs	SSE	*Carcharhinus limbatus*	No	1	8	4	Lethal	Lethal	Lethal	0	100
Fis-23062	Elasmobranchs	SSE	*Carcharhinus macloti*	No	1	8	4	Lethal	Lethal	Lethal	19	170
Fis-23068	Elasmobranchs	SSE	*Carcharhinus sealei*	No	1	8	4	Lethal	Lethal	Lethal	4	40
Fis-23070	Elasmobranchs	SSE	*Carcharhinus sorrah*	No	1	8	4	Lethal	Lethal	Lethal	1	140
Fis-47835	Elasmobranchs	SSE	*Carcharhinus tilstoni*	No	1	8	4	Lethal	Lethal	Lethal	16	150
Fis-29388	Elasmobranchs	SSE	*Carcharias taurus*	No	1	8	5	Lethal	Lethal	Lethal	15	191
Fis-25889	Elasmobranchs	SSE	*Chaenogaleus macrostoma*	No	1	8	5	Lethal	Lethal	Lethal	6	59
Fis-161209	Elasmobranchs	SSE	*Dentiraja confusus*	Yes	1	8	7	Lethal	Lethal	Lethal	18	390
Fis-164471	Elasmobranchs	SSE	*Dentiraja lemprieri*	No	1	8	7	Lethal	Lethal	Lethal	0	170
Fis-160925	Elasmobranchs	SSE	*Electrolux addisoni*	No	1	8	7	Lethal	Lethal	Lethal	9	35
Fis-23128	Elasmobranchs	SSE	*Eusphyra blochii*	No	1	8	3	Lethal	Lethal	Lethal	9	50
Fis-25900	Elasmobranchs	SSE	*Furgaleus macki*	No	1	8	7	Lethal	Lethal	Lethal	27	220
Fis-159583	Elasmobranchs	SSE	*Glaucostegus granulatus*	No	1	8	7	Lethal	Lethal	Lethal	12	119
Fis-159582	Elasmobranchs	SSE	*Glaucostegus halavi*	No	1	8	7	Lethal	Lethal	Lethal	4	40
Fis-28552	Elasmobranchs	SSE	*Glaucostegus thouin*	Yes	1	8	7	Lethal	Lethal	Lethal	11	100
Fis-159584	Elasmobranchs	SSE	*Glaucostegus typus*	No	1	8	7	Lethal	Lethal	Lethal	10	100
Fis-47368	Elasmobranchs	SSE	*Gymnura australis*	No	1	8	7	Lethal	Lethal	Lethal	5	50
Fis-160267	Elasmobranchs	SSE	*Halaelurus sellus*	No	1	8	7	Lethal	Lethal	Lethal	72	164

Fis-23146	Elasmobranchs	SSE	*Haploblepharus edwardsii*	No	1	8	7	Lethal	Lethal	Lethal	40	130
Fis-29411	Elasmobranchs	SSE	*Haploblepharus fuscus*	No	1	8	7	Lethal	Lethal	Lethal	7	25
Fis-156398	Elasmobranchs	SSE	*Hemigaleus australiensis*	No	1	8	7	Lethal	Lethal	Lethal	29	170
Fis-31570	Elasmobranchs	SSE	*Hemigaleus microstoma*	No	1	8	7	Lethal	Lethal	Lethal	30	200
Fis-48194	Elasmobranchs	SSE	*Hemipristis elongata*	No	1	8	7	Lethal	Lethal	Lethal	14	130
Fis-161480	Elasmobranchs	SSE	*Himantura leoparda*	No	1	8	7	Lethal	Lethal	Lethal	8	70
Fis-26148	Elasmobranchs	SSE	*Himantura uarnak*	No	1	8	7	Lethal	Lethal	Lethal	23	50
Fis-28553	Elasmobranchs	SSE	*Himantura undulata*	No	1	8	7	Lethal	Lethal	Lethal	6	50
Fis-160938	Elasmobranchs	SSE	*Holohalaelurus favus*	No	1	8	7	Lethal	Lethal	Lethal	299	1000
Fis-23158	Elasmobranchs	SSE	*Holohalaelurus punctatus*	No	1	8	7	Lethal	Lethal	Lethal	244	440
Fis-23159	Elasmobranchs	SSE	*Holohalaelurus regani*	No	1	8	7	Lethal	Lethal	Lethal	150	1075
Fis-139820	Elasmobranchs	SSE	*Hypnos monopterygius*	No	1	8	7	Lethal	Lethal	Lethal	26	240
Fis-25902	Elasmobranchs	SSE	*Hypogaleus hyugaensis*	No	1	8	7	Lethal	Lethal	Lethal	60	230
Fis-54787	Elasmobranchs	SSE	*Irolita waitii*	No	1	8	7	Lethal	Lethal	Lethal	66	200
Fis-29421	Elasmobranchs	SSE	*Isistius plutodus*	No	1	8	7	Lethal	Lethal	Lethal	75	200
Fis-29436	Elasmobranchs	SSE	*Loxodon macrorhinus*	No	1	8	7	Lethal	Lethal	Lethal	16	100
Fis-31602	Elasmobranchs	SSE	*Mustelus antarcticus*	No	1	8	4	Lethal	Lethal	Lethal	40	350
Fis-160464	Elasmobranchs	SSE	*Mustelus ravidus*	No	1	8	4	Lethal	Lethal	Lethal	127	300
Fis-161457	Elasmobranchs	SSE	*Narcinops ornata*	Yes	1	8	7	Lethal	Lethal	Lethal	56	132
Fis-54860	Elasmobranchs	SSE	*Narcinops westraliensis*	Yes	1	8	7	Lethal	Lethal	Lethal	16	70
Fis-58273	Elasmobranchs	SSE	*Narke capensis*	No	1	8	7	Lethal	Lethal	Lethal	37	183
Fis-161495	Elasmobranchs	SSE	*Neotrygon leylandi*	No	1	8	7	Lethal	Lethal	Lethal	12	80
Fis-161491	Elasmobranchs	SSE	*Neotrygon picta*	No	1	8	7	Lethal	Lethal	Lethal	14	96
Fis-160886	Elasmobranchs	SSE	*Orectolobus floridus*	No	1	8	7	Lethal	Lethal	Lethal	46	85
Fis-159132	Elasmobranchs	SSE	*Orectolobus hutchinsi*	No	1	8	7	Lethal	Lethal	Lethal	0	106
Fis-160887	Elasmobranchs	SSE	*Orectolobus parvimaculatus*	No	1	8	7	Lethal	Lethal	Lethal	22	135
Fis-25414	Elasmobranchs	SSE	*Paragaleus pectoralis*	No	1	8	7	Lethal	Lethal	Lethal	30	100

Fis-140161	Elasmobranchs	SSE	*Paragaleus randalli*	No	1	8	7	Lethal	Lethal	Lethal	2	18
Fis-159136	Elasmobranchs	SSE	*Rhinobatos sainsburyi*	No	1	8	7	Lethal	Lethal	Lethal	80	200
Fis-23239	Elasmobranchs	SSE	*Rhizoprionodon acutus*	No	1	8	7	Lethal	Lethal	Lethal	22	200
Fis-29500	Elasmobranchs	SSE	*Rhizoprionodon oligolinx*	No	1	8	7	Lethal	Lethal	Lethal	3	36
Fis-23243	Elasmobranchs	SSE	*Rhizoprionodon taylori*	No	1	8	7	Lethal	Lethal	Lethal	34	300
Fis-25664	Elasmobranchs	SSE	*Rhynchobatus djiddensis*	No	1	8	7	Lethal	Lethal	Lethal	6	50
Fis-161456	Elasmobranchs	SSE	*Rhynchobatus palpebratus*	No	1	8	7	Lethal	Lethal	Lethal	10	61
Fis-23273	Elasmobranchs	SSE	*Sphyrna lewini*	No	1	7	3	Lethal	Lethal	Lethal	0	1000
Fis-23274	Elasmobranchs	SSE	*Sphyrna mokarran*	No	1	7	3	Lethal	Lethal	Lethal	1	300
Fis-23277	Elasmobranchs	SSE	*Sphyrna zygaena*	No	1	7	3	Lethal	Lethal	Lethal	0	200
Fis-160691	Elasmobranchs	SSE	*Squalus hemipinnis*	No	1	8	7	Lethal	Lethal	Lethal	11	100
Fis-26902	Elasmobranchs	SSE	*Telatrygon Zugei*	Yes	1	8	7	Lethal	Lethal	Lethal	6	50
Fis-24377	Elasmobranchs	SSE	*Torpedo torpedo*	No	1	8	7	Lethal	Lethal	Lethal	48	400
Fis-161470	Elasmobranchs	SSE	*Trygonoptera galba*	No	1	8	7	Lethal	Lethal	Lethal	111	210
Fis-47420	Elasmobranchs	SSE	*Urolophus bucculentus*	No	1	8	7	Lethal	Lethal	Lethal	113	230
Fis-61406	Elasmobranchs	SSE	*Urolophus orarius*	No	1	8	7	Lethal	Lethal	Lethal	23	50
Fis-54691	Elasmobranchs	SSE	*Urolophus paucimaculatus*	No	1	8	7	Lethal	Lethal	Lethal	20	150
Rep-2666	Sea turtles	ST	*Caretta caretta*	No	1	3	3	Lethal	Pot.lethal	Pot.lethal	0	40
Rep-2941	Sea turtles	ST	*Chelonia mydas*	No	1	3	3	Lethal	Pot.lethal	Pot.lethal	0	95
Rep-4381	Sea turtles	ST	*Dermochelys coriacea*	No	1	3	3	Lethal	Pot.lethal	Pot.lethal	0	2000
Rep-5181	Sea turtles	ST	*Eretmochelys imbricata*	No	1	3	3	Lethal	Pot.lethal	Pot.lethal	0	140
Rep-6936	Sea turtles	ST	*Lepidochelys olivacea*	No	1	3	3	Lethal	Pot.lethal	Pot.lethal	0	95
Rep-8732	Sea turtles	ST	*Natator depressa*	No	1	3	3	Lethal	Pot.lethal	Pot.lethal	0	95

Table S4.3.3: Fifteen species groups and number of species catchable in at least one gear, with proportions of species listed on CITES, CMS, and the IUCN Red List (Thr. = Vulnerable, Endangered, or Critically Endangered, Unknown = Data Deficient or Not Assessed)

Tax group	Subgroup	Code	Count species	Percent of species listed			
				CITES	CMS	Thr.	Un-known
Cetaceans	Baleen whales	BW	10	90.0	80.0	30.0	10.0
Cetaceans	Large oceanic dolphins	LOD	7	100.0	42.9	0.0	14.3
Cetaceans	Oceanic toothed & beaked whales	OCTBW	15	100.0	13.3	6.7	60.0
Cetaceans	Shallow inshore dolphins & porpoises	SINDP	7	100.0	100.0	57.1	0.0
Cetaceans	Small oceanic & coastal dolphins	SOCCOD	12	91.7	66.7	0.0	8.3
Elasmobranchs	Demersal generalist elasmobranchs	DGE	61	0.0	0.0	14.8	29.5
Elasmobranchs	Deep sea elasmobranchs	DSE	50	0.0	0.0	8.0	28.0
Elasmobranchs	Deep shelf pelagic elasmobranchs	DSPE	11	0.0	9.1	27.3	36.4
Elasmobranchs	Inshore elasmobranchs	INE	50	6.0	6.0	36.0	28.0
Elasmobranchs	Oceanic elasmobranchs	OCE	7	85.7	85.7	85.7	0.0
Elasmobranchs	Pelagic filter feeder elasmobranchs	PFFE	9	55.6	55.6	77.8	11.1
Elasmobranchs	Pelagic generalist elasmobranchs	PGE	24	12.5	25.0	54.2	8.3
Elasmobranchs	Reef elasmobranchs	RE	25	4.0	0.0	8.0	8.0
Elasmobranchs	Shallow shelf elasmobranchs	SSE	73	12.3	4.1	32.9	13.7
Sea turtles	Sea turtles	ST	6	83.3	83.3	83.3	16.7

Appendix 5: Supplementary Materials for Chapter 7

Table S5.1: Estimated parameters for parametric coefficients, smoothed effects, and fixed effects for the best model for the 16 species-fishery interactions. Perc. in light = Percent of shot duration in daylight. Depth= depth of fishing gear (either a min or max depth depending on the fishery). All models included an effort offset. "Importance" is estimated by the dredge function.

Shortfin makos (Tuna longlines)						
Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-1.04E+00	7.95E-02	-13.139	< 2e-16	
	Target catch	1.67E-03	2.64E-04	6.321	2.88E-10	1.0000
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	8.9068	8.997	39.188	< 0.001	1.0000
	s(Month)	9.0796	10	43.630	< 0.001	1.0000
	s(Lat, Lon)	1.9941	2	419.223	< 0.001	1.0000
	s(% in light)	8.1137	8.7745	26.015	< 0.001	1.0000
	s(Shot duration)	7.382	8.3485	8.271	< 0.001	1.0000
Fixed Effects						
	Target cluster					1.0000
	Vessel					1.0000
	Operation type					1.0000
Summary stats	**Deviance explained**		**Num. observations**			
	25.30%		4242			

Shearwaters (Tuna longlines)						
Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-5.90E+00	1.08051	-5.462	4.98E-08	
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	2.65	3.3369	46.007	< 0.001	1.0000
	s(Month)	7.8977	10	43.916	< 0.001	1.0000
	s(Lat, Lon)	1.9267	2	67.845	< 0.001	1.0000
	s(% in light)	3.4626	4.2049	23.194	< 0.001	1.0000
	s(Vessel)	57.4526	133	2.714	< 0.001	1.0000
Fixed Effects						
	Target cluster					0.3995
	Operation type					0.9997
Summary stats	**Deviance explained**		**Num. observations**			
	82.50%		4242			

Petrels (Tuna longlines)						
Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**

	Intercept	-9.78901	2.51329	-3.895	9.98E-05	
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	3.8934	4.6854	17.364	< 0.001	1.0000
	s(Month)	5.4927	10	42.038	< 0.001	1.0000
	s(Lat, Lon)	1.4714	2	17.731	0.021	0.5513
	s(% in light)	2.1926	2.63	13.318	< 0.001	0.9984
	s(Vessel)	47.0819	133	1.666	< 0.001	1.0000
Fixed Effects						
	Operation type					0.4212
Summary stats	**Deviance explained**		**Num. observations**			
	84.10%		4242			

Albatrosses (Tuna longlines)

Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-5.0356	0.4426	-11.376	<2e-16	
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	1	1	58.142	< 0.001	1.0000
	s(Month)	5.915	10	6.175	< 0.001	1.0000
	s(Lat, Lon)	1.7986	2	50.694	< 0.001	1.0000
	s(% in light)	2.2447	2.6794	16.752	< 0.001	0.9999
	s(Vessel)	29.4666	133	0.652	< 0.001	0.7531
Fixed Effects						
	Operation type					0.7289
Summary stats	**Deviance explained**		**Num. observations**			
	52.30%		4242			

Sea snakes (Prawn trawl)

Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-5.26E+00	4.45E-01	-11.820	< 2e-16	
	Target catch	-2.33E-05	6.37E-06	-3.660	0.00026	1.0000
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	8.2682	8.8556	34.596	< 0.001	1.0000
	s(Month)	1.8119	2	9.291	< 0.001	0.9897
	s(Lat, Lon)	3.7762	4	17.131	< 0.001	1.0000
Fixed Effects						
	Target cluster					0.9994
	Vessel					1.0000
Summary stats	**Deviance explained**		**Num. observations**			
	84.80%		4377			

Hammerheads (Prawn trawl)

Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-30.8618	24486.875	-0.001	0.999	

Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	8.2671	8.7596	15.592	< 0.001	0.9999
	s(Month)	1.9164	2	19.151	< 0.001	0.9999
	s(Lat, Lon)	3.6897	4	19.478	< 0.001	0.0001
	s(Depth)	6.3773	7.3336	6.246	< 0.001	1.0000
Fixed Effects						
	Vessel					1.0000
Summary stats	**Deviance explained**		**Num. observations**			
	95.40%		4377			

Shearwaters (Demersal longlines)

Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-7.4016	0.7819	-9.466	<2e-16	
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	**Importance**
	s(Month)	1.7652	2	14.247	0.002	0.9647
	s(Vessel)	4.6623	16	1.058	< 0.001	0.9835
Fixed Effects						
Summary stats	**Deviance explained**		**Num. observations**			
	16.10%		1987			

Petrels (Demersal longlines)

Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-1.34E+02	8.10E+01	-1.653	0.0985	
	Target catch	-2.33E-03	9.33E-04	-2.501	0.0125	0.4506
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	0.9543	9	3.998	< 0.001	0.9998
	s(Lat,Lon)	1.8205	2	24.457	< 0.001	0.9959
	s(Depth)	3.9711	9	4.056	< 0.001	1.0000
	s(Vessel)	5.0809	16	3.825	< 0.001	1.0000
Fixed Effects						
	Target cluster					1.0000
Summary stats	**Deviance explained**		**Num. observations**			
	44.20%		1987			

Albatrosses (Demersal longlines)

Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-6.5939	0.8396	-7.854	6.58E-15	
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	1.3132	9	5.613	< 0.001	0.9869
	s(Month)	1.3614	2	2.489	0.045	0.7323
	s(Lat,Lon)	1.8465	2	28.156	< 0.001	0.8856
	s(Depth)	2.1342	9	5.148	0.002	0.9371
	s(Vessel)	6.8429	16	3.645	< 0.001	1.0000

Fixed Effects						
	Target cluster					0.0285
Summary stats	**Deviance explained**		**Num. observations**			
	27%		1987			

Shearwaters (Otter bottom trawl)						
Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-1.23E+01	1.58E+00	-7.772	9.23E-15	
	Target catch	2.59E-04	3.51E-05	7.382	1.80E-13	1.0000
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	5.997	9	10.562	< 0.001	1.0000
	s(Month)	3.8623	10	6.581	0.009	0.6815
	s(Lat,Lon)	1.3596	2	90.511	0.059	0.6507
	s(Vessel)	22.4045	57	1.878	< 0.001	1.0000
Fixed Effects						
Summary stats	**Deviance explained**		**Num. observations**			
	66.80%		5227			

Pinnipeds (Otter bottom trawl)						
Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-9.028192	0.4400382	-20.520	<2e-16	
	Target catch	0.0002452	0.0000241	10.180	<2e-16	1.0000
Smoothed terms	**Term**	**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Month)	2.7034	10	9.234	< 0.001	0.9860
	s(Lat,Lon)	1.8532	2	516.996	< 0.001	0.9941
	s(dpth_min)	0.8767	9	3.201	0.004	0.9766
	s(Vessel)	31.7655	57	2.724	< 0.001	1.0000
Fixed Effects						
Summary stats	**Deviance explained**		**Num. observations**			
	46.30%		5227			

Petrels (Otter bottom trawl)						
Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-9.67E+00	1.27E+00	-7.645	2.48E-14	
	Target catch	2.54E-04	3.85E-05	6.604	4.40E-11	1.0000
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	5.2392	9	46.102	< 0.001	1.0000
	s(Vessel)	26.3911	57	2.375	< 0.001	1.0000
Fixed Effects						
Summary stats	**Deviance explained**		**Num. observations**			
	70.30%		5227			

Albatrosses (Otter bottom trawl)						
Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-6.13E+00	4.90E-01	-12.510	<2e-16	
	Target catch	2.43E-04	2.40E-05	10.100	<2e-16	1.0000
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	4.989	9	41.606	< 0.001	1.0000
	s(Month)	4.8035	10	11.341	< 0.001	0.8462
	s(dpth_min)	3.7249	9	7.847	< 0.001	0.9801
	s(Vessel)	29.1687	57	2.820	< 0.001	1.0000
Fixed Effects						
Summary stats	**Deviance explained**		**Num. observations**			
	51.30%		5227			

Shearwaters (Set gillnets)						
Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-7.8705	0.5727	-13.740	<2e-16	
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	0.565	9	0.144	0.12	0.5301
	s(Month)	1.671	2	4.785	0.003	0.9421
	s(Lat,Lon)	0.5504	2	0.548	0.14	0.4981
	s(Vessel)	0	42	0.000	0.64	0.4880
Fixed Effects						
Summary stats	**Deviance explained**		**Num. observations**			
	12%		2115			

Dolphins (Set gillnets)						
Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-19.97966	2.53933	-7.868	5.72E-15	
	Target catch	-0.04123	0.0137	-3.010	0.00264	0.8492
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	
	s(Year)	0.7419	9	40.488	0.049	0.7946
	s(Lat,Lon)	1.0618	2	247.512	0.012	0.8933
	s(Vessel)	12.3574	42	1.176	< 0.001	0.9938
Fixed Effects						
Summary stats	**Deviance explained**		**Num. observations**			
	72.30%		2115			

Albatrosses (Set gillnets)						
Coefficients	**Term**	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**	**Importance**
	Intercept	-7.4155	0.5437	-13.640	<2e-16	
Smoothed terms		**Est. df**	**Ref df**	***F***	***P*-value**	

	s(Lat,Lon)	1.1383	2	10.939	0.003	0.6839
	s(Depth)	0.9346	9	1.816	0.007	0.9041
	s(Vessel)	12.3066	42	0.560	0.004	0.9564
Fixed Effects						
Summary stats	**Deviance explained**		**Num. observations**			
	20%		2115			