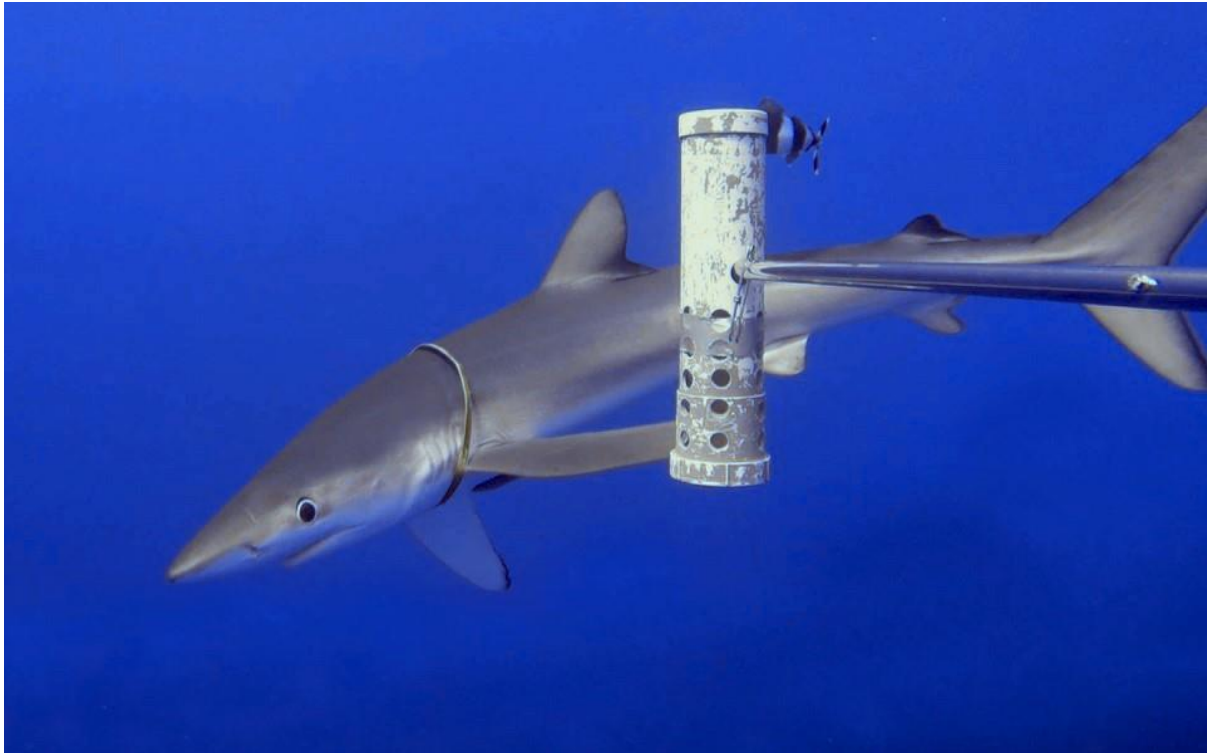


PELAGIC ECOLOGY AND SOLUTIONS FOR A TROUBLED OCEAN

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BA, AO



A blue shark *Prionace glauca* entangled in a plastic box strapping as observed via midwater stereo- BRUVS.



**This thesis is presented for the degree of Doctor of Philosophy of The
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School of Biological Sciences**

2019

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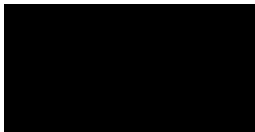
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The following approvals were obtained prior to commencing the relevant work described in this thesis: AU-COM2012-170, AU-COM2018-426, PA2018-00036-1, PA2018-00091-1, PA2018-00091-2, PA2018-00079, DPAW 01-000049-4, DPAW 01-000049-7, DPAW 01-000049-8, CMR-17-000526, CMR-16-000426, CMR-18-000550, and Fisheries Exemption Numbers 2853 and 3172.

This thesis contains published work and/or work prepared for publication, some of which has been co-authored.

Signature:

A solid black rectangular box redacting the signature of the author.

Date: 18/11/2019

Abstract

Our oceans are plagued by the triple threat of climate change, overfishing and their use as a global garbage bin, including for waste plastic. Combined, these threats are driving biodiversity loss at an unprecedented rate, with this period in our planet's history dubbed "the Anthropocene" as it is we humans who are driving the sixth mass global extinction.

Marine protected areas (MPAs) are increasingly seen as essential to delivering biodiversity and fisheries outcomes, while also building ocean resilience. While such outcomes are well documented for coastal MPAs, large offshore MPAs are a relatively recent phenomenon: the world's first fully no-take large offshore MPA was only established in 2010. This recent history, combined with the difficulty and expense of sampling offshore environments, has hampered our ability to understand the benefits of MPAs for pelagic species.

For the first three chapters of this Dissertation (Chapters 2-4), I used mid-water stereo baited remote underwater video systems (BRUVS) to document pelagic fish assemblages across the western region of Australia's Exclusive Economic Zone, with a focus on ten of the region's 18 offshore MPAs. Mid-water stereo-BRUVS were chosen because they are non-lethal and provide a permanent video record that allows diversity, abundance and biomass to be estimated through time and space. In total, 57,634 fishes representing 147 taxa were counted and identified at 15 locations across nearly 23° of latitude.

I used these data to answer a number of conservation-related ecological questions. In Chapter 2, I document the likely presence of a shortfin mako *Isurus oxyrinchus* pupping ground along the Western Australia (WA) coast. Neonates were exclusively found at three locations and nowhere else along the WA coast, nor in a complementary global database. Identifying pupping grounds is important for species such as the shortfin mako given its conservative life history and listing as "Endangered" by the IUCN. In Chapter 3, I demonstrate stability in the spatial distribution of ocean wildlife at the Perth Canyon Marine Park, Western Australia (WA) over a five-year period. Some arguments against offshore MPAs focus on the dynamism of pelagic ecosystems. However, the observed stability through time suggests that MPAs with fixed boundaries can generate benefits for pelagic fishes. In Chapter 4, I document large-scale patterns in species presence and turnover in Australian marine parks located in the eastern Indian Ocean. I identified locations with lower than expected species diversity and abundance, correlated with human impacts. These biogeographical results provide guidance for prioritising improvements to the marine park network, enhancing biodiversity outcomes and providing a baseline against which benefits of protection can be evaluated.

Although I was able to collect a large amount of data for ten Australian National Marine Parks, there are an additional 32 parks within the Australian network for which limited or no pelagic baselines exist. A major step-change in offshore marine sampling is thus required, both inside and outside marine parks. To this end, Chapter 5 of this Dissertation explores the integration of a wide range of technologies around automation, artificial intelligence, and machine learning with existing and cutting-edge methodologies such as mid-water stereo-BRUVS and environmental DNA to envision an offshore research capacity that will transform our ability to understand pelagic ecology.

Finally, the pernicious impacts of waste plastic on the oceans is fundamentally undermining ocean health and does not stop at the borders of marine parks. In Chapter 6, I argue that the “Sea the Future” initiative, a voluntary contribution on fossil-fuel derived polymers, the current building blocks of plastics, will disrupt the current linear plastic economy to create a circular one. This voluntary contribution will create a cashable commodity from plastic waste, shifting fundamental economics and incentivising recovery, and reduce fossil fuel-based polymer production via polymer-to-polymer recycling. By making waste plastic valuable, an entirely new shared economy that redirects plastic waste away from harmful systems of disposal will be unlocked.

Loss of wildlife from the most underexplored region of our planet poses inestimable risks beyond those directly experienced by the directly affected wildlife. The Anthropocene’s impact on our oceans reduces their resilience to climate change, erodes food security and exacerbates social injustice. Our understanding of the ocean must be expanded before irreversible consequences occur, if they have not already occurred. MPAs represent a key strategy to returning our oceans to a flourishing state, particularly when combined with accelerated research and an Amazonian disruption on waste plastic.

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First and foremost, I would like to thank the parents of my supervisor, Professor Jessica Meeuwig - Jaap and Elaine Hummel Meeuwig. They clearly instilled in Professor Meeuwig boundless commitment to explore and excel in her own field of dreams, pursued with huge heart, determination, character depth and grace. When combined with generosity of spirit to share that developing knowledge widely, they gifted in her all the attributes of a great teacher. As Professor Meeuwig's students and graduates exemplify, specialising in her chosen field of science of ocean and marine ecological conservation, this immense character can only wellspring from enlightened and encouraging parenting. It is these instilled values that Professor Meeuwig now enthusiastically and just as firmly, passes on to her students.

Not many professors have the desire or the capacity to approach ocean science in the multidisciplinary fashion that is essential to the conservation of our marine environments, and I am hugely grateful to her for transferring some of that knowledge to me in the four years I have had the honour of being her student. I am also grateful to the amazing Centre for Marine Futures lab at the University of Western Australia, in particular the field-work and in-house technicians – The Fab 5 – Alex McLennan, Adam Jolly, Claire Raphael, Nikki DeCampe, and Vyv Summers, who gave me the honour of hauling in countless tens of kilometres of rope alongside them, as we recovered innumerable underwater camera systems. The laborious thousands of hours of underwater video footage processing helped inspire the fifth manuscript of this Dissertation.

Thank you to former UWA Vice Chancellor and current Forrest Hall Warden Professor Paul Johnson who, after hearing of my lifelong passion to do contribute to healthy oceans, strongly suggested I do my Masters degree or PhD at UWA, when we were immersed deeply in a Leeuwin concert four years ago. I am so glad I took his advice.

I was privileged to have been examined by two marine science global heavyweights, Professor Callum Roberts and Professor Doug McCauley. I am grateful for their peer review and resulting insightful wisdom that, like my supervisors' observations, sharpened this Dissertation.

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It was Professor Dunlop who threw down an initially unwelcome gauntlet to go further than the lengthy initial three chapters of this Dissertation. Indeed, my Examiners have since observed these chapters are sufficient to earn a PhD at their universities. However, this challenge led me directly to both ROMP and Sea the Future manuscripts that helped propel this work out as a solutions-orientated Dissertation. I am of course grateful now to Professor Dunlop for her gauntlet.

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AUTHORSHIP DECLARATION - CO-AUTHORED PUBLICATIONS

This Dissertation is comprised of the following work that has been published or prepared for publication. Variation in presentation reflects journal requirements and editorial policies.

Details of the work:

Forrest JAH, Barley SC, Turner J, Tickler D, Bouchet PJ and Meeuwig JJ. Global distribution of shortfin mako *Isurus oxyrinchus* and possible identification of a first Indian Ocean pupping ground. *Conservation Science and Practices*, in review.

Location in thesis: Chapter 2

Student contribution to work:

I developed the idea with input from JJM, designed and conducted field work, processed imagery, completed the data analysis, drafted and revised the manuscript based on feedback from co-authors, and submitted it to the journal.

Details of the work:

Forrest JAH, Bouchet PJ, Barley SC, Meeuwig JJ. Stable patterns in ocean wildlife support the establishment of marine protected areas. *Ecosphere*, in review.

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Student contribution to work:

I developed the idea with input from JJM, designed and conducted field work, processed imagery, completed the univariate and multivariate data analyses, drafted and revised the manuscript based on feedback from co-authors, and submitted it to the journal. The spatial modelling was contributed by co-author PJB.

Details of the work:

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I developed the idea with input from JJM and DP, designed and conducted field work, processed imagery, completed the data analysis, and drafted and revised the manuscript based on feedback from co-authors.

Details of the work:

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Location in thesis: Chapter 5

Student contribution to work:

I developed the idea with input from JJM and SD. I then pursued a range of discussions commencing in 2019 to better understand how such a vessel would be developed and drafted and revised the manuscript based on feedback from co-authors. Assistance was provided to develop images by a graphic artist under my direction.

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Student contribution to work:

I developed the idea and then pursued a range of discussions commencing in 2019 to better understand how such a voluntary contribution would be developed and drafted and revised the manuscript based on feedback from co-authors.

Details of the work:

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Location in thesis: Appendix 1

Student contribution to work:

I contributed to the interpretation of the slavery data based on my lengthy experience with the Global Slavery Index as well as contributing to writing and editing of the manuscript. JJM developed the idea and DT was largely responsible for the data analysis and initial drafting. The manuscript was revised based on feedback from co-authors including myself.

Student signature

Date:

18 November, 2019

I, Professor Sarah Dunlop, certify that the student's statements regarding their contribution to each of the works listed above are correct.

As all co-authors' signatures could not be obtained, I hereby authorise inclusion of the co-authored work in the thesis.

Coordinating supervisor signature:

Date: 18 November 2019

STATEMENT OF CANDIDATE CONTRIBUTIONS

“If you want to go quickly, go alone. If you want to go far, go together. We need to go far quickly” – Al Gore

This Dissertation contains a General Introduction (Chapter 1), five data chapters (Chapters 2-6) each of which is in the form of a manuscript that is in review (Chapters 2 and 3), about to be submitted (Chapters 4 and 5), or published (Chapter 6), and a General Discussion (Chapter 7). I developed the ideas and hypotheses that underpin this Dissertation with input from my supervisors, Professor Jessica Meeuwig, Professor Sarah Dunlop, and Professor Daniel Pauly and revised the manuscripts with input from other colleagues who co-author the specific chapters.

The Dissertation contains a bracing amount of new data that supports Chapters 2-4 and largely relies on data generated by the Great West Ozzie Transect (GWOT) – a sampling programme within the Marine Futures Lab that has run from the Kimberley in the north to the Recherche Archipelago in the south and commenced in 2013. Of the 29 GWOT surveys, 20 were conducted under the direct auspices of my doctoral research and were funded through a combination of sources including personal funding and grants to the Marine Futures Lab (Ian Potter Foundation, TeachGreen, the National Environmental Science Program, and the Clough Foundation; see Appendix 2, Table A.2. 1). For the first three chapters that depend on empirical data collected from the field, I was directly engaged in survey design, field work, and image analysis, noting that the sheer scale of the work required a large team of field- and lab-based technicians to deliver over 57,000 records of marine life. For Chapter 2, these data were also integrated with those collected under the auspices of National Geographic’s Pristine Seas programme by my fellow lab member and PhD candidate, Mr. Chris Thompson (see Appendix 2, Table A.2. 2).

Chapters 5 and 6 emerged from peer-to-peer discussions during 2019, in which I engaged during 170+ meetings with global business, technology, research and political leaders to better understand how ocean research can be accelerated (Chapter 5; Appendix 6) and the market engaged to address systemic ocean threats (Chapter 6; Appendix 7). These two data chapters were self-funded and directly reflect the integration of my experience in business with my burgeoning knowledge of marine science.

Ocean research, as I outlined in my fifth chapter is time consuming and expensive to conduct. I would thus like to acknowledge that this research was supported by an Australian Government Research Training Program (RTP) Scholarship. Furthermore, my Dissertation harnessed data, the collection of which was supported by National Geographic’s Pristine Seas programme, the Darwin Foundation, the Ian Potter Foundation, the Bertarelli

Foundation, the UK Blue Belt programme, the Australian Academy of Science's Margaret Middleton Award, Teach Green, the National Environmental Science Program – Marine Biodiversity Hub, the Clough Foundation and the Institut de Recherche pour le Développement (New Caledonia, France) (see Appendix 2, Table A.2. 1).

All experimental protocols were approved by the University of Western Australia's Animal Ethics Committee (RA/3/100/1484) and were carried out in accordance with the approved guidelines and under the following Australian and Western Australian permits: AU-COM2012-170, AU-COM2018-426, PA2018-00036-1, PA2018-00091-1, PA2018-00091-2, PA2018-00079, DPAW 01-000049-4, DPAW 01-000049-7, DPAW 01-000049-8, CMR-17-000526, CMR-16-000426, CMR-18-000550, and Fisheries Exemption Numbers 2853 and 3172.

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Chapter 1: General introduction

“We have... to face the established fact that the... fisheries are not only exhaustible, but in rapid and continuous process of exhaustion.”

- from *The Impoverishment of the Sea* by Walter Garstang (1900)

1.1 PELAGIC THREATS AND SOLUTIONS

Our oceans cover just over 70% of the planet’s surface and reach a depth of 10,994 m – equivalent to one’s height above the earth’s surface when looking out the window of a commercial airline at cruising altitude. As such, 99% of the habitable space on our planet lies below the ocean’s surface. Whilst our knowledge of ocean wildlife distribution and ecology comes largely from the sunlit euphotic zone, the top 200 m of the ocean’s depth, increased exploration and research is highlighting hitherto unappreciated diversity and connectivity between the ocean’s surface, mesophotic (30 – 150 m), abyssal (3 – 6 km), and hadal (6 – 11 km) habitats (Jamieson et al. 2010, Danovaro et al. 2014, Stefanoudis et al. 2019).

The existential threats to our oceans are climate change (Hughes et al. 2017), wildlife overexploitation (McCauley et al. 2015), industrialisation (Smith 2000), and plastification (Eriksen et al. 2014, Lebreton et al. 2018). The ocean is currently warming by approximately $0.005^{\circ}\text{C yr}^{-1}$ (Roemmich et al. 2015) and its pH is acidifying at a rate of 0.0019 units yr^{-1} (Doney 2010). The consequences of this global change for ocean wildlife include poleward shifts in species distributions (Cheung et al. 2010), reductions in fisheries yields (Cheung et al. 2013), disruption of ecological phenomena that underpin ecosystem health – the so-called match-mismatch hypothesis (Edwards & Richardson 2004, Lindén 2018) – and expansion of oxygen minimum zones (Deutsch et al. 2015, Breitburg et al. 2018), amongst others. Ocean industrialisation takes the form of seismic activities, expanding installation of infrastructure such as offshore oil and gas platforms, seabed mining, coastal development, and transport-driven chemical and noise pollution (Smith 2000, McCauley et al. 2017).

Plastic pollution is a pernicious threat with more than 15 million tonnes (Mt) of plastics entering the oceans annually at a cost of almost US\$ 2.2 trillion per year in environmental damage and lost ecosystem services (World Economic Forum and Ellen MacArthur Foundation 2017, Beaumont et al. 2019). Increasing leakage of plastic waste to the environment (Lebreton & Andrady 2019) is predicted to generate one tonne of plastic for every three tonnes of fish in the ocean by 2025 (The Ellen MacArthur Foundation 2017). Whilst the obvious forms of plastic pollution include garbage patches in oceanic gyres (Eriksen et al. 2014), entanglement of marine animals and ingestion of plastics (Bernardini et

al. 2018), plastic pollution in the ocean also extends to nano- and microplastics accumulating in the deepest reaches of the ocean (Jamieson et al. 2019) as a permanent record of human wastefulness, irresponsibility and stupidity.

Fishing, whether it be industrial, subsistence, or recreational, is inexorably reducing the diversity, abundance and biomass of wildlife in our oceans, particularly those species that have sustained humanity for millennia (Roberts 2007) and upon which some of the most impoverished communities in the world rely (Kittinger et al. 2015). Reconstructed fisheries data document that humanity hit “peak fish” in 1996 and that the global catches have since been declining (Pauly & Zeller 2016). Indeed, declining catch is occurring despite a continuing expansion of the global fishing fleet (Tickler et al. 2018b). Technological advances and longer-distance fishing capacities effectively mean fish are now exploited at previously unreachable depths, latitudes and longitudes covering some 73% of the ocean’s extent (Kroodsma et al. 2018). Consequences of these fisheries-driven declines in ocean health extend beyond simply the loss of ocean wildlife. The oceans produce protein, via fisheries, on which it has been estimated that half the world’s population relies, and that approximately 40 million are directly, and 160 million indirectly employed, with many times that number of people benefiting from these activities (FAO 2018). The dependency of at least 3 billion people on ocean-derived protein sets up a potential tragedy in terms of food security and raises significant questions around how we ensure ongoing access to seafood among dependent communities. Indeed, seafood is amongst the most-traded food commodities globally, with the World Bank estimating that the “first sale” value of the 171 Mt of fish that were landed in 2016 was worth US\$362 billion (Asche et al. 2015, Gephart & Pace 2015). Declines in ocean wildlife also drive labour abuses, with fishing operations resorting to modern slavery in order to reduce operational costs in a fundamentally non-profitable sector (ILO 2013, Tickler et al. 2018a).

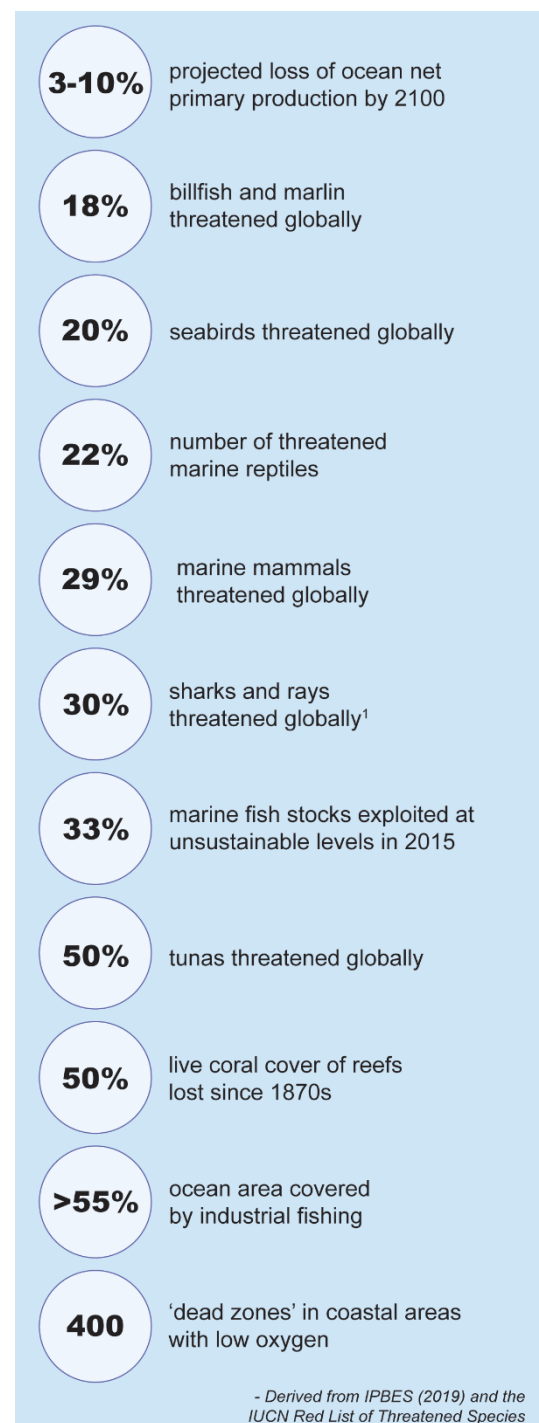
Plastics pollution is ubiquitous in our oceans (Gago et al. 2018, Angiolillo 2019, Barboza et al. 2019) and is increasing in scale in coastal and oceanic waters (Lebreton et al. 2018, Ostle et al. 2019). Marine plastic pollution harms marine megafauna (Galgani et al. 2019) including turtles (Lynch 2018, Wilcox et al. 2018), mammals (Panti et al. 2019), birds (Wilcox et al. 2015) and sharks (Parton et al. 2019), with ingestion and entanglement having fatal consequences (Stelfox et al. 2016, Provencher et al. 2017). Millions of marine animals belonging to more than 260 species die annually due to interactions with visible macroplastics (Thompson et al. 2009, Barboza et al. 2019, Claro et al. 2019). These numbers are likely to increase as smaller and more elusive organisms are examined (Jamieson et al. 2019). Macroplastics also transport invasive species and pathogens between marine regions (Rech et al. 2016, Lamb et al. 2018), inhibit gas exchange between

sea water and seafloor sediments (Goldberg 1997), and smother fragile seafloor inhabitants (Gregory 2009).

Insidiously, marine plastic pollution also includes microplastics, particles <5 mm (Hidalgo-Ruz et al. 2012), and nanoplastics, <1 µm (Gigault et al. 2018), released directly into the environment or created by the fragmentation of larger items. Nano- and microplastics contain dyes, flame retardants and plasticizers, some of which are persistent, bio-accumulative toxins (Rani et al. 2015). These compounds, along with water-borne pollutants absorbed into nano- and microplastics (Engler 2012), can be transferred to organisms upon ingestion (Cole et al. 2011, Neves et al. 2015).

Much like was the case with dichlorodiphenyltrichloroethane, or DDT (Carson 1962), nano- and microplastics accumulate in the food chain to human consumers of seafood. Observed effects of nano- and microplastics on experimental organisms include increased mortality (e.g., Lee et al. 2013, Ziajahromi et al. 2018), disease (e.g. Vasseur and Cossu-Leguille 2006, Chae et al. 2018), inflammation in digestive and respiratory systems (e.g. Murray & Cowie 2011, Jin et al. 2018), increased oxidative stress (e.g. Bhattacharya et al. 2010, Gomiero et al. 2018), disturbed feeding behaviour (e.g. Cedervall et al. 2012, Nasser & Lynch 2016) and compromised fecundity and reproduction (Angielczyk et al. 2015, Sussarellu et al. 2016).

The term “Anthropocene” (Crutzen 2002) was coined to refer to the first geological period of time where the planet has been shaped by a single species, *Homo sapiens*. While the term was first used in the context of oceans in 2001, the link between the Anthropocene and negative effects on marine biodiversity was not fully established until 2008 (Jackson 2008). The threats discussed above are already, and will continue to be, the main challenges to climate regulation, biodiversity maintenance, oceanic ecosystem services, coastal



protection and food security as we progress through the Anthropocene (IPBES 2019). These threats are existential because the oceans are fundamental to basic ecosystem services on the planet and indeed life. Via photosynthesis, the ocean produces at least 50% of the oxygen in our atmosphere (Harris 1986). Its large volume absorbs heat and carbon, regulating the climate and mitigating climactic threats (Siegenthaler & Sarmiento 1993). A substantial proportion of marine species are already threatened with extinction, including 30% of sharks and rays, 33% of reef-building corals and 37% of marine mammals (IUCN 2019 and Box 1). Indeed, a seminal paper by McCauley et al. (2015) warned that the extinction rate in the oceans has been relatively low compared to terrestrial environments not because of the resilience of ocean wildlife or the vastness of the habitat, but simply because humans have been slower to industrialise and pollute the oceans.

1.2 OCEAN WILDLIFE

Here, I define “ocean wildlife” as any animal that inhabits a marine ecosystem, from the intertidal zone to the abyssal plains. To date, humans have primarily interacted with demersal (seabed) and pelagic (water column) ecosystems in coastal waters, however the open ocean and its euphotic zone (<~100 m) have been subject to increasing levels of exploitation in recent decades.

Pelagic wildlife remains poorly understood with respect to its

abundance, distribution, productivity and ecology (Webb et al. 2010). As

such, the establishment of

sustainable levels of exploitation

remains challenging and populations are highly vulnerable to overfishing (Dulvy et al. 2008).

Many pelagic species could be described as “cryptic” because they occur in low numbers and are patchily distributed across vast three-dimensional space. For instance, the

distribution of the longfin mako *Isurus paucus* appears circumglobal (**Fig 1.1a**) but is based

on limited records of its presence (**Fig 1.1b**) and largely relies on what is known about its

depth and temperature preferences (Kai et al. 2017).

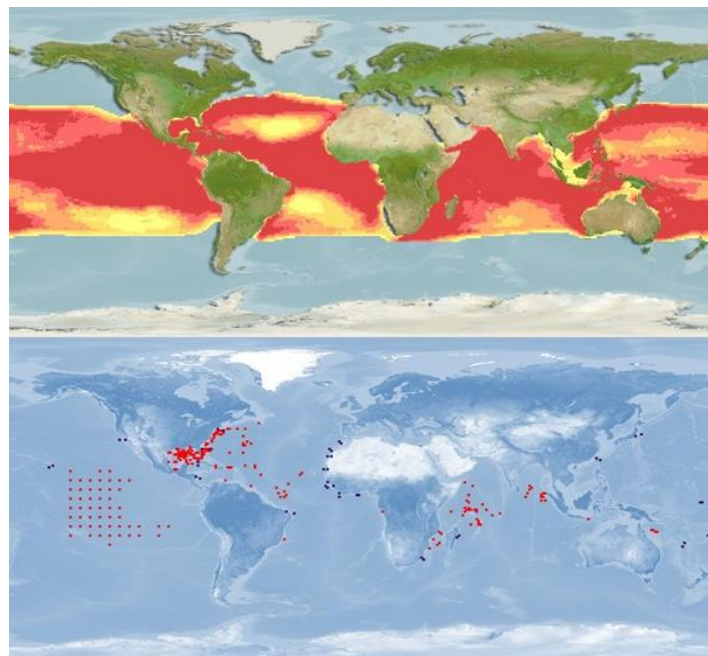


Figure 1. 1 Predicted distribution of the longfin mako *Isurus paucus* based on habitat preferences (top) and existing records that underpin the predicted distribution (bottom) (derived from www.fishbase.de; accessed 30/08/2019).

Biogeographical studies are comparatively few for pelagic sharks (Harrison et al. 2018): only 59 studies were returned for a Web of Science search based on the terms “biogeography”, “shark” and “pelagic or ocean” (accessed 7/08/2019). Habitat use by pelagic species is also relatively poorly understood. In general, pelagic species are considered highly mobile, travelling vast distances: the oceanic whitetip shark *Carcharhinus longimanus*, for example, can undertake migrations of over 14,000 km (Howey-Jordan et al. 2013). These migrations occur along “highways” that we are now beginning to discover (Bruce et al. 2006, Letessier et al. 2017). These highways can lead to locations that support key biological functions such as reproduction and feeding, with the ‘white shark café’ a well-established example in the mid North Pacific Ocean (Boustany et al. 2002). However, pelagic animals can also be residential. For instance, some populations of yellowfin tuna *Thunnus albacares* are less migratory than previously considered, both in the Atlantic (Richardson et al. 2018) and Indian (Govinden et al. 2013) oceans. Our perception of pelagic wildlife as highly mobile may also be an artefact of the tendency of fisheries to preferentially eliminate the more residential individuals in a population, as they tend to be more accessible. It is already recognised that fishing mortality can drive evolutionary changes in growth rates and size at maturity (Law 2000, de Roos et al. 2006), thus it is not unexpected that evolutionary changes in relative mobility may also occur. The reverse of this may also occur, with evolutionary modelling suggesting that the protection from exploitation afforded by large marine parks should lead to genetic selection for more residential individuals within these areas (Mee et al. 2017).

Telemetry has helped advance our understanding of animal movements with implications for their distribution and biogeography (Bruce et al. 2006, Garla et al. 2006, Nasby-Lucas et al. 2019). Chapman et al. (2015) reviewed telemetry studies in 31 species of largely coastal sharks with respect to residency and philopatry. Specific studies such as that by Francis et al. (2019) have challenged the dogma of pelagic species such as the shortfin mako *Isurus oxyrinchus* as basin scale “ocean nomads”. Telemetry studies have also predicted interactions of mobile species with fishing fleets and the conservation and management implications thereof (Queiroz et al. 2016). Moreover, recent advances in tag technology have also led to an improved understanding of vertical habitat use and connectivity between the euphotic and mesophotic zones (Andrzejaczek et al. 2018). However, although many advances have been made in telemetry-derived knowledge of horizontal movements, our overall knowledge of how animals move between oceanic “roadhouses”, which I define as those areas where they consistently return to feed, breed, pup and grow, remains limited, particularly for many threatened species such as the whale shark *Rhincodon typus*, shortfin mako and hammerhead sharks *Sphyrna* sp. (Gallagher & Klimley 2018, McCoy et al. 2018, Nasby-Lucas et al. 2019).

Many shark species such as the shortfin mako experience the “double jeopardy” of their conservative life history (low fecundity, long gestation times, late age of sexual maturity and longevity) (Collette et al. 2011) and high value fins (Clarke et al. 2007). Despite growing recognition of this double jeopardy, exploitation of threatened sharks remains a serious issue (Clarke et al. 2006), and is the subject of multiple strategies for its control, including bans, transport and trade restrictions (de Mitcheson et al. 2018).

1.3 SOLUTIONS

This Dissertation seeks to improve our understanding of the ocean wildlife of the Indian and Southern oceans and explores why and how such wildlife may be conserved. In doing so the proximate threats to ocean biodiversity of overexploitation and ocean plastification are considered as well as proposed solutions to these threats. My three-pronged approach is to explore solutions around: (1) the role of large offshore marine protected areas (MPAs) in biodiversity conservation; (2) acceleration of research to drive evidenced-based policies that promote flourishing oceans; and (3) market mechanisms to eliminate plastic waste.

SOLUTION 1: LARGE MPAS TO PROTECT BIODIVERSITY, SUPPORT FISHERIES RECOVERY AND BUILD OCEAN RESILIENCE

The establishment of MPAs enhances marine biodiversity, supports fisheries and builds ocean resilience. Early and iconic MPAs were established in 1975 on the Great Barrier Reef (Day 2002) and in Cape Rodney-Okakari Point, New Zealand (Silva et al. 1986). In the case of the latter, the work of Dr. Bill Ballantine who proposed in 1971 that closing parts of the ocean to fishing would provide multiple benefits, was transformational (Ballantine 2014). Indeed, this was the beginning of a body of research from hundreds of marine ecologists over 40 years that has convincingly demonstrated that highly protected (IUCN II and stronger) coastal MPAs in result in higher diversity, abundance and biomass relative to fished areas (Polunin & Roberts 1993, Micheli et al. 2004, McCook et al. 2010, Aburto-Oropeza et al. 2011). Highly protected areas generate fisheries benefits, including higher catch rates adjacent to MPAs (Roberts & Polunin 1991, Roberts et al. 2001, Gell & Roberts 2003, Pelc et al. 2010, Kerwath et al. 2013, Hopf et al. 2016). There is also evidence that commercially important species in highly protected areas can generate up to 80% of the target species in adjacent areas open to fishing (Harrison et al. 2012).

Less anticipated was the role that coastal MPAs play in ocean resilience. Specifically, research has demonstrated that areas where no fishing occurs recover faster from flooding, disease and pest outbreaks (Micheli et al. 2012, Bates et al. 2013, Lamb et al. 2015, Mellin et al. 2016). The interactions that create such resilience remain opaque, but myriad contributions suggest that highly protected MPAs preserve intact ecosystems, and top

predators may be more resilient in the face of anthropogenic challenges (Ruppert et al. 2013, Barley et al. 2017). Finally, an extensive body of research demonstrates that full protection (IUCN II and above) provides greater conservation benefits than partial protection (Sciberras et al. 2013, Giakoumi et al. 2017, Sala & Giakoumi 2018). For example, a global meta-analysis by Sala and Giakoumi (2018) found that fish biomass was 670% greater in fully protected reserves than in unprotected areas, but only 183% greater in partially protected reserves. Moreover, fish biomass was fully restored over time in highly protected MPAs, a trend that was not observed in partially protected MPAs (Sala & Giakoumi 2018). Indeed, in some cases, partial protection has been shown to have no conservation benefits compared to areas open to fishing (Lester & Halpern 2008, Aburto-Oropeza et al. 2011, Turnbull et al. 2018). It has been argued that partially protected marine reserves, in which commercial fishing is banned, act as a “beacon” to recreational fishers, leading, counterintuitively, to the biomass of fish being lower in the protected area than outside it (Francour 1994, Denny & Babcock 2004). Moreover, partial protection may be up to twice as expensive to manage than full protection due to increased enforcement costs (Ban et al. 2011).

The benefits of large scale MPAs to pelagic ecosystems have yet to be documented relative to their coastal cousins. This is primarily because few large highly protected MPAs have been created in open ocean pelagic environments and because it is more expensive and challenging to conduct research in offshore ecosystems than in coastal areas (Letessier et al. 2017). The UK government created history when, in 2010, it established the world’s largest, contiguous MPA in which industrial fishing was prohibited (Koldewey et al. 2010). At 644,000 km², the British Indian Ocean Territory Marine Reserve was a brave move to halt fishing in a relatively pristine region that was subject to significant political pressure due to the deportation of Chagossians between 1967-1973 and the establishment of a US military facility on the southernmost atoll, Diego Garcia (Evers & Kooy 2011). Large MPAs have since increasingly been established (WDPA 2019) but suffer from allegations of being “paper parks” (Wilhelm et al. 2014, Edgar et al. 2018), with fishing generally continuing within their boundaries (**Fig. 1.2**). To date, it is estimated that only 7.7% of the surface area of the world’s oceans is protected by any form of MPA, with only 1.2% of Areas Beyond National Jurisdiction “protected” (WDPA 2019). A mere 2.5% of oceans are strictly protected in IUCN II or stronger (WDPA 2019). These rates of protection are well below the political targets of conserving at least 10% of coastal and marine areas by 2020 under the United Nations Sustainable Development Goal 14 (UN 2015) and the Convention on Biological Diversity (Aichi agreement) Target 11, and well below scientific recommendations of 30% of the oceans under high levels of protection from extraction (Barr & Possingham 2013, O’Leary et

al. 2016). The widespread presence of partial protection (IUCN III-VI) and the residual nature of some parks (Devillers et al. 2015) renders evaluating benefits of offshore MPAs challenging, noting that there is great value in remote large MPAs (O’Leary et al. 2018).

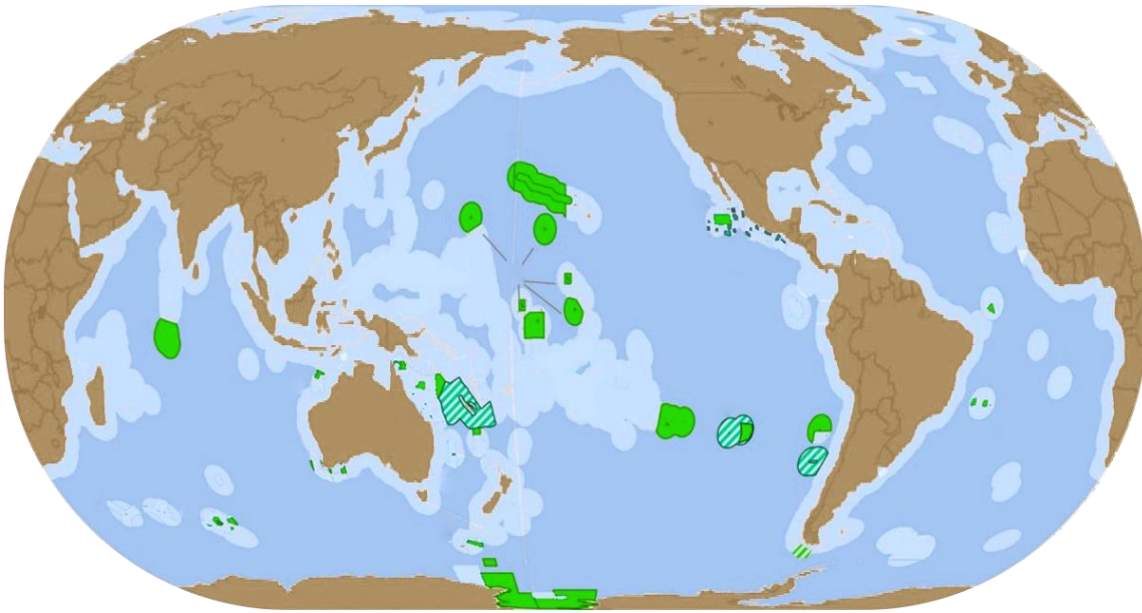


Figure 1. 2 Global map of marine protected areas larger than 100,000 km². No-take MPAs are represented by green. Unimplemented and proposed MPAs are pictured with a crossline pattern. Adapted from original by MPAtlas.org (31/01/2019).

SOLUTION 2: ACCELERATING RESEARCH

The vastness of the oceans has conventionally constrained our ability to generate the necessary knowledge to ensure its protection. The key challenge is to rapidly obtain data, convert data into knowledge, and apply knowledge to change. A significant challenge for the open ocean environment remains how to document the status of the wildlife occupying this habitat. Methods used in this space are reviewed in Letessier et al. (2017) and have conventionally included fisheries data, telemetry and acoustics.

Baited Remote Underwater Video systems (BRUVs) are currently the most widely-used non-destructive method of assessing pelagic fish populations and communities and can provide a wealth of information on the distribution, diversity, abundance, size, and biomass of fish assemblages (Whitmarsh et al. 2017, Letessier et al. 2019). When analysed with respect to environmental data, BRUVS are a powerful tool for examining how populations are distributed and what factors regulate community structure. Traditionally stereo-BRUVS have been used in seabed habitats, where they were first applied in 1995 (Harvey & Shortis 1995). In the 25 years since this time, stereo-BRUVS have answered questions about large-scale biogeography (Cappo et al. 2007, Malcolm et al. 2007), the effects of protection (Espinoza et al. 2014, Malcolm et al. 2015), and how they relate to fisheries data (Langlois et

al. 2012). Stereo-BRUVS have also been deployed at great depth (Priede et al. 2006, Williams et al. 2019). More recently, the method was adapted to the pelagic environment, mid-water BRUVS (Letessier et al. 2013, Letessier et al. 2015). Mid-water stereo-BRUVS have been used to determine correlations between observations from non-destructive stereo-BRUVS and destructive fisheries sampling (Santana-Garcon et al. 2014), for spatial modelling of wildlife distributions (Bouchet et al. 2015) and to identify where are the last refuges of pelagic wildlife globally (Letessier et al. 2019).

More recently, other non-destructive methods such as environmental genomics or “eDNA”, have been developed (Ficetola et al. 2008). Environmental DNA consists of harvesting fragments of genetic material from soil or the water column and sequencing this material. This method has been used to sample community composition (Andruszkiewicz et al. 2017), test for the presence of invasive species (Rees et al. 2014) and explore population genetics (Sigsgaard et al. 2016) and is largely a measure of diversity, rather than abundance, sex, age composition or size (Deiner et al. 2017). Borrowing from cancer research (Hanash et al. 2002), the collection of the whole genome of individuals, an approach I refer to as “eCell”, has the potential to revolutionise how and what we can measure with respect to ocean wildlife abundance and distribution. By virtue of being able to sequence whole genomes, eCell analysis may allow measurement of the abundance of species (i.e. the number of individuals detected in water samples), their age, sex and reproductive condition (Jarman et al. 2015).

Despite their limitations, stereo-BRUVS and eDNA have revolutionised our understanding of patterns in marine diversity (Stat et al. 2018). Significant advances have also been made in remote sensing, drones, ROVS, multibeam sonar and artificial intelligence and machine learning technologies to analyse massive and complex datasets and signal a capacity to create a major step-change in how we sample, and consequently understand, our oceans.

SOLUTION 3: MARKET MECHANISMS TO ADDRESS THE SYSTEMIC OCEAN CHALLENGE OF PLASTIC POLLUTION

The pollution of our oceans by waste plastic is now well recognised. Indeed, in studying pelagic ecology and the role of MPAs, I frequently came across the argument that excluding fishing does not protect fish from plastic pollution (and climate change) as a justification for not establishing MPAs. While it is increasingly recognised that MPAs build resilience (Micheli et al. 2012, Bates et al. 2013, Lamb et al. 2015, Mellin et al. 2016), it is clear that the flow of plastic waste to the ocean must be stopped. However, to date, no systemic solutions have been implemented to deal with this global scourge at the required scales. Strategies to date have included: (1) recycling and the creation of waste management infrastructure; (2) clean-

up programmes both prior to entry into the ocean and direct removal from the ocean; (3) marketing campaigns to educate and change consumer habits; (4) bans on items such as single use plastic bags and water bottles by corporations (“Supermarket ban sees 80pc drop in plastic bag consumption nationwide, retail association says” ABC (2018), “QANTAS operates world's first zero waste flight” QANTAS (2019)), cities (“The San Francisco Bag Ban” 1 bag at a time (2019)), and countries (“Kenya brings in world's toughest plastic bag ban” ABC (2017), “Vanuatu moves to ban more single-use plastic products” ABC (2019)). These strategies have relied on a combination of government and business initiatives, with significant input from civil society including via the philanthropic community. Despite these efforts, plastic continues to flow into our oceans. Moreover, policy changes such as those instigated by China, Malaysia and the Philippines, with other countries following suit, where plastic waste is rejected from exporting countries such as Australia, Canada and the US, have created an industry- and sovereign-scale bottleneck of unmanaged garbage. Finally, innovation has been squelched such that this bottleneck has evaporated the remaining value of some of the plastic in the garbage and eliminated any remaining incentive price for new technology to source waste plastic as its feed-commodity.

1.4 AIMS OF RESEARCH

Ocean management faces a number of challenges and solutions are necessary. Globally, peak fish catch was reached in 1996 with annual declines since on the order of 1 billion kg per year (Pauly & Zeller 2016). The Food and Agriculture Organisation of the United Nations estimates that over one third of global fisheries are overexploited (FAO 2018) and as of 2010, two-thirds of open ocean fish stocks that are managed by regional fisheries management organisations are either “depleted” or “overexploited,” largely due to rampant illegal fishing and the challenges associated with monitoring such immense areas (Cullis-Suzuki & Pauly 2010). Such declines represent a significant challenge for ocean managers that are often hamstrung by limited resources and limited data. Whilst there are some success stories in fisheries management, declining fish populations in many parts of the world suggest that conventional fisheries management is struggling in its aim of halting and reversing global declines in ocean wildlife (Bundy et al. 2008, Cullis-Suzuki & Pauly 2010).

In response, large (>10,000 km²) MPAs are increasingly being established to protect marine biodiversity, support fisheries and build resilience. However, as large MPAs have only recently been established, with the first large no-take MPA created in the British Indian Ocean Territory in 2010 (Koldewey et al. 2010), there is a need to better understand the role that they can play and the ecology of the habitats in which they are placed. Moreover, it is

also necessary to accelerate research given the nature of the Anthropocene and to halt the flow of plastic waste to the oceans.

As such, the goal of this PhD Dissertation is to expand our understanding of pelagic wildlife ecology and propose solutions around: (1) protection; (2) accelerated research; and (3) the eradication of plastic pollution.

The key questions that my PhD will address are:

- (1) Can we identify open ocean habitats critical to endangered species and thus in need of protection? This question is addressed in Chapter 2:

Forrest JAH, Barley SC, Turner J, Tickler D, Bouchet PJ and Meeuwig JJ. Global distribution of shortfin mako *Isurus oxyrinchus* and possible identification of first Indian Ocean pupping ground. *Conservation Science and Practices*, in review.

- (2) Can we identify ocean “roadhouses” – those locations that a range of wildlife consistently utilise, where protection is warranted and effective? This question is addressed in Chapter 3:

Forrest JAH, Bouchet PJ, Barley SC, Meeuwig JJ. Stable patterns in ocean wildlife support the establishment of marine protected areas. *Ecosphere*, in review.

- (3) What are the large-scale patterns of ocean biodiversity in the eastern Indian Ocean and is this region adequately protected? This question is addressed in Chapter 4:

Forrest JAH, Pauly D, McLennan AG, Meeuwig JJ. Human impacts mask natural patterns in pelagic biogeography. *Global Ecology and Biogeography*, in prep.

- (4) Given that nations have an economic and conservation interest in their marine Exclusive Economic Zones (EEZ), and that all nations have an interest in having healthy and flourishing High Seas, how can ocean habitats and wildlife be accurately measured and a baseline established before permanent destruction takes place. This question is addressed in Chapter 5:

Forrest JAH, McAuley RB, Munier R, Bunce M, Madin LP, McLennan AG, Dunlop SA, Pauly D, Meeuwig JJ. The Revolutionising Ocean Measurement Project driving a return to flourishing oceans. *Conservation Letters*, in prep as invited contribution.

- (5) Are there market solutions that can halt the pollution of oceans by waste plastic? This question is addressed in Chapter 6:

Forrest JAH, Giacobazzi L, Dunlop SA, Reisser J, Tickler D, Jamieson A, Meeuwig JJ. Eliminating plastic pollution: how an industry-led voluntary contribution will drive the circular plastics economy. *Frontiers in Marine Science*

<https://doi.org/10.3389/fmars.2019.00627>

1.5 ADDITIONAL INFORMATION

I have also included eight appendices in this Dissertation.

Whilst on this PhD journey, I have also connected my previous work on the prevalence of modern slavery (GSI 2016, 2018) to issues of low profitability in fisheries that drive labour abuses. The reliance on slavery in some fisheries due to low catch is part of the compelling argument for transforming ocean management with the establishment of MPAs. This work (Tickler et al. 2018a), which I co-authored, is published in *Nature Communications* and included here as Appendix 1:

Tickler D, Meeuwig JJ, Bryant K, David F, Forrest JAH, Gordon E, Larsen JJ, Oh B, Pauly D, Sumaila UR, Zeller D (2018). Modern slavery and the race to fish. *Nature Communications* 9(1), 4643. <https://doi.org/10.1038/s41467-018-07118-9>

Appendix 2 provides a summary of the expeditions that generated the data that underpins Chapters 2 to 4. Appendix 3 provides maps for each location showing the position of the sampling units that underpin the generation of predictor variables used in Chapter 4. Also relevant to Chapters 3 and 4, Appendix 4 provides information on the life history and conservation status of the taxa that were observed in the study, their conservation status and the taxon-specific coefficients used to calculate weight based on recorded fork lengths. Some individuals could not be identified to species and were instead allocated to a genus or a family. Appendix 5 thus provides detailed information on the species pool to which an individual could be allocated depending on the location it was observed and the distribution and depth range of congeners and confamilials. Appendices 6 and 7 list the positions and organisations of those individuals with whom I held discussions for Chapters 5 and 6 respectively. Appendix 8 is comprised of the two articles that I co-authored for *The Conversation* as part of contributing to the public communication of research.

1.6 SUMMARY

The marine environment of Australia's Exclusive Economic Zone adjacent to Western Australia has a rich ecology extending from the tropics to temperate waters and two oceans, the Indian and the Southern. It has a significant need for more research in order to generate wildlife baselines and understanding so that effective conservation and sustainable fisheries management practices can be applied, with mid-water stereo-BRUVS providing an effective method to do so (Santana-Garcon et al. 2014, Bouchet & Meeuwig 2015).

Complacency about our understanding of the ocean and its resilience dates to Pliny the Elder in his *Historial naturalis* (AD 77), in which he stated, "By Hercules, in the sea and in the

ocean, vast as it is, there exists nothing that is unknown to us, and, a truly marvellous fact, it is with those things that nature has concealed in the deep that we are best acquainted!". The Dutch jurist Hugo Grotius (Russ & Zeller 2003), typical for his time, was also complacent and strategic as he established the doctrine of *Mare Liberum* in 1609 on the grounds that "...the sea is common to all, because it is so limitless that it cannot become a possession of any one, and because it is adapted for the use of all, whether we consider it from the point of view of navigation or of fisheries" (Grotius 1609). The sea is clearly endangered by extinction rates of ocean species now rivalling those of terrestrial fauna as a function of ocean industrialisation (Smith 2000, McCauley et al. 2015). Conventional fisheries management has largely struggled to ensure sustainability, both nationally and regionally, and plastics are accumulating in our oceans at unprecedented rates.

This Dissertation will focus on solutions to existential threats. In this context, I will also explore the fundamental relationship between science and business. Exploratory and fundamental research has often been considered as adversarial to objectives of international business, and, particularly in the recent political era, has led to science being viewed as an ideological vessel, often portrayed as political in nature and anti-business (Parsons et al. 2015). Conversely, there is also common belief in society, and within some sectors of the scientific community, that the global marketplace and business are often hostile to the environment. However, recognising that science is necessary but not sufficient to solve our oceans' challenges such as the "wicked problems" of overexploitation and plastic waste, partnerships between science and business are essential. Time is of the essence and we must seek both knowledge and solutions through such partnership if we are to go far, quickly.

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CHAPTER 2: GLOBAL DISTRIBUTION OF SHORTFIN MAKO *ISURUS OXYRINCHUS* AND POSSIBLE IDENTIFICATION OF FIRST INDIAN OCEAN PUPPING GROUND

Keywords: marine protected areas, BRUVS, pelagic, Perth Canyon, prey and predators, risk, sharks, behaviour

2.1 ABSTRACT

Large pelagic animals are threatened globally due to their high fisheries value and K-selected life histories, with reductions in biomass as high as 95% for some species of shark. Fundamental to their population recovery is the identification and protection of pupping grounds. The shortfin mako shark *Isurus oxyrinchus* has been assessed as “Vulnerable” since 2011 and “Endangered” since 2018 on the International Union for Conservation of Nature’s (IUCN) Red List, yet population numbers have continued to deteriorate. However, in 2019, the species gained global protection from fishing and trade via the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Although the shortfin mako shark *Isurus oxyrinchus* has a circumglobal distribution, and fisheries and tagging surveys have recorded both neonates and juveniles in the Atlantic, Pacific and Southern oceans, pupping grounds have yet to be identified in the Indian Ocean. We documented the diversity, abundance and size of pelagic animals from 3,605 mid-water stereo-baited remote underwater video samples collected globally between 2012 and 2018 at 27 offshore locations and over 36 expeditions. Sixteen shortfin mako were observed at nine of the locations across 14 expeditions, and individuals ranged in fork length (FL) from 59 cm to 278 cm. Three young-of-the-month (YOM) and two young-of-the-year (YOY) with black nose spots and fork lengths less than 100 cm FL were observed at three locations in the eastern Indian Ocean, all off the coast of Western Australia: Shark Bay (n=1), the Perth Canyon (n=3) and Geographe Bay (n=1). While a formal definition for pelagic shark pupping grounds does not exist, the Perth Canyon satisfies key criteria used to identify coastal shark pupping grounds: (i) YOY are more common at the site than at other locations; (ii) conspecific adults are absent; (iii) food is relatively abundant; and (iv) relatively warm temperatures are present. Our results, in conjunction with a tagging study indicating that the Perth Canyon is visited by female shortfin makos, suggest that this location is a potential pupping ground and thus a critical habitat for the reproductive fitness of the species.

2.2 INTRODUCTION

Pelagic sharks are amongst the most endangered vertebrates globally, with over 50% of species listed as “threatened” by the International Union for Conservation of Nature (IUCN) (Dulvy et al. 2003). Pelagic shark species are prone to “double jeopardy”, being both K-selected and commercially valuable (Collette et al. 2011) and population declines have the potential to alter the structure and function of pelagic food webs, particularly in light of their relatively simple trophic structure (Baum & Myers 2004, Bornatowski et al. 2017; but see Grubbs et al. 2016). Despite these declines, relatively little is known about the breeding behaviour of most pelagic shark species, with the location of pupping grounds typically inferred based on the migratory patterns of sexually mature females (Howey-Jordan et al. 2013, Campana et al. 2015). Yet juvenile survivorship is critical to the recovery of overexploited shark populations (Heithaus 2007) and there is therefore an urgent need to identify and protect potential pupping grounds and understand ontogenetic behaviour.

Efforts to identify shark pupping grounds have traditionally focused on coastal species (Garla et al. 2006, DeAngelis et al. 2008, Chapman et al. 2009, Oh et al. 2017), providing a useful framework to identify pelagic pupping grounds (Vélez-Marín & Márquez-Farías 2009).

Accepted definitions of coastal pupping grounds identify four criteria: (1) Young-of-the-year (YOY) are more common at the site than in other areas; (2) YOY have been recorded over multiple years at the site; (3) sharks tend to remain or return to the site for extended periods of time; and (4) the site provides environmental advantages relative to other areas, ranging from improved protection from predators (i.e. an absence of larger individuals or more physical refugia), prey availability, higher productivity and optimal temperature ranges (Heupel et al. 2007). Evidence also suggests that ontogeny influences behaviour across taxa with younger animals showing greater boldness (Mata et al. 2013, Sherratt & Morand-Ferron 2018). As such, there is a risk that bold behaviour by young animals in unprotected pupping grounds can lead to disproportionate mortality (Andersen et al. 2018) and thus population declines where young animals fail to mature and reproduce.

The shortfin mako shark *Isurus oxyrinchus* is listed as “Endangered” by the IUCN Red List due to its relatively late sexual maturation and low fecundity (Cailliet et al. 2017), with recent evidence suggesting that fishing mortality rates may be ten-fold higher than those reported by fisheries (Byrne et al. 2017). While juvenile and adult shortfin makos are known to show residency (Corrigan et al. 2018, Byrne et al. 2019, Francis et al. 2019), relatively little is known about the early life stages of shortfin makos nor their behaviour. Newborn, young-of-the-month (YOM) shortfin mako pups are 60-70 cm in total length (TL) while individuals between 100 and 120 cm TL are typically considered YOY (Bustamante & Bennett 2013).

Higher than expected frequencies of small shortfin mako have been recorded. For instance, on the Atlantic Ocean's Grand Banks, a tagging study recorded YOM but no large sized individuals within the warmer Gulf Stream but not in adjacent areas, a reproductive strategy that may enable wider dispersion of neonates and reduce the likelihood of predation by large sharks (Casey & Kohler 1992). In the Eastern Pacific Ocean, offshore of Chile, YOM and an absence of adults have also been recorded in fisheries catches, an occurrence that was attributed to optimal temperature ranges (~18-22 °C) and prey richness in that region (Bustamante & Bennett 2013). Further north, YOM were recorded in fisheries catches in Baja California and the Southern California Bight but in low numbers (Vélez-Marín & Márquez-Farías 2009).

An improved understanding of the location of breeding grounds and behaviour is urgently needed for the shortfin mako. We used mid-water stereo-baited remote underwater video systems (stereo-BRUVS) to globally document the abundance, size, biomass and behaviour of pelagic species. Mid-water stereo-BRUVS are particularly well-suited to identifying pupping grounds for shortfin mako given that they are deployed on mid-outer continental shelf habitats at a depth of 10 m, where YOY are most likely to occur (Branstetter 1990). Based on previous definitions of shark pupping grounds and studies that have reported shortfin mako pupping grounds (Branstetter 1990), we hypothesized that shortfin mako YOY would occur on continental shelf areas associated with (1) prominent bathymetric features, (2) warm currents providing an optimal temperature range, (3) oceanographic fronts and/or high productivity and (4) relatively high densities of potential prey such as small (<40 cm) scombrids and carangids. We also hypothesized that potential pupping grounds would be characterised by (5) strong size segregation, such that adult shortfin mako and indeed other large predatory shark species would be either rare or absent if YOY were present. We also use the stereo-BRUVS footage to compare behaviour in small and large shortfin mako, hypothesizing that that smaller individuals will display bolder behaviours.

2.3 METHODS

2.3.1 MID-WATER STEREO-BAITED REMOTE UNDERWATER VIDEO SYSTEMS

Mid-water stereo-BRUVS are an adaptation of seabed stereo-BRUVS (**Supplementary Fig. 2.1**), a well-established method used to document reef fish assemblage structure (Cappo et al. 2006), and represent a standardised and non-destructive sampling method to characterise populations of rare, highly mobile pelagic species (Bouchet & Meeuwig 2015, Letessier et al. 2017) in addition to behaviour (Turner 2016). Mid-water stereo-BRUVS consist of a rig comprised of a central, 1.45 m-long stainless steel frame that supports two high definition action cameras. The frame is mounted on a cross bar 95 cm in length,

perpendicular to a bait arm (180 cm). The cameras on the frame are 80 cm apart, inwardly converging at an angle of 8 degrees. Each camera is set to record in medium field of view to maximise the area captured in the video frame and improve detection rates up to a distance of ca. 10 m. The rigs are suspended at a depth of 10 m for a minimum of 120 minutes, and deployed in sets of five units, with individual units separated by 200 m of line. Each rig is attached to a 45 cm long perforated PVC pipe bait chamber that contains 1 kg of pilchards *Sardinops sagax*. The container ensures the slow release and diffusion of bait into the water column throughout the duration of the deployment and acts as a rudder to minimise rotational movement of the unit, such that a down-stream field of view is maintained.

2.3.2 DATA COLLECTION

Stereo-BRUVS were deployed during 35 expeditions at 23 locations (n=3,605 deployments) between 2012 and 2018 (**Supplementary Table 2.1**). All surveys occurred within the distribution of the shortfin mako (61°N to 56°S; Froese and Pauly 2019). All sampling was undertaken during daylight hours between 7:00 and 17:00 to minimise any effects of crepuscular behaviour in fish (Axenrot et al. 2004, Birt et al. 2012). We used a stratified random sampling approach or generalised random tessellation stratified (GRTS) approach (Stevens & Olsen 2004), depending on the purpose of the specific survey. Individual stereo-BRUVS were deployed in long-line “strings” of three or five rigs, separated by 200 m of line for a minimum of two hours soak time or in moored sets of five, depending on the vessel size and conditions (**Supplementary Table 2.1**). Surveys were conducted from multiple vessels under UWA ethics permit RA/3/100/1484, and in the case of private vessels, under exemptions from the Australian Marine Safety Authority (EX2016/A185; EX2017/A007, respectively). All required jurisdictional permits were obtained.

2.3.3 DATA PROCESSING AND TREATMENT

Stereo-BRUVS were calibrated prior to field work in an enclosed pool, using software CAL (SeaGIS Pty Ltd) and following standard protocols that allow length measurements (Harvey & Shortis 1998). Videos were imported into the EventMeasure software package (SeaGIS Pty Ltd <http://www.seagis.com.au/>) for processing. Video was synchronised in the lab based on a slow hand clap recorded immediately before deployment of each unit in the field. Processing commenced once the stereo-BRUVS had stabilised at a depth of 10 m and ran for a total of 120 minutes. All observed individual animals were identified to the lowest taxonomic level possible, with relative abundance estimated as the maximum number of individuals of a given species in any frame (Cappo et al. 2006); fork length (FL) was measured in stereo.

The dataset was then filtered to extract all records for shortfin mako. Size could not be directly measured from stereo imagery for two individuals. To estimate their length, we first measured the length between the tip of the snout and the first gill slit (head length) on all individuals for which measurements were possible and then calculated the ratio between head and fork lengths. As the dimensions of the bait canister were known, we then estimated head length for the two sharks with unknown fork lengths at the point that the individual's head was in contact with the bait canister. Estimated head length was then used to determine fork length. Shortfin mako are considered YOM at sizes less than approximately 70 cm TL and YOY at sizes between 100 and 120 cm TL (Bustamante & Bennett 2013). We converted TL to FL based on the equation: $FL = -1.71 + TL \cdot 0.929$ (Casey & Kohler 1992) and based on FL, individuals were assigned to two classes: YOM (<75 cm FL); YOY (75 - 110 cm FL) and juveniles or adults (>110 cm FL).

2.3.4 PREY AND PREDATORS

Shark predation is constrained by gape size and varies between 35% and 40% of fork length (Barley et al. in press). To this end, we identified potential prey and potential predators for YOM and YOY shortfin mako as individuals smaller than 40 cm FL and greater than 200 cm FL (excluding baleen whales) respectively; individuals between 41 cm and 200 cm were classified as "neither". The classification was based on observed lengths or potential prey and predators rather than reported maximum or common lengths. The abundances of potential prey and potential predators were calculated for each survey based on the mean abundance per string or set of moored rigs to maximise independence of samples, with the average of these values calculated for the survey. The percentage of potential prey and predators was then determined relative to total abundance of all individuals to control for differences among locations in total abundance. A one-tailed t-test was used to determine whether a location identified as a potential pupping ground differed from non-pupping grounds with respect to the percentage of potential prey and the percentage of potential predators (Zar 1999).

2.3.5 BEHAVIOUR

All video records of shortfin mako were analysed in terms of behavioural events based on an ethogram developed from a combination of previously defined shark behaviours (Supplementary Table 2.2). Behaviours included entries into field of view, direct interactions with the bait canister and indirect behaviours and were tallied for each individual. The frequencies of four behaviours (direct and indirect behaviours, approaches and nictitating) were examined as they are predicted to provide substantial information about boldness (Turner 2016). Finally, three time-based behavioural metrics were calculated. Differences in

behavioural counts between the two size classes were tested using chi-square contingency tests (Zar 1999). The nonparametric Mann Whitney U test was used to test for differences in ranked timing metrics between the two size classes (Zar 1999).

2.4 RESULTS

2.4.1 ABUNDANCE AND SIZE OF SHORTFIN MAKO

In total, 40,564 individuals of fishes (93% including rays), sharks (5.9%), invertebrates (0.7%), marine mammals (0.2%), and marine reptiles (0.1%) were observed across the 3,605 samples over 36 surveys and 24 locations. These individuals represent 232 taxa and 60 families. We recorded 16 shortfin mako at nine of the 23 locations (**Fig. 2.1**). Shortfin mako ranged in fork length from 57.2 cm to 381.8 cm FL (**Table 2.1**). Based on the size thresholds of Bustamante and Bennett (2013) and the presence of black nose spots, we identified three YOM individuals (57.2 – 69.5 cm FL) and two YOY individuals (96.1 and 97.1 cm FL). All five of the YOM and YOY individuals were observed on the West Australian coastline with three at the Perth Canyon, one at West Dirk Hartog Island and one at Geographe Bay. At the Perth Canyon, where there were multiple makos observed, the YOM and YOY individuals were generally near the canyon head with the exception of the smallest individual who was further seaward at greater seabed depth (**Fig. 2.2**). The two YOM and one YOY were all observed in the Austral spring of 2016 whilst a single large female (222.8 cm FL) was observed in the Austral autumn of 2013 (**Table 2.1**). Notable is that when YOM and YOY were present, the sea surface temperature (SST), derived from the NASA Multi-scale Ultra-high Resolution (MUR) data (https://podaac.jpl.nasa.gov/Multi-scale_Ultra-high_Resolution_MUR-SST) was lower than that when the female was observed, but still higher than regional temperatures at the time (**Fig. 2.2**).

In Geographe Bay, a single YOY was observed at the same time of year, the Austral summer, as a juvenile female (163.9 cm FL), albeit not concurrently as the records were from two years, 2018 and 2017, respectively. West of Dirk Hartog Island, a single YOM shortfin mako was observed in the dry season, corresponding nominally to the Austral spring; this was the smallest animal (**Table 2.1**). No YOM or YOY were observed beyond Australia's west coast, with 11 juveniles and adults recorded elsewhere, eight of which were outside Australia's EEZ (**Table 2.1**).

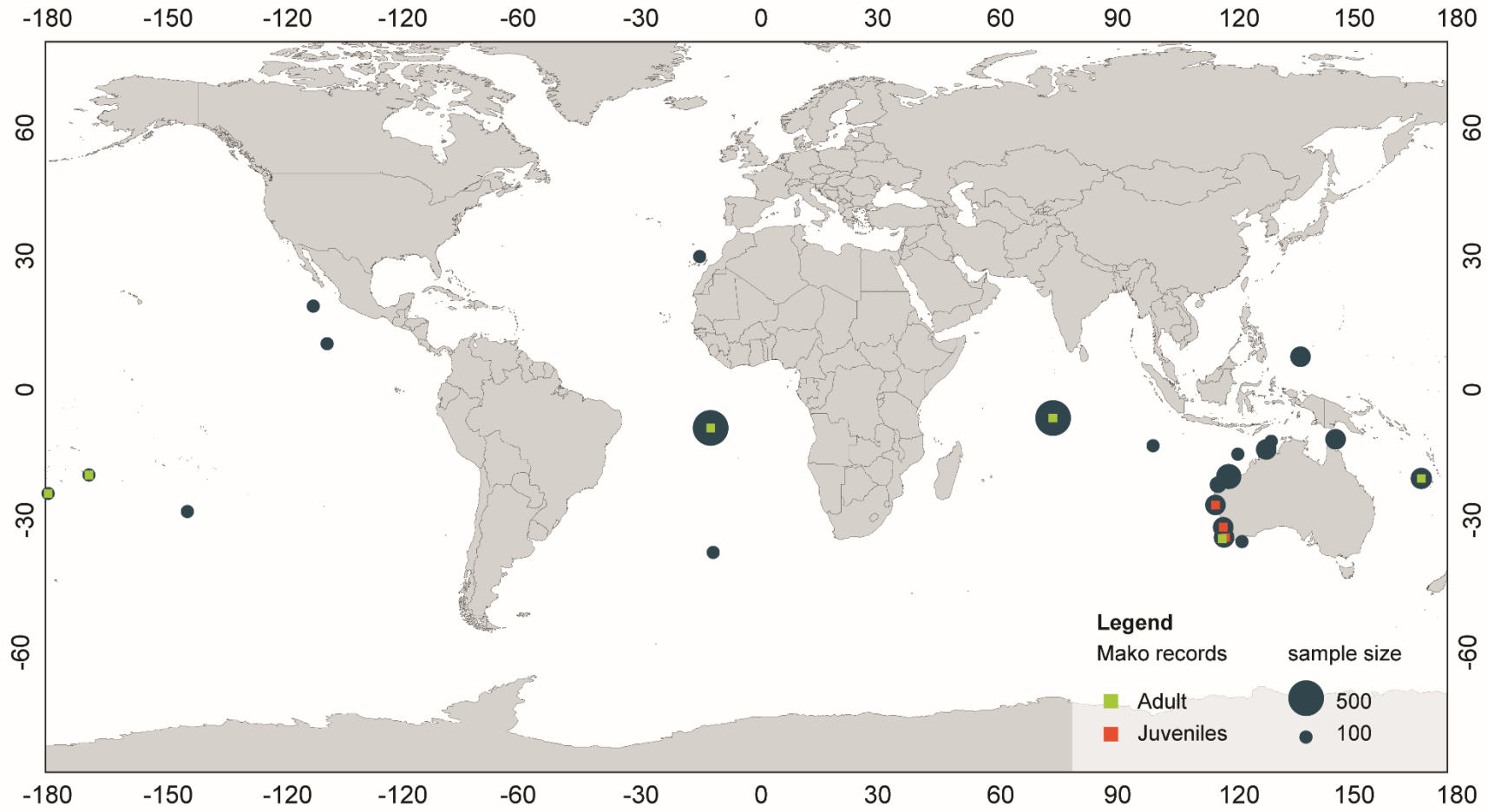


Figure 2. 1 Locations where small (< 110 cm; red), large (>110 cm; green) or no (grey) shortfin mako *Isurus oxyrinchus* were recorded, with sampling effort indicated by diameter of the location point.

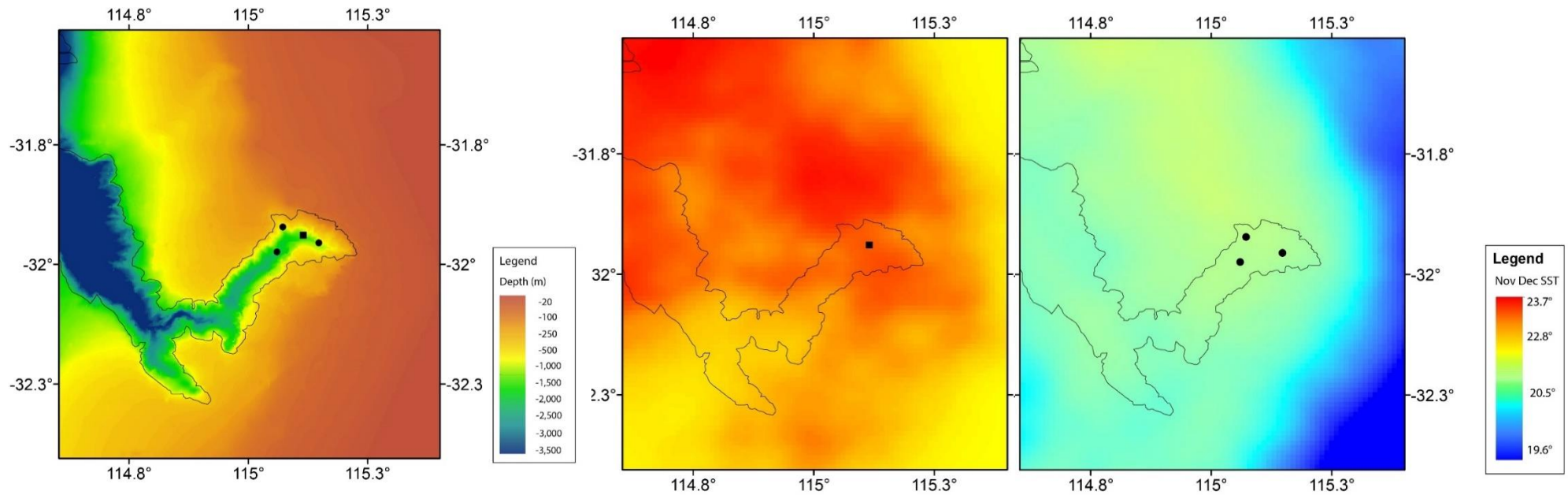


Figure 2. 2 Map of Perth Canyon indicating locations of small (< 110 cm; circle) and large (>110 cm; square) shortfin mako *Isurus oxyrinchus* relative to bathymetry (left) and sea surface temperature (middle, right) at the time of observation (centre and right). The bathymetry is based on the Australian Bathymetry and Topography Grid, June 2009 (<https://ecat.ga.gov.au/geonetwork/srv/eng/catalog.search?node=srv#/metadata/a05f7892-fae9-7506-e044-00144fdd4fa6>) and the SST is from NASA (https://podaac.jpl.nasa.gov/Multi-scale_Ultra-high_Resolution_MUR-SST).

Table 2. 1 Location (decimal degrees), season, fork (FL) and total length (TL), sex/status and date of observation for all observed shortfin mako *Isurus oxyrinchus*.

Location	FL (cm)	TL (cm)	Sex/Status	Latitude	Longitude	Date
West Dirk Hartog Island II - Dry	57.2	59.7	YOM	-26.43293	113.16209	19/09/2017
Perth Canyon III - Spring	60.2	63.0	YOM	-31.95510	115.14820	2/12/2016
Perth Canyon III - Spring	69.5	73.0	YOM	-31.97380	115.06000	25/11/2016
Geographe Bay II - Summer	96.1	101.6	YOY	-33.40280	115.18866	10/02/2018
Perth Canyon III - Spring	97.1	102.7	YOY	-31.92210	115.07240	30/11/2016
Gascoyne - Dry	144.8	154.0	Female	-21.92940	113.26610	18/09/2016
Tonga - Dry	157	167.2	Female	-23.82940	-179.16940	11/07/2013
BIOT I - Wet	157.8	168.0	Female	-5.38645	72.30937	3/12/2012
Geographe Bay I - Summer	163.9	174.6	Unknown	-33.54394	115.31281	4/02/2017
BIOT I - Wet	183.7	195.9	Female	-5.20513	72.17417	7/12/2012
New Caledonia II - Dry	194.4	207.4	Male	-19.63300	163.30770	28/09/2014
Perth Canyon I - Autumn	222.8	238.0	Female	-31.9375	115.114	1/04/2013
Ascension I - Summer	224.1	239.4	Male	-6.13255	-11.65129	3/02/2017
Ascension II - Autumn	239.4	255.9	Male	-9.73355	-12.08791	31/05/2017
BIOT I - Wet	361.3	387.1	Male	-5.16288	72.15462	6/12/2012
Niue - Dry	381.8	409.1	Male	-20.07710	-167.76930	2/10/2016

Table 2. 2 Total counts of observed behaviours in shortfin makos *Isurus oxyrinchus* and the percentage of behaviours displayed by sharks smaller than 110 cm FL (%Small).

Behaviour	Count	% Small	χ^2 †	
Entries	705	59.3	122.4	***
Rapid withdrawal	3	100	no test	
Approach	173	44.5	7.8	**
Patrolling	29	37.9	0.4	ns
Charge	4	100	no test	
Jaw gaping	74	91.9	59.5	***
Bite	98	88.8	70.1	***
Nudge	83	84.3	50	***
Head shake	6	83.3	no test	
Eye roll	14	50.0	1.2	ns
Head snap	12	41.7	0.4	ns

† Calculated Chi-square values for the contingency test ($\chi_{20,05,1}^2 = 3.48$); no test was conducted with small samples sizes and “ns” indicates a non-significant test result at $p=0.05$, with significance indicated at $p<0.05$ (*), $p<0.01$ (**) and $p<0.001$ (***).

2.4.2 PREY AND PREDATOR ABUNDANCE

Across all surveys, 85% of the recorded individuals were classified as potential prey for YOM and YOY makos (<40 cm FL) whilst 2% of individuals were classified as potential predators of YOM and YOY makos (>2 m FL). Mean percentage prey abundance was significantly higher at the single location, Perth Canyon (86.6%), where multiple YOM and YOY were observed, relative to the other 22 locations (75.1%) ($t=2.66$; $p<0.005$). No potential predators were observed at the Perth Canyon, whereas potential predators comprised a significantly higher 3.2% of the abundance at the other locations ($t=4.04$; $p<0.0005$).

2.4.3 BEHAVIOUR

Striking in the video records was the boldness of the small shortfin makos relative to behaviours of larger individuals, with boldness inferred from behaviours such as earliness and frequency of approaches and direct contact with the bait canister (**Table 2.2**). Significant behavioural differences were detected for YOM and YOY shortfin makos ($n=5$), which entered the camera’s field of view 1.7 times more frequently than juvenile and adult shortfin makos ($n=11$) and approached the stereo-BRUVS 1.3 times more frequently. YOM and YOY individuals charged,

nudged and bit the bait canister 2.3 – 2.8 times more frequently than their larger conspecifics. Between YOM and YOY animals and juvenile and adult animals, there was no effect of size class on the time of first arrival (54.1 min \pm 16.5 min SE vs. 64.1 min \pm 17.7 min SE), time spent in the field of view (6.7 min \pm 2.9 min SE vs. 4.8 min \pm 1.6 min SE) nor the span of time from first arrival to last departure (18.5 min \pm 8.9 min SE vs. 17.0 min \pm 5.2 min SE), respectively.

2.5 DISCUSSION

Our results suggest that the head of the Perth Canyon is a pupping grounds for shortfin makos. The Perth Canyon meets most of the pupping grounds criteria (*sensu* Heupel et al. 2007) in that: (1) multiple YOM and YOY were observed; (2) clear size segregation existed with no juvenile or adult shortfin makos observed concurrently, consistent with the results of Nosal et al. (2019); (3) elevated regional temperature was observed; and (4) potential prey were relatively abundant and no potential predators present. We did not see YOM and YOY in multiple years as per one of the criteria identified by Heupel et al. (2007). This may be due to the limited number of surveys or reflect the estimated three year reproductive cycle (Mollet 1990) if a limited number of philopatric females are using the canyon as a pupping ground.

That we observed single individual YOM and YOY at two additional locations along this coast highlights the potential importance of Western Australia's continental shelf over some 7° of latitude. To date, no other study has recorded exclusively YOM individuals in the Indian Ocean, although YOY and juveniles have been caught alongside adults on the Agulhas Bank in South Africa, with the smallest animals occurring where the Atlantic and Indian oceans meet (Groeneveld et al. 2014). As such, the Perth Canyon may constitute the first potential pupping ground identified for shortfin mako in the Indian Ocean.

Our findings are also the first to suggest that a submarine canyon may act as a “keystone structure” (Vetter et al. 2010) for reproduction and the earliest life stages of shortfin mako. The Perth Canyon is a 220 km-long shelf incision that begins ~50 km offshore and increases rapidly in depth from 281 to 4,683 m (Huang et al. 2014). Covering an area of 1,820 km², the canyon is the second largest on Australia's continental margin (von der Borch 1968) and one of only three on the continent to extend fully to the abyssal sea floor (Heap & Harris 2008). The remainder of Western Australia's shelf and slope is notably depauperate in significant submarine features and primarily classified as undifferentiated (Heap & Harris 2008), noting however that there are multiple small canyons along the coast (Huang et al. 2014). Telemetry data indicate that female shortfin mako consistently visit Perth Canyon, suggesting philopatric movements and a

preference for submarine canyons and other topographic features (Rogers et al. 2015). This is consistent with microsatellite data that support philopatry in female shortfin mako as a mechanism to maximise juvenile survival and imprint female offspring to appropriate pupping grounds (Schrey & Heist 2003). Our results therefore further suggest that bathymetric or other cues associated with the Perth Canyon may act as a navigational beacon for breeding individuals.

The case for the Perth Canyon as a keystone structure for early life stages of the shortfin mako is strengthened by the canyon's environmental conditions, as submarine canyons are known to act as biodiversity hotspots on otherwise species-poor continental shelves (Farrugio 2012, Bouchet et al. 2017, Fernandez-Arcaya et al. 2017). The optimal temperature range for the shortfin mako is 17-22°C in the case of juveniles and adults (Casey & Kohler 1992) although YOY may prefer slightly warmer temperatures of 18-27°C (Vélez-Marín & Márquez-Farías 2009). The Leeuwin Current flows south over the Perth Canyon, delivering a ~100 km-wide band of warm water to a depth of 250 m and raising the temperature of the upper layer by ~6-8°C relative to adjacent waters (Rennie et al. 2006, Rennie et al. 2009). With an average sea surface temperature of ~21°C, the Leeuwin Current is also 5°C warmer in the vicinity of the Perth Canyon than at the same latitude on the eastern borders of the Pacific and Atlantic Oceans (Feng et al. 2003). As such, it is possible that the Perth Canyon represents a suitable pupping ground for YOM and YOY shortfin mako in part due to its anomalous temperature regime. This interpretation of the results is consistent with Casey and Kohler (1992), who attributed observations of YOY shortfin mako at the Grand Banks in the Atlantic Ocean to warm temperatures associated with the Gulf Stream, and with McAuley et al. (2007), who, similarly, identified the warming Leeuwin Current as a potential reason for high abundances of juvenile sandbar shark *Carcharhinus plumbeus* at a location in southwest Australia.

Previous research also suggests that shortfin mako pupping grounds such as those in Baja California and Chile are typically highly productive sites where warm and cold water masses mix (Vélez-Marín & Márquez-Farías 2009). We note that, similarly, the Perth Canyon represents a convergence of tropical and sub-tropical oceanic water masses (Cresswell & Domingues 2009, Rogers et al. 2015). In addition, the abrupt changes in depth associated with the Perth Canyon itself interact with the Leeuwin Undercurrent, a northward flowing deep layer, creating nutrient-trapping cyclonic eddies within the canyon and generating upwelling events that reach the surface when the Leeuwin Current is weak (Rennie et al. 2006, Rennie et al. 2009) This

mechanism boosts local primary productivity within the Perth Canyon to levels 2.5 times higher than within the adjacent, relatively oligotrophic waters (Hanson et al. 2005, Rennie et al. 2009).

Shark pupping grounds are characterised by an abundance of prey and a scarcity of predators (Branstetter 1990, Beck et al. 2001, McAuley et al. 2007). Consistent with these hypotheses, we found that potential prey for YOM and YOY shortfin mako were on average more abundant at the Perth Canyon than other sampled locations and we observed no potential predators. Like all fishes, small shortfin mako have higher metabolisms than larger individuals and must consume great amounts of energy to grow (Clarke & Johnston 1999), particularly given that they are endothermic (Watanabe et al. 2015). The growth of juvenile shortfin mako is estimated to be approximately 50-61 cm year⁻¹ for the first year of life (Maia et al. 2007). This rapid growth rate allows individuals to reduce the amount of time spent at predator-vulnerable sizes (Post & Evans 1989) but likely drives a high demand for prey. This demand may explain our observations that small shortfin mako were more likely to engage in bold foraging behaviours such as repeated entries into the vicinity of the bait canister, nudging and biting than adults, with little apparent fear in response to risk given the lack of predators.

Our sample size of shortfin mako is small, with only 16 individuals observed, despite the large scale of sampling effort. As such, any inferences about the species must be made cautiously. Increasingly, methods are being developed to deal with sparse data, including Bayesian approaches applied to telemetry (Kinney et al. 2017) and zero-inflated modelling of BRUVS data (Udyawer et al. 2014). However, even in cases such as ours that do not lend themselves to statistical modelling, it is important to publish these results as a building block towards improved understanding of this threatened species. Moreover, it is unlikely that we are simply sampling the wrong habitat for shortfin makos: Nosal et al. (2019) found that YOY shortfin makos in the Southern California Bight spend 97% and 25% of their time in waters shallower than 40 m and 10 m respectively. However, we also note that sample sizes of fewer than ten individuals are not uncommon in shark pupping studies (Thorpe et al. 2004). Moreover, we cannot rule out that the low numbers of shortfin mako recorded on the stereo-BRUVS reflect the scarcity of shortfin makos globally. Other pelagic shark species were amply represented in the dataset, including the Galapagos shark *Carcharhinus galapagensis* (n=423), silky shark *Carcharhinus falciformis* (n=324), silvertip shark *Carcharhinus albimarginatus* (n=202). Even the scalloped hammerhead *Sphyrna lewini* (n=29) and great hammerhead *Sphyrna mokarran* (n=18) were more commonly observed than shortfin mako, despite their IUCN “Endangered” status. That the classification of shortfin makos was recently downgraded from “Vulnerable” to “Endangered” is consistent with

studies that have shown their fishing mortality to be underestimated by ten-fold (Byrne et al. 2017).

Heupel et al. (2007) argued that definitions of shark pupping grounds must become more stringent. In particular, they suggested that the mere presence of juveniles is insufficient to warrant a habitat being classified as a pupping ground. Consistent with the multiple criteria provided by Heupel et al. (2007), we have shown that the Perth Canyon may be an important pupping ground for the shortfin mako shark. While research on pupping grounds for pelagic shark species remains in its infancy, it is increasingly clear that YOM and YOY may rely on a precariously limited number of “suitable” oceanic habitats such as thermal fronts, mid-ocean ridges, seamounts and other high-productivity hotspots (Queiroz et al. 2016). Moreover, there is a growing understanding of the higher-than-expected residency of juvenile and adult shortfin makos, underpinning the need for local (Corrigan et al. 2018, Byrne et al. 2019, Francis et al. 2019) as well as regional management (Vaudo et al. 2017). Given that pelagic sharks are already amongst the most endangered vertebrates globally, our results suggest that there is an urgent need to not only identify sites of potential importance to their reproductive fitness but also to protect them.

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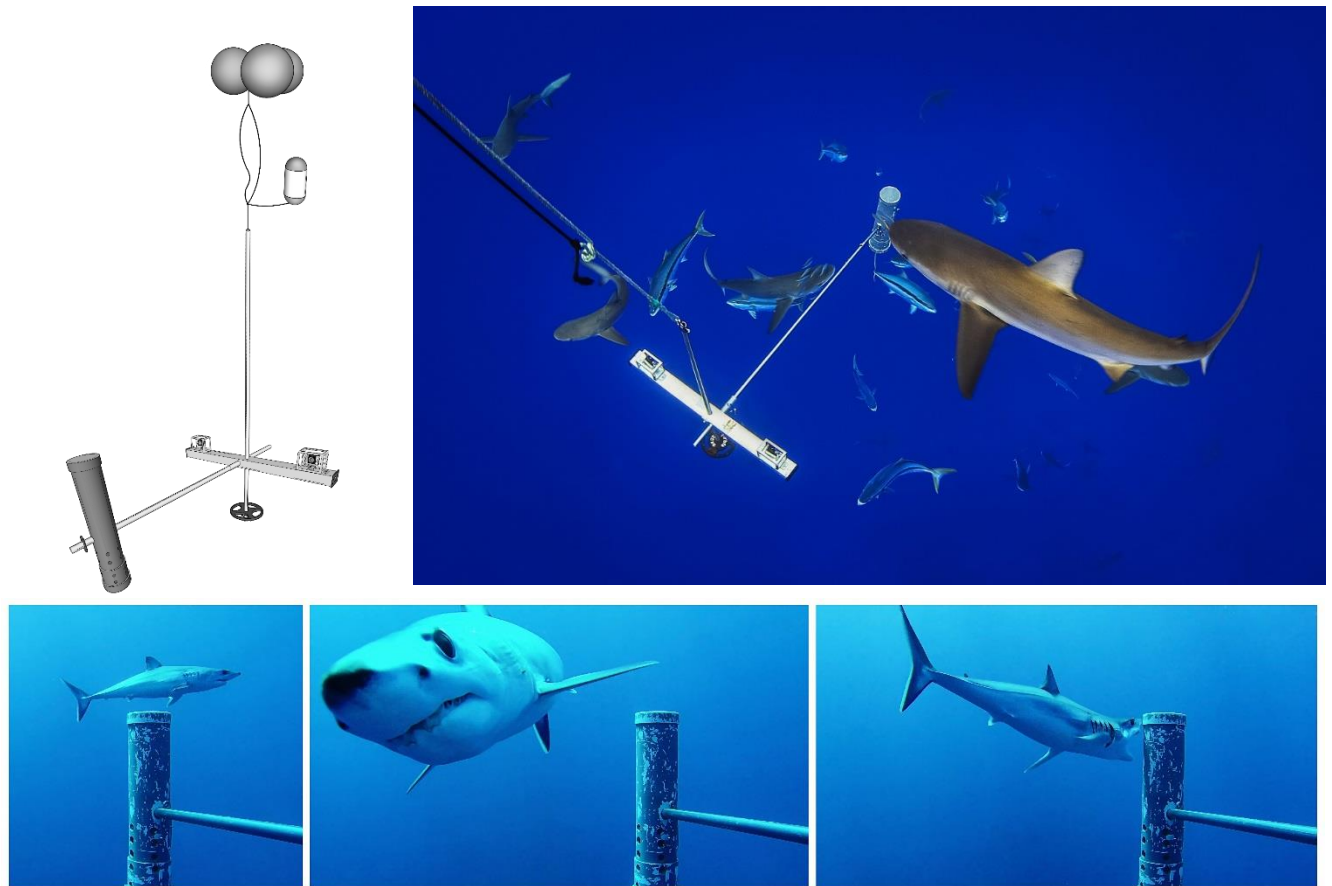
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2.9 SUPPLEMENTARY MATERIALS



Supplementary Figure 2. 1 Clockwise from upper left, a schematic of a mid-water baited remote underwater video system, a top down view of a deployed BRUV (courtesy of Manu San Felix), and three instances of two very small (<70 cm FL) shortfin mako *Isurus oxyrinchus* interacting with the BRUVS with the diagnostic black nose visible.

Supplementary Table 2. 1 Metadata on individual expeditions, including year, number of samples (n), configuration of the BRUVS, sampling design, location (decimal degrees) and start and end dates.

Expedition	Year	n	Config ^a	Design ^b	Latitude	Longitude	Start Date	End Date
Ascension I	2017	410	LL5	SRS	-7.9437	-14.3628	16/01/17	4/06/17
Ascension II	2017	85	LL5	SRS	-9.5784	-12.0800	26/05/17	3/06/17
BIOT I - Wet	2012	126	LL5	SRS	-5.7502	72.1747	22/11/12	7/12/12
BIOT II - Wet	2015	260	LL5	SRS	-6.3528	72.4446	10/01/15	24/01/15
BIOT III - Wet	2016	160	LL5	SRS	-6.1210	71.9649	7/02/16	21/02/16
Bremer Canyon	2017	100	LL5	SRS	-34.6234	119.8604	27/02/17	8/03/17
Clipperton Island	2016	51	LL3	SRS	10.2814	-109.2015	13/03/16	22/03/16
Cocos (Keeling) Islands	2016	109	LL5	SRS	-12.8126	97.5253	11/10/16	11/12/16
Far North Queensland I - Dry	2017	79	LL5	SRS	-11.5151	143.2280	7/06/17	18/06/17
Far North Queensland II - Wet	2017	85	LL5	SRS	11.4977	-143.4429	28/11/17	6/12/17
French Polynesia	2013	45	LL5	SRS	-20.6652	-137.9938	6/06/13	15/06/13
Gascoyne	2016	80	LL5	SRS	-21.8418	113.5772	15/09/16	22/09/16
Geographe Bay I - Autumn	2017	50	LL5	GRTS	-33.7187	115.4797	4/02/17	8/02/17
Geographe Bay II - Autumn	2018	100	LL5	GRTS	-33.4462	115.2603	9/02/18	13/02/18
Gracetown	2018	100	LL5	GRTS	-34.0148	114.7642	21/05/18	2/06/18
New Caledonia I - Dry	2012	45	LL5	SRS	-20.2169	164.4706	18/10/12	26/10/12
New Caledonia II	2014	115	LL5	SRS	-22.5110	166.3842	31/01/14	12/02/14
New Caledonia III - Dry	2016	80	LL5	SRS	-21.8418	113.5772	15/09/16	22/09/16
Niue	2016	100	LL3	SRS	-19.5306	-168.8279	27/09/16	8/10/16

Expedition	Year	n	Config ^a	Design ^b	Latitude	Longitude	Start Date	End Date
Northwestern Australia	2017	160	LL5	SRS	-13.8630	125.7347	30/06/17	21/07/17
Palau	2014	147	LL5	SRS	7.4106	134.4479	2/09/14	22/09/14
Perth Canyon I - Autumn	2013	52	LL5	GRTS	-31.9466	115.1035	30/03/13	17/05/13
Perth Canyon II - Autumn	2016	50	LL5	GRTS	-31.9466	115.1035	27/04/16	4/05/16
Perth Canyon III - Spring	2016	55	LL5	GRTS	-31.9466	115.1035	25/11/16	2/12/16
Perth Canyon IV - Autumn	2018	100	LL5	GRTS	-31.9780	115.1302	17/04/18	12/05/18
Pilbara I - Dry	2017	90	M5	SRS	-20.1394	116.3238	1/05/17	9/05/17
Pilbara II - Dry	2017	96	M5	SRS	-20.1480	116.3112	28/09/17	5/10/17
Rapa	2014	53	LL3	SRS	-27.6830	-144.0926	18/10/14	29/10/14
Revillagigedo	2016	75	LL3	SRS	18.8114	-112.7549	30/03/16	11/04/16
Rowley Shoals	2017	115	LL5	SRS	-14.7959	118.8485	16/11/17	22/11/17
Selvagens Islands	2015	57	LL3	SRS	30.1025	-15.9573	6/09/15	15/09/15
Timor Sea	2012	116	M5	SRS	-11.8096	127.1486	16/09/12	2/10/12
Tonga	2013	36	LL5	SRS	-23.5117	-178.5142	5/07/13	14/07/13
Tristan da Cunha	2017	81	LL3	SRS	-37.0998	-12.4341	16/01/17	2/02/17
West Dirk Hartog Island I - Wet	2012	67	M5	SRS	-26.1348	113.1712	18/04/12	25/04/12
West Dirk Hartog Island II - Dry	2017	75	LL5	SRS	-26.2854	113.2309	16/09/17	21/09/17

- a) Configuration refers to whether mid-water BRUVS were deployed in longline sets of 3 (LL3) or 5 (LL5) or were moored in groups of 5 (M5).
b) Design refers to whether stratified random sampling (SRS) or generalised random tessellation stratified (GRTS) approach (Stevens & Olsen 2004).

Supplementary Table 2. 2 Ethogram classifying behaviours of sharks occupying the pelagic environment, consisting of previously identified behaviours (indicated by superscript). FL = fork length and Hz = frequency per second. Martin (2007); Myrberg Jr and Gruber (1974); Thompson (2014); Turner (2016).

Class	Behaviour	Definition
Entries/Exits	Enter	First point at which the shark is visible in both camera's fields of view
	Exit	Last point at which the shark is visible in both camera's fields of view
Direct behaviours	Approach ^d	Approaching the stereo-BRUVS to within ~2-3 FL, as if to make contact, before veering away
	Bite ^c	Closure of jaw on receiver, bait arm or animal and contact with teeth
	Charge ^a	Fast (~2-3+ FL/s) approach towards the stereo-BRUVS, often terminated by veering away on a perpendicular course within a distance of ~1-2 FL
	Feeding ^d	Consumption of bait (released from bait canister)
	Nictitating ^c	Closing of the nictitating membrane >30% of the eye
	Nudge ^d	Contact with the bait canister with fin, snout or head
	Ramming ^a	A shark using its rostrum to forcefully strike a receiver, often causing it to retreat or recoil
Indirect behaviours	Circling ^d	Swimming in a loose looping trajectory whilst maintaining usual swimming position
	Head shake ^a	Rhythmic, exaggerated lateral shaking of the head, usually rapid (>2 Hz) and through an arc of >30°
	Head snap ^d	Singular, exaggerated lateral whip of the head
	Hunch ^b	Arching of the back with the tail slightly lowered and head slightly raised, so that the body forms a reversed "U" shape
	Jaw gape ^{a,c}	A slow, exaggerated opening of the jaw (\pm approximately 30-90°, estimated as the angle formed at the mouth commissure), conspicuously wider than during ram ventilation
	Patrolling ^b	Relatively straight-line swimming pattern, with usual swimming position
	Pectoral fin depression ^a	A sustained (>5 s), bilateral lowering of the pectoral fins from usual swimming position
	Rapid, tight pattern swimming ^a	Fast swimming marked by sudden and frequent changes in direction

Class	Behaviour	Definition
	Rapid withdrawal ^a	Rapid movement of the shark away from a display the stereo-BRUVS at ≥ 3 FL/s, initiated by several strong tail beats followed by a long glide covering ≥ 5 m
Timing	Time of first arrival	Total time from stereo-BRUVS settlement to first entry
behaviours	Total time in view	Total time between an animal's entry and exit.
	Span	Total time from first entry to last exit.

2.10 SUPPLEMENTARY REFERENCES

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CHAPTER 3: STABLE PATTERNS IN OCEAN WILDLIFE SUPPORT THE ESTABLISHMENT OF MARINE PROTECTED AREAS

Keywords: pelagic fish and sharks; annual and seasonal variability; mid-water BRUVS; Perth Canyon; Western Australia; MPAs

3.1 ABSTRACT

The global impact of human activity in the Anthropocene is driving unprecedented declines in marine wildlife, with ocean extinction rates predicted to rival those already observed terrestrially. Reductions of up to 98% in the abundance of pelagic taxa such as tunas and sharks demonstrate the challenges faced by conventional fisheries management in many parts of the world and suggest a need for additional measures to ensure the conservation and sustainable exploitation of marine wildlife. In coastal systems, marine protected areas (MPAs) have been shown to increase fish diversity, abundance, size, and biomass, and to provide both economic benefits to fisheries and environmental services such as ecosystem resilience. The effectiveness of MPAs for pelagic taxa is less clear, with arguments against the establishment of pelagic MPAs typically based on high spatial and temporal variability in pelagic assemblages due to their mobility. Arguments against MPA benefits for pelagic taxa are typically based on high spatial and temporal variability in pelagic assemblages due to their mobility. We used mid-water baited remote underwater video systems to document the status of pelagic wildlife at the Perth Canyon Marine Park in Western Australia, over a 5-year period. We found interannual stability in the spatial patterns of pelagic taxa richness, and variability between the Austral seasons, consistent with ecological theory. Our results suggest that spatial and temporal patterns in ocean wildlife are sufficiently clear that the effects of protection, once on-water protection within the MPA begins, can be determined. This is a requirement if the conservation benefits of protected areas are to be appropriately determined.

3.2 INTRODUCTION

Pelagic fishes, including sharks, are amongst the most threatened vertebrates globally, with 25% of extant taxa at risk of extinction (Dulvy et al. 2008, Dulvy et al. 2014). The most worrying examples include ecologically critical apex predators such as hammerhead (*Sphyrna* sp.) and thresher (*Alopias* sp.) sharks, which have been reduced in abundance to between 2% and 10% of their pristine, pre-industrial fishing levels. Tunas and their relatives have experienced similar declines (Juan-Jordá et al. 2011), with 11% of assessed taxa listed as threatened by the International Union for the Conservation of Nature (IUCN) (Collette et al. 2011). The inherent noisiness of fisheries data means that there remains discussion as to the precise magnitude of these declines (Hampton et al. 2005). However, the overall precipitous nature of these downward trajectories is clear (Juan-Jordá et al. 2011). Efforts to improve the management and recovery of declining fish stocks have seen limited success globally, both within national Exclusive Economic Zones (EEZs) and on the high seas through Regional Fisheries Management Organizations (RFMOs) (Beddington et al. 2007, Cullis-Suzuki and Pauly 2010), with only few exceptions (Neilson et al. 2013). The breadth and scale of declines means that these exceptions have little impact on global trends in ocean resilience and health (Pauly and Zeller 2016, Zeller and Pauly 2018).

Marine protected areas (MPAs) are increasingly recognized as a key strategy to re-establish flourishing oceans. Whilst focused primarily on biodiversity conservation and maintenance of ecosystem services (Lubchenco et al. 2003, Halpern et al. 2010), MPAs can also play an important role in the management and recovery of commercially targeted fish populations (Vandeperre et al. 2011, Rice et al. 2012), securing foraging areas (Young et al. 2015) building ecosystem resilience (Davies et al. 2017, Roberts et al. 2017) and improving social equity (Sumaila et al. 2015). To be effective, MPAs need to represent all major marine ecosystems with a minimum coverage of 30% of the world's oceans in order to meet the UN Sustainable Development Goals (O'Leary et al. 2016) and protect both the seabed and water column (O'Leary & Roberts 2018). Key attributes of MPA success include being large, no-take and well-enforced (Edgar et al. 2014), noting that smaller MPAs can also deliver specific benefits such as increased fisheries productivity (Marshall et al. 2019). These scientific recommendations stand in stark contrast to targets in multinational agreements: for instance, the Convention on Biological Diversity (CBD) and Aichi are for 10% of the marine environment conserved in ecologically representative networks by 2020 (CBD 2010). As is typically the case with multinational agreements, this low target represents political pragmatism rather than achievement of a goal that will obtain the desired outcome of a return to flourishing oceans.

A key argument against the implementation of MPAs in open ocean environments is that animals will simply swim out of the reserve (Kaplan et al. 2010). Specifically, a common view is that the greater mobility of pelagic taxa such as sharks and tunas means that they will not benefit from MPAs (Runge et al. 2014). Moreover, the association of pelagic taxa with dynamic oceanographic features such as fronts implies that dynamic MPAs with moving boundaries should be preferred over MPAs that are fixed in space (Hobday et al. 2014, Miller & Christodoulou 2014, Scales et al. 2014). However, the composition of pelagic fish assemblages is complex, including both taxa that undertake large-scale movements as well as taxa with more restricted home range, including “reef” sharks that spend time in pelagic environments (White et al. 2017). The former, including sharks and tunas, may cross ocean basins but telemetry studies have shown many do so predictably e.g. along migration corridors. These studies also show that such animals consistently use key locations such as canyons and sea mounts (Fiedler & Bernard 1987, Holland & Grubbs 2007, Walli et al. 2009). Photographic identification, genetic and isotope data have also exposed philopatry in sharks (Bernard et al. 2016, Flowers et al. 2016). More recently, a growing body of research has highlighted associations between topography and predators in terrestrial and marine systems (Bouchet et al. 2015). In Western Australia, for instance, abundance hotspots for commercially exploited pelagic fishes are linked to seabed features including large submarine canyons over a decade (Bouchet et al. 2017). As such, it is increasingly clear that mobile animals use a network of migratory highways (Sequeira et al. 2018) that they follow to aggregation areas, with the latter supporting critical life-history stages such as pupping grounds (Forrest et al. 2019) and activities such as breeding (Erisman et al. 2017) and feeding (Rennie et al. 2009, Morato et al. 2010).

Our oceans’ “unnatural history” (*sensu* Roberts 2007) may also have left us with relatively mobile populations of pelagic taxa as a function of selection pressures. Historical fishing was largely coastal and thus fishing mortality disproportionately removed residential, less-mobile, individuals within a population. Evolution predicts that those individuals escaping early coastal exploitation were more mobile and thus the remnant populations today comprise the more migratory individuals of the taxa (Mee et al. 2017). Such supposition reflects evidence from tagging data that consistently demonstrate variation among individuals with respect to residential vs. long distance movements (Rogers et al. 2015, Hays et al. 2016), including for tuna (Fromentin & Lopuszanski 2014). The loss of residential “stocklets” creates management challenges (Carpi et al. 2017) but is also a strong argument for why large protected areas will generate benefits. Indeed, modelling by Mee et al. (2017) suggests that fishing mortality will selectively remove individuals which travel outside MPAs, reversing previous impacts. As such, the more residential individuals of a given taxa will contribute

more offspring to future generations that, like reef fish (Goñi et al. 2008, Halpern et al. 2009), will spill over to those areas accessible to fishing. Additionally, protection of critical habitat such as spawning and puppery grounds generates significant population benefits even if adults are mobile (Kerwath et al. 2008).

The diverse nature of pelagic taxa, their patchy distributions and relative low abundance across seascapes means that designing offshore MPAs is challenging. However, if spatial patterns are stable and include important habitats (Kerwath et al. 2013), offshore MPAs may assist pelagic population recovery. Here, we test the degree to which the distribution and composition of pelagic fish assemblages in the Perth Canyon Marine Park (PCMP) is stable among years and between seasons. Established by the Australian Government in 2012, the PCMP is part of the national network of Australian Marine Parks (AMPs) and includes both IUCN II (no-take) and IUCN IV (habitat protection) zones, as per new management plans which came into effect in 2018. The pelagic assemblage of the PCMP was first surveyed in 2013 using mid-water baited remote underwater video systems (BRUVS) (Bouchet & Meeuwig 2015). Mid-water BRUVS are increasingly used to document the relative diversity, abundance and size distribution of pelagic fishes and sharks as an alternative to lethal, fisheries-based sampling (Bouchet et al. 2017). We build on this previous work with subsequent surveys completed in 2016 and 2018 to determine the degree to which the spatial distribution of pelagic wildlife is stable through time, both inter-annually and seasonally.

3.3 METHODS

3.3.1 STUDY AREA

The Perth Canyon lies approximately 60 kilometers west of the Australian mainland and Perth, the region's capital and largest population center (**Fig. 3.1**). This submarine canyon is the second largest shelf-incising feature on Australia's continental margin (von der Borch 1968), and is topographically complex, with three major identified heads (Huang et al. 2014). Larger than America's Grand Canyon, it extends to 4,000 m depth and covers a surface area of approximately 1,820 km². Located in a largely oligotrophic region (Cresswell 1991, Hanson et al. 2007, Rennie et al. 2009), the canyon system intersects the equatorward-flowing Leeuwin Undercurrent, a driver of nutrient upwelling in the region. Consequently, the Perth Canyon supports relatively high levels of seasonal productivity, particularly within the canyon heads (Rennie et al. 2009). The PCMP was established in 2012 with the 2013 management plan placing a small IUCN II zone (58.3 km²) at the most landward canyon head; this zone, approximately 0.8% of the 7,409 km² PCMP, was the focus of our study. In 2018, this IUCN II zone was moved offshore to a less steep canyon head, approximately 40

km to the southwest of the original location and our focal study areas was rezoned as IUCN IV “habitat protection”, providing protection to the seabed but not the water column.

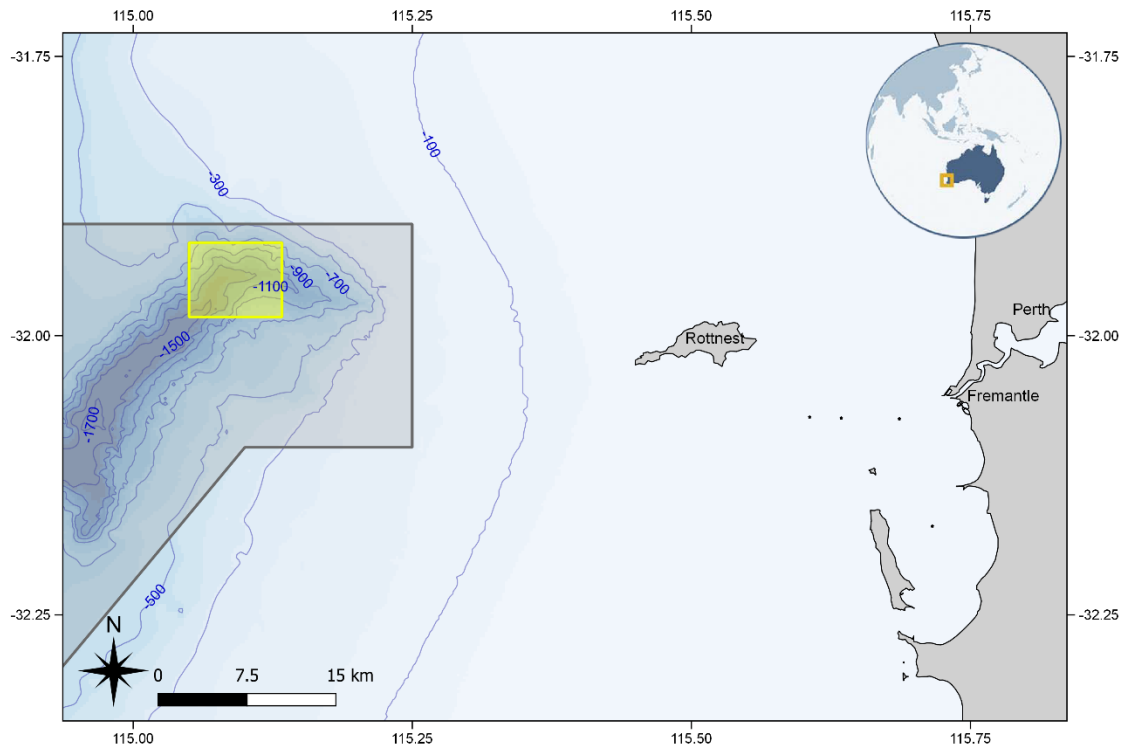


Figure 3. 1 Location and bathymetry of the Perth Canyon showing zoning as per the 2018 in effect management plan. Yellow shaded area is the IUCN IV “habitat protection” zone and grey shaded area is IUCN VI “multiple use” zone.

3.3.2 MID-WATER STEREO-BAITED REMOTE UNDERWATER VIDEO SYSTEMS

We sampled the pelagic wildlife assemblage of the PCMP using mid-water stereo-baited remote underwater video systems (BRUVS) (Letessier et al. 2013, Bouchet & Meeuwig 2015, Letessier et al. 2015, Letessier et al. 2017). Mid-water BRUVS are a modification of the seabed BRUVS that have been successfully used to document benthic fish assemblages (Cappo et al. 2006). These stereo-BRUVS consist of a central stainless steel frame (full height = 1,450 mm) that supports two high definition GoPro HD Hero 4 cameras with backscatter flat-port underwater housings, mounted on a cross bar (950 mm) perpendicularly to an adjustable bait arm (1,800 mm; **Fig 3.2**). The cameras converge inwardly at an angle of 4 degrees and are set to record in wide field-of-view mode to maximize the area captured in each video frame and improve detection rates of animals up to a distance of 10 m. The bait arm supports a 45 cm long perforated PVC pipe bait container that ensures the slow release and diffusion of bait in the water column throughout the duration of the deployment. It also acts as a rudder to minimize rotational movement of the unit such that a down-stream field of view is maintained. All rigs were deployed at a suspension depth of 10 m.

Building on increasing experiences with mid-water sampling, we have moved from moored (Letessier et al. 2013) and free-drifting (Bouchet & Meeuwig 2015) deployments to a longline configuration. The stereo-BRUVs in this study were deployed in longline sets of five rigs, with individual units tethered by 200 m of line. The move from the 2013 format of free-drifting deployments to a longline configuration increases sampling efficiency, simplifies retrieval and minimizes loss of equipment, but noting that individual rigs are not independent given they are deployed from a single point of entry.

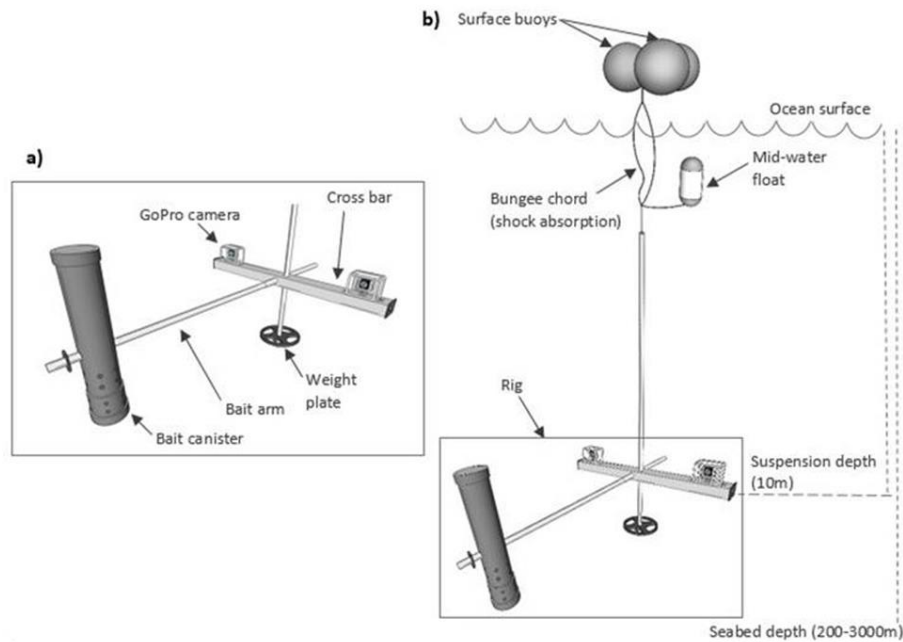


Figure 3. 2 Schematic of mid-water BRUVs indicating (a) a rig and (b) in-water configuration.

3.3.3 FIELD SAMPLING

The initial Austral autumn survey in 2013 (Bouchet & Meeuwig 2015) was followed by two Austral autumn surveys in 2016 and 2018, and an Austral spring survey in 2016 (**Fig. 3.3**). All were completed in and adjacent to the 2013 IUCN II zone. Sampled seabed depths ranged from 635 m to 1,580 m. There was no on-water MPA-specific management of fishing activities during the course of these “baseline” surveys.

Mid-water stereo-BRUVS were deployed at ten (Autumn 2013 and 2016), 15 (Autumn 2018) and 11 (Spring 2016) sites (**Supplementary Table 3.1**). The 2013 deployments, although not in longline configuration, largely drifted together from the point of deployment (**Fig. 3.3**) and were treated as “sets”. All sampling was undertaken during daylight hours between 7:00 and 17:00 (GMT +8) to minimize any effects of differential crepuscular and nocturnal behavior in fish (Axenrot et al. 2004, Birt et al. 2012). We largely repeated the sampling plan of Bouchet and Meeuwig (2015) which was based on a generalized random tessellation stratified (GRTS) design (Stevens & Olsen 2004), targeting similar areas within the focal study location (**Fig. 3.3**).

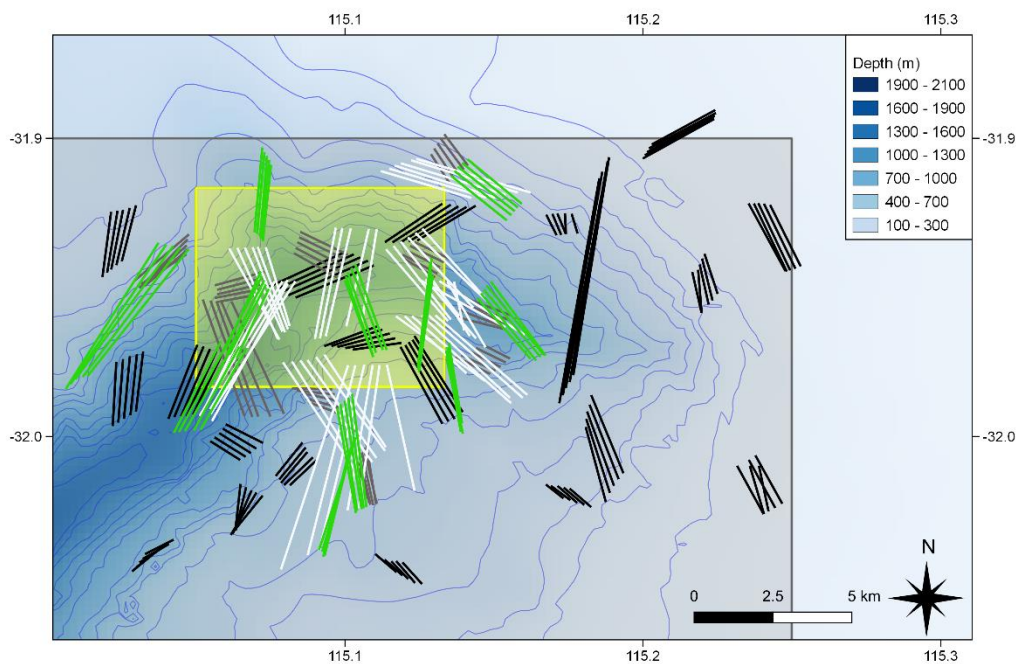


Figure 3.3 Location of sampling and tracks of drifting mid-water BRUVS in Austral Autumn 2013 (white), Austral Autumn 2016 (grey), Austral Spring 2016 (green) and Austral Autumn 2018 (black) with the internal yellow shaded box representing the boundaries of the IUCN IV zone at the head of the canyon as of the 2018 management plan.

Camera pairs were synchronized immediately before deployment by a slow hand clap, having previously been independently calibrated in an enclosed pool environment using the software CAL (SeaGIS Pty Ltd) and following the protocol described by (Harvey & Shortis 1998). Rigs were deployed for a minimum of two hours and then retrieved. Rigs were also fitted with GPS loggers that recorded their positions at regular, one-second intervals. The surveys were conducted from the vessels *Whale Song*, *Thalanyji* and *NEGU* under an exemption from the Australian Marine Safety Authority (EX2016/A185; EX2017/A007A) and field work was approved under ethics permit RA/3/100/1484.

3.3.4 IMAGE ANALYSIS

Videos were imported into the EventMeasure software package (SeaGIS Pty Ltd <http://www.seagis.com.au/>) for processing. Processing commenced once the rig was stabilized at the suspension depth of 10 m and ended at 2 hours. We identified all observed individuals within the field of view to the lowest taxonomic level possible, and estimated relative abundance as the maximum number of individuals of a given taxa in a single frame (MaxN; Cappo et al. 2006). We also took stereo-measurements of all observed individuals to determine their distance from the camera and generate estimates of fork length. Image analysis included independent checks on taxa identifications performed by experienced observers.

3.3.5 DATA TREATMENT AND STATISTICAL ANALYSIS

The first part of the analysis focused on determining whether univariate attributes of the assemblage, namely total unique number of taxa (TR_U), taxonomic richness (TR), total abundance (TA), total biomass (TB; kg) and fork length (FL; cm), varied between surveys. Taxa rather than species were analyzed as not all individuals were identified to species. Taxa included all observed animals, including marine mammals and reptiles. We first calculated these values from the video for each of the five rigs on a given longline. We then calculated the mean values for each attribute for each longline. We were also interested in the degree of redundancy within the longlines so we calculated the percentage of rigs (%rigs) on which a given taxa was observed within a longline set.

We applied a number of approaches for statistical analysis to explore differences among the four surveys. Taxa accumulation curves were calculated for all surveys combined and for each of the four surveys individually (Clarke et al. 2014). Chao 2 was used as it uses the incidence of species rather than their abundance (Gotelli & Colwell 2011), with Michaelis-Menten used to estimate the pool size and the number of samples needed to record 50% of the taxa. Variation in univariate measures of TR and TA were tested using a one-way permutational analysis of variance (PERMANOVA) based on unrestricted permutations (Anderson 2017) with “survey” as the factor; the permutational approach was chosen as it is robust to heterogeneity in the data (Anderson 2017) and because ANOVA is insensitive to heterogeneity and all but extreme departures from normality, assuming comparable sample sizes (Zar 1999), which is the case here. A Euclidean distance matrix was calculated as the data are continuous. The multivariate composition data, representing the abundance of each taxa per rig, was square-root transformed, with the Bray-Curtis resemblance matrix then calculated. Homogeneity of the multivariate data was tested with PERMDISP and then a one-way PERMANOVA based on unrestricted permutations with “survey” as a factor was

also applied. A principal coordinates analysis (PCO) was then completed to visualize an unconstrained ordination of the data. All analyses were completed in Primer 6 with the PERMANOVA+ add-on (<https://www.primer-e.com/>).

To determine whether the spatial distribution of taxa was consistent in time, we conducted a number of analyses in the statistical software package R v3.4.3 (R Development Core Team 2016). First, we built maximum entropy (MaxEnt) models of wildlife occurrence following the methods of Bouchet and Meeuwig (2015). We considered an identical set of candidate explanatory variables, with the addition of daytime remote-sensed chlorophyll-a (Chl-a in mg/m^3), derived from 8-day AQUA MODIS composite images (available at 4 km resolution). Values were obtained via the Environmental Data Connector (<http://www.pfeg.noaa.gov/products/EDC/>) and both seasonal means and variances calculated using purpose-built scripts. Seasons were defined as spanning from April 1 to June 30 (austral Autumn) and October 1 to December 31 (austral Spring) following Sequeira et al. (2012). We pooled all visual observations of pelagic taxa and subsequently thinned them to reduce the effects of spatial auto-correlation and improve potential model transferability (Verbruggen et al. 2013). We used the *ENMEval* package (Muscarella et al. 2014) for tuning, allowing optimal feature classes (linear or quadratic, Sequeira et al. (2012) and regularization multipliers (β , from 1 to 8 in one-unit increments) to be chosen automatically according to Akaike's second-order information criterion score (AICc). We then identified the most relevant predictors with the help of iterative functions from the *MaxentVariableSelection* package (Jueterbock et al. 2016), run at a contribution threshold of 5%. Reduced models were implemented in *dismo* (Hijman et al. 2016), with an equal presence to background (pseudo-absence) point ratio, the latter being randomly extracted from a custom bias grid defined as per Bouchet and Meeuwig (2015) and rescaled to range between 1 and 20, as recommended by Elith et al. (2010). Models were trained on 75% random partitions of the data (repeated $n=50$ times), reserving the remaining 25% for performance evaluation based on the true skill statistic (TSS, Allouche et al. (2006). We interpreted MaxEnt's logistic output as a measure of relative occurrence probability and mapped final predictions across the entire study region.

To allow fair and more robust comparisons of spatial patterns across time periods, we only truncated model predictions made in areas characterized by novel conditions relative to the training data, using the multivariate environmental similarity surface (MESS) tool (Elith et al. 2011). This tool quantifies the 'distance' (in environmental space) between each pixel and the set of reference points contained in the original sample, and can help diagnose errors to which models are prone when projected into non-homologous systems (Dormann et al. 2007, Fitzpatrick & Hargrove 2009). We computed MESS indices for each survey, and only

retained those intersecting grid cells that yielded positive MESS values in all years as areas where model predictions are likely most credible (Elith et al. 2010, Verbruggen et al. 2013). We then compared maps based on the structural similarity (SSIM) index proposed by Jones et al. (2016), which accounts for spatial dependencies. Specifically, the SSIM index accounts for structural similarities in means, variance and covariance patterns in geographic space, offering potential insights into underlying ecological processes responsible for patterns of similarity/dissimilarity (Jones et al. 2016). The SSIM index ranges between -1 and 1, indicating negative and positive relationships respectively. We also performed a visual comparison of the distribution of probability values using kernel density estimates (KDE), in keeping with Langlois et al. (2012). The approach relies on a null model of no difference and a permutation test performed from reshuffles of the data into random pairs (**Supplementary Fig. 3.2**).

3.4 RESULTS

During the three 2016 and 2018 surveys, we captured a total of 360 hours of underwater video in addition to the existing 104 hours from 2013, corresponding to 232 samples on 46 longline sets (**Supplementary Table 3.1**). In total, 763 individuals representing 28 taxa from 15 identified families in addition to translucent larval fish of unknown species (**Table 3.1**). Across all four surveys, the most common taxa were mackerel scads *Decapterus* sp. (45.4%), small jacks (Carangidae) (14.9%) and larval fish (13.7%) (**Table 3.1**). Importantly, we recorded the presence of three young-of-the-month and young-of-the-year shortfin mako sharks *Isurus oxyrinchus* (Forrest et al. In Review). In the 2016 Austral spring survey, we recorded 89 individuals from 13 taxa representing six families (**Table 3.1**). During this survey, small jacks and scads were common with the only observations of southern bluefin tuna *Thunnus maccoyii* and great hammerhead shark *Sphyrna mokarran* recorded. In the 2016 Austral autumn survey, we recorded 175 individual fishes representing seven taxa from five families. The most common taxa were small scads and common dolphinfish *Coryphaena hippurus* and the size range went from translucent fish larvae up to 2 cm to a 4.7 m dwarf minke whale *Balaenoptera acutorostrata*. The 2018 Austral autumn survey recorded 189 individuals from 12 taxa of 10 families and was dominated by small jacks, small scads, small leatherjacks and larval fish.

Of the 28 taxa observed, 50% showed no redundancy along a given longline, appearing only on a single rig when present (**Table 3.2**). The remaining 14 taxa, when present, were observed on 18% of the rigs within a given longline, or on average, slightly fewer than 2 rigs within the line. In considering the 128 combinations of taxa and longlines, there were 13 instances where a given taxa was observed on four or more rigs within the longline, with

small scads, and small jacks accounting for 10 of these incidences (**Supplementary Table 3.2**). Large taxa such as shortfin mako, blue and hammerhead sharks, wahoo *Acanthocybium solandri* and black marlin *Istiompax indica* showed no redundancy despite their ability to move rapidly along the 200 m distances separating individual rigs. Larval fish appeared on between 1 and 3 rigs per longline, indicating their patchy nature given their limited swimming speed.

Taxa accumulation curves were estimated for individual rigs (n=232 deployments) rather than longlines (n=46) given the low level of redundancy between individual rigs along the longline, and the low sample size for individual surveys (n=10-15 longlines). The curve for the combined survey data (**Supplementary Figure 3.1**) suggests a taxa pool of 29 and that 42 samples (rigs) will capture half of the taxa pool. For individual surveys, taxa pools were estimated between 8 and 18, with between 4 and 12 samples needed to capture 50% of the taxa pool. The autumn 2018 survey was notable in its separation from the other 3 surveys. The total number of unique taxa observed across the longlines ranged from 0 to 7, with an overall mean of 2.8 ± 0.26 (SE), and did not vary significantly among locations ($p=0.24$; **Table 3.3** and **Fig. 3.4a**). Mean taxa richness across the longlines ranged between 0 and 2.6 per rig, with an overall mean of 1.03 ± 0.10 (SE), and varied significantly among surveys ($p=0.022$; **Table 3.3** and **Fig. 3.4b**) with the autumn 2018 survey having significantly lower diversity than any of the other three surveys. Mean total abundance across the longlines ranged from 0 to 22.2 individuals per rig, with a mean of 3.14 ± 0.56 (SE) and a significant difference in \log_{10} transformed abundance across the four surveys ($p=0.043$; **Table 3.3** and **Fig. 3.4c**), with the autumn 2018 survey again lower than the previous surveys. Total biomass excluding cetaceans ranged from 0 to 38,289 kg, with an overall mean of $4,352 \text{ kg} \pm 1,303$ (SE), and \log_{10} -transformed, did not vary significantly among surveys ($p=0.96$; **Table 3.3** and **Fig. 3.4d**). Mean \log_{10} -transformed fork length also did not vary significantly among surveys ($p=0.36$; **Table 3.3** and **Fig. 3.4e**), with mean longline values ranging between 1.6 cm and 107.6 cm, with a mean of $22.2 \text{ cm} \pm 3.7 \text{ cm}$.

Table 3. 1 Pelagic species (scientific and common names) observed in the focal study area across all surveys, ordered by family.

Family	Scientific Name	Common name	Autumn 2013		Autumn 2016		Spring 2016		Autumn 2018	
			n [†]	\bar{x} [‡]	n	\bar{x}	n	\bar{x}	n	\bar{x}
Balaenopteridae	<i>Balaenoptera acutorostrata</i>	dwarf minke whale	3	0.07
Carangidae	<i>Atule mate</i>	yellowtail scad	1	0.02
	Carangidae	jacks	3	0.06	30	0.46	91	1.82
	<i>Decapterus muroadsi</i>	amberstripe scad	5	0.08
	<i>Decapterus</i> sp.	scads	199	4.38	81	1.80	15	0.23	38	0.76
	<i>Naucrates ductor</i>	pilotfish	36	0.76	4	0.09	2	0.03	2	0.04
Carcharhinidae	<i>Carcharhinus brachyurus</i>	copper shark	2	0.03
	<i>Prionace glauca</i>	blue shark	1	0.02
Coryphaenidae	<i>Coryphaena equiselis</i>	pompano dolphinfish	2	0.03
	<i>Coryphaena hippurus</i>	common dolphinfish	48	1.00	21	0.47	17	0.26	8	0.16
Echeneidae	<i>Echeneis naucrates</i>	live sharksucker	1	0.02
	<i>Remora</i>	shark sucker	6	0.13
Fistulariidae	<i>Fistularia</i> sp.	cornetfishes	2	0.04	1	0.02
Istiophoridae	<i>Istiompax indica</i>	black marlin	1	0.02
Lamnidae	<i>Isurus oxyrinchus</i>	shortfin mako	1	0.02	3	0.06
Molidae	<i>Mola mola</i>	ocean sunfish	2	0.06
Monacanthidae	<i>Aluterus monoceros</i>	unicorn leatherjacket	1	0.02
	Monacanthidae	leatherjackets	4	0.08
Myliobatidae	<i>Mobula</i> sp.	mobula rays	1	0.02
Nomeidae	<i>Psenes cyanophrys</i>	freckled drifffish	4	0.08	3	0.06
Scombridae	<i>Acanthocybium solandri</i>	wahoo	2	0.04
	<i>Katsuwonus pelamis</i>	skipjack tuna	2	0.04
	<i>Thunnus maccoyii</i>	southern bluefin tuna	7	0.11
	<i>Thunnus</i> sp.	tunas	8	0.17
Sphyraenidae	<i>Sphyraena</i> sp.	barracudas	1	0.03
Sphyrnidae	<i>Sphyrna mokarran</i>	great hammerhead	1	0.02
Syngnathidae	Syngnathidae	pipefishes	1	0.02
		larval fish	62	1.38	5	0.08	35	0.70

[†]Total number of sightings per species.

[‡]Mean sightings per survey.

Table 3. 2 Fork lengths (FL; cm) of pelagic species observed in the focal study area in the Perth Canyon Marine Park, ordered by family.

Family	Scientific Name	Common name	FL (cm) [†]	% Red [‡]	% mean TA [§]
Balaenopteridae	<i>Balaenoptera acutorostrata</i>	dwarf minke whale	470.0	40	0.4
Carangidae	<i>Atule mate</i>	yellowtail scad	5.1	0	0.1
	<i>Carangidae.</i>	jacks	4.3	24	14.9
	<i>Decapterus muroadsi</i>	amberstripe scad	5.8	5	0.5
	<i>Decapterus</i> sp.	scads	11.8	34	45.4
	<i>Naucrates ductor</i>	pilotfish	16.8	14	5.8
Carcharhinidae	<i>Carcharhinus brachyurus</i>	copper shark	240.9	20	0.2
	<i>Prionace glauca</i>	blue shark	160.6	0	0.1
Coryphaenidae	<i>Coryphaena equiselis</i>	pompano dolphinfish	36.6	20	0.2
	<i>Coryphaena hippurus</i>	common dolphinfish	58.2	12	12.0
Echeneidae	<i>Echeneis naucrates</i>	live sharksucker	45.9	0	0.1
	<i>Remora remora</i>	shark sucker	13.0	0	0.8
Fistulariidae	<i>Fistularia</i> sp.	cornetfishes	15.3	10	0.4
Istiophoridae	<i>Istiompax indica</i>	black marlin	203.4	0	0.1
Lamnidae	<i>Isurus oxyrinchus</i>	shortfin mako	146.0	0	0.5
Molidae	<i>Mola mola</i>	ocean sunfish	109.2	25	0.4
Monacanthidae	<i>Aluterus monoceros</i>	unicorn leatherjacket filefish	16.8	0	0.1
	Monacanthidae sp.	leatherjackets	6.6	10	0.5
Myliobatidae	<i>Mobula</i> sp.	mobula rays	100.0	0	0.1
Nomeidae	<i>Psenes cyanophrys</i>	freckled driftfish	3.3	6	0.9
Scombridae	<i>Acanthocybium solandri</i>	wahoo	117.8	0	0.3
	<i>Katsuwonus pelamis</i>	skipjack tuna	57.5	0	0.3
	<i>Thunnus maccoyii</i>	southern bluefin tuna	108.1	20	0.7
	<i>Thunnus</i> sp.	tunas	48.4	0	1.1
Sphyraenidae	<i>Sphyraena</i> sp.	barracudas	83.1	0	0.2
Sphyrnidae	<i>Sphyrna mokarran</i>	great hammerhead	264.8	0	0.1
Syngnathidae	Syngnathidae sp.	pipefishes	14.4	0	0.1
		larval fish	2.0	16	13.7

[†] Average fork length of species observed over all surveys.

[‡] Percentage of redundancy calculated as the mean percentage of individual rigs on which a species was observed by longline.

[§] Percentage of total abundance (%TA) calculated as the grand mean of the average total abundance by longline.

Table 3. 3 Results of one-way PERMANOVAs testing the effect of survey on total taxa, mean taxa richness, mean total abundance, and composition, quantified as mean abundance by species as the average of the rigs by longline. Note the difference in sample size for length and composition reflects exclusion of longlines where no fish were present.

Model	df	SS	MS	Pseudo-F	P(perm)	perms
Total Taxa						
Survey	3	13.2	4.39	1.48	0.24	661
Res	42	124.7	2.97			
Total	45	137.8				
Mean Taxa						
Survey	3	4.3	1.43	3.58	0.022	999
Res	42	16.7	0.40			
Total	45	21.0				
Log10(TA+1)						
Survey	3	0.93	0.31	3.25	0.043	999
Res	42	4.00	0.09			
Total	45	4.91				
Log10(TB+1)						
Survey	3	0.84	0.28	0.11	0.96	999
Res	42	103.8	2.47			
Total	45	104.6				
Log10(FL)						
Survey	3	0.83	0.28	1.15	0.36	997
Res	36	8.60	0.24			
Total	39	9.43				
Composition						
Survey	3	37735	12578	5.87	1.00E-05	91502
Res	36	77089	2141			
Total	39	1.15E+05				

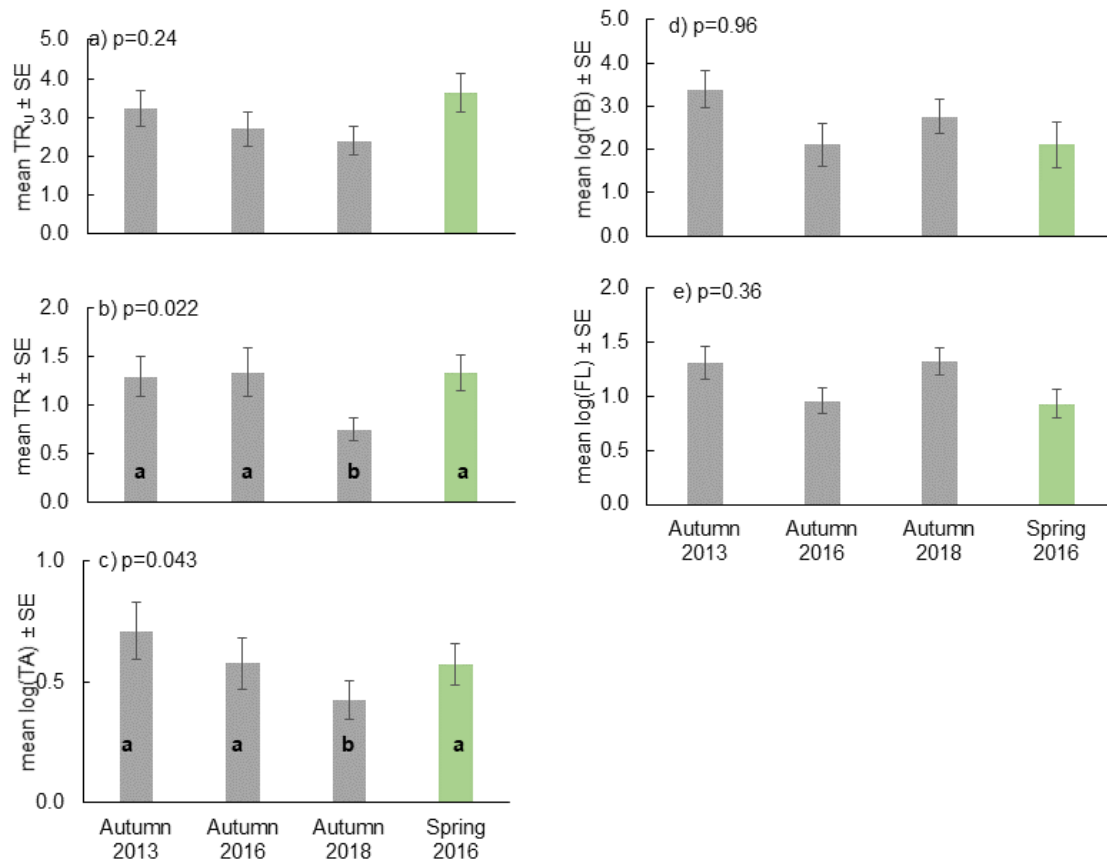


Figure 3.4 Variation in **(a)** total unique taxonomic richness (TRU), **(b)** mean taxonomic richness (TR), **(c)** mean log₁₀ total abundance (TA; no.), **(d)** mean log₁₀ total biomass (TB; kg), excluding a single minke whale *Balaenoptera acutorostrata* in the autumn 2016 survey and **(e)** mean log₁₀ fork length (FL; cm) by survey. Bars labeled with a 'b' highlight surveys that are significantly different from the others labeled with an 'a', if there are no labels then there is no significantly different surveys.

Dispersion tests indicated homogeneity of variance in taxa composition for pairwise tests of the 2013 and 2016 surveys ($p=0.77-0.87$), however the 2018 autumn survey had greater dispersion than the other surveys ($p<0.001$). Subsequent PERMANOVA indicated that survey affected taxa composition (**Table 3.3**), with *post-hoc* tests indicating that taxa composition varied significantly among all surveys ($p<0.0001$). The autumn 2013 and 2016 surveys are characterized by small scads and pilot fish, with autumn 2016 also having wahoo present. The 2018 autumn survey was characterized by small jacks, unicorn leatherjackets and pompano dolphinfish and was more similar to the other autumnal surveys than the spring 2016 surveys which was only characterized by small jacks (**Fig. 3.5**).

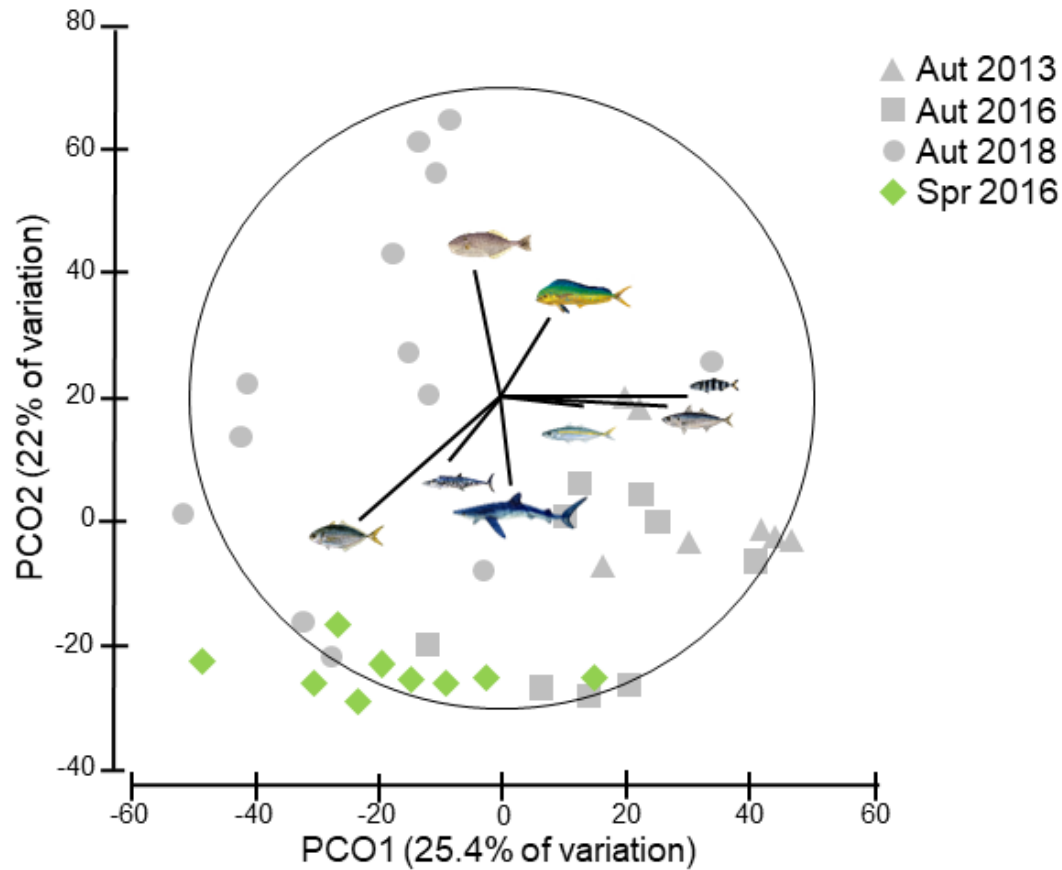


Figure 3. 5 Principal coordinates analysis indicating differences in taxonomic composition between the autumn (Aut) and spring (Spr) surveys based on a square root transform of abundance data and Bray Curtis similarity matrix. Species separating assemblages clockwise from the top are the unicorn leatherjacket *Aluterus monoceros*, common dolphinfish *Coryphaena hippurus*, pilotfish *Naucrastes ductor*, small scads *Decapterus* sp., yellowtail scad *Atule mate*, blue shark *Prionace glauca*, wahoo *Acanthocybium solandri*, and small jacks *Carangidae* sp. (Illustrations courtesy of anima.net.au).

Spatial comparisons of model predictions between the four surveys provide evidence for interannual stability but seasonal variability in the occurrence patterns of marine wildlife throughout the study area (**Fig. 3.6**). The SSIM values indicate that, in the Austral autumn, there are strong and positive correlations among the spatial distributions of wildlife across years (**Supporting Information**). In contrast, significant seasonal variation exists between the Austral autumn and Austral spring in 2016 with an inverse relationship between areas of high and low probability of wildlife presence. The relationships between the spring survey in 2016 and the autumn 2013 and 2018 surveys were weak.

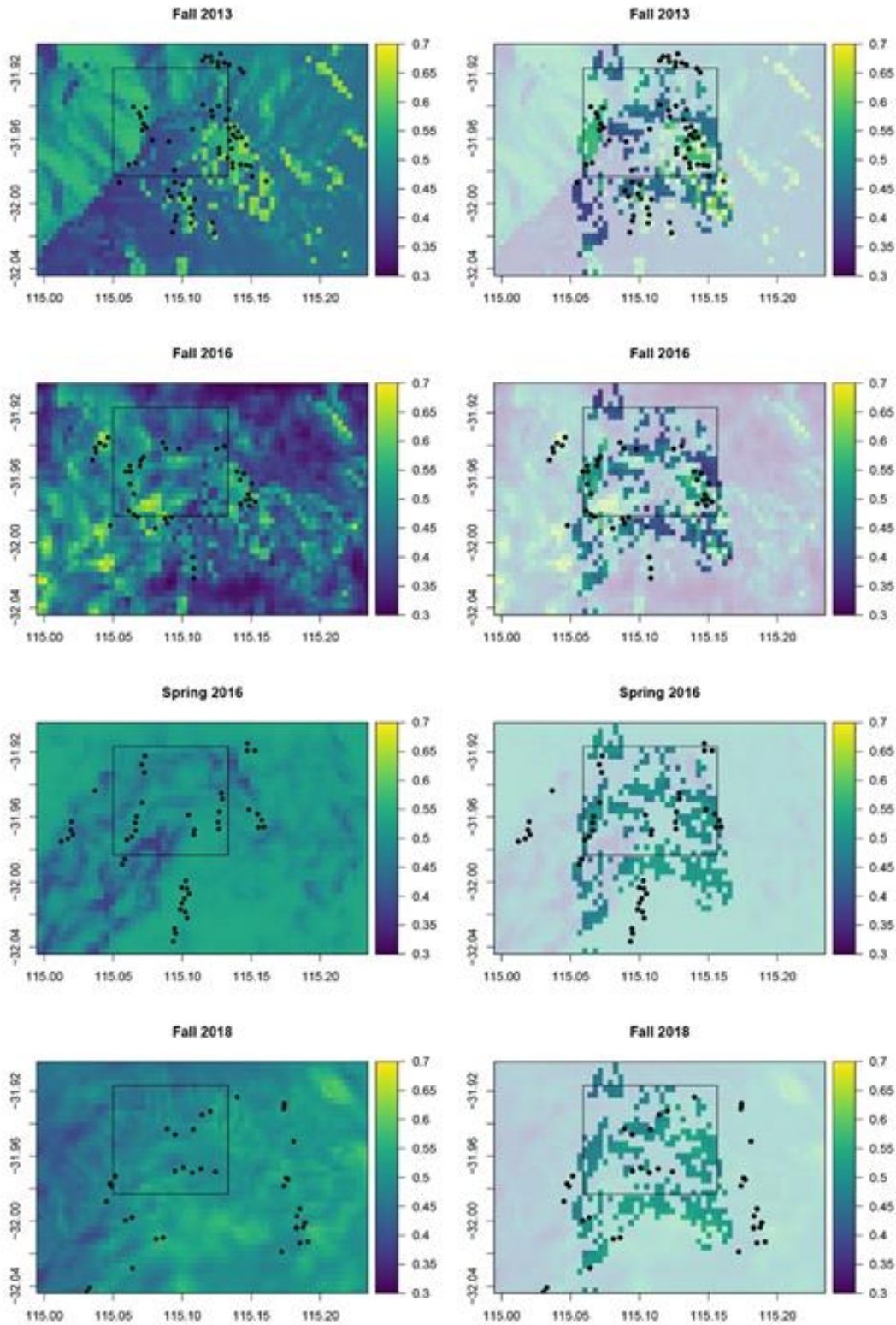


Figure 3. 6 Predicted distribution of pelagic wildlife relative to habitat suitability based on conditions of best model prediction within analogue environmental conditions (left panels) and for the entire study area (right panels).

3.5 DISCUSSION

3.5.1 THE PERTH CANYON MARINE PARK: A STABLE AGGREGATION AREA

It has been argued that “static” pelagic marine parks are poorly suited to conserving oceanic taxa, primarily due to the vagility of marine species and their tendency to associate with dynamic ocean features such as thermal fronts (Hobday 2010, Kaplan et al. 2010). However, the Perth Canyon has already been recognized as a predictable and “static” aggregation area for marine mammals (Rennie 2005). What has remained unclear is the degree to which the distribution of marine wildlife more generally is stable in space and time, providing support for the establishment of a marine park at this location. Over five years, we documented the distribution and abundance of a wide range of functional groups from fish larvae to forage taxa and apex predators including marine mammals. Our results show that the number of unique taxa, total biomass and mean fork length was stable among years, noting that only a single spring survey has been completed to date. The spatial distribution of wildlife was positively correlated across the three autumn and reversed in the single spring survey completed. Such relatively small-scale stability at the Perth Canyon has also been observed at continental scales, where for instance, fish catches are spatially predictable over decades and frequently associated with undersea features (Bouchet et al. 2015, Bouchet et al. 2017). Despite the stability in the number of unique taxa, biomass and size, the composition of the wildlife assemblage varied among all surveys, suggesting that the presence of specific taxa is difficult to predict. Only small scads, pilotfish and common dolphinfish were observed on all surveys. Our observations are consistent with recognized temporal patchiness in marine environments where, for instance, temporal variation in predator presence is driven by the patchiness of their prey (Ainley et al. 2017).

We also observed significant differences in mean taxonomic richness and mean abundance of animals with the autumnal 2018 survey being lower by approximately 50% relative to all other surveys. One possible explanation is that, with the park’s declaration in 2012 and then subsequent suspension of management in 2013 by the incoming government, “a race to fish” may have occurred prior to enforcement of on-water management on 1 July 2018 (McDermott et al. 2019). The suspension of the Perth Canyon Marine Park’s management plan, along with those for the majority of other Australian Marine Parks, also resulted in rezoning such that the former IUCN II zone within and around which this research was conducted is now IUCN IV and offers only seabed protection whilst the water column remains open to recreational and commercial fishing. Given the popularity of the canyon head to recreational and charter fishing (Fletcher et al. 2018), increased fishing pressure represents a highly plausible albeit speculative explanation for the observed results. We note

that such behaviour has been documented at the Phoenix Islands Protected Area with the authors suggesting that if such races to fish are consistently triggered by announced protection, the percentage of overexploited fisheries will increase from 65% to 72% globally (McDermott et al. 2018). As such, races to fish have both consequences for conservation and our understanding of subsequent benefits of protection of marine wildlife that has experienced higher than expected exploitation in the short-term.

Patterns in the reproduction, recruitment, and movements of oceanic wildlife taxa are often strongly seasonal (Rogers et al. 2015, Beaugrand & Kirby 2018). However, in terms of the number of unique taxa, and mean taxonomic richness, abundance, biomass and length, our results show no seasonal difference within 2016, or indeed between the spring of 2016 and autumn of 2013. We had expected higher abundance and biomass in the Austral autumn given the higher productivity of this period (Koslow et al. 2008). It is possible that particular species are strongly seasonal and that this seasonality is masked by species preferring opposite seasons and effectively swapping out (Luckhurst & Trott 2000, Kai et al. 2017). This is supported by the relatively strong separation of the spring season from the majority of the autumn samples. Additionally, the spatial analysis shows an inverse albeit weak correlation between autumn and spring 2016 yet stronger correlations between autumns of each survey year. Such locational differences may reflect seasonal oceanographic processes interacting with topography (Rennie et al. 2007).

The opportunities for foraging and reproduction at this canyon head are well documented in terms of upwelling and its high productivity relative to the surrounding seascape (Rennie et al. 2009, Forrest et al. 2019). The current study adds to this understanding, in terms of the high number of larval fish that were nearly twice as common and over twice as abundant as comparable surveys at other locations globally (JJ Meeuwig, University of Western Australia, 2019, unpubl. data). These animals, less than 2 cm in length, are likely dependent on the relative concentration of plankton at the canyon and are themselves likely prey for the high numbers of small clupeids and carangids that we observed.

3.5.2 CONSERVATION IMPLICATIONS

There are two main conservation implications of this research. First, the utility of mid-water BRUVS that non-destructively and robustly document pelagic wildlife allows us to demonstrate their spatial and temporal patterns in relation to protection. We were able to demonstrate that there is very little redundancy between rigs across a given longline. Whilst not truly independent (*sensu* Hurlbert 1984), little spatial autocorrelation appears to exist between rigs. Moreover, the species accumulation curve of deployments combined across all surveys and those for the individual surveys, with the exception of the depauperate 2018

autumn survey, were consistent in suggesting that 50-75 samples capturing 50% of the species (Keating & Quinn 1998) and indeed, of the approximately 35 species in our taxa pool, individual surveys capture about 15. This is analogous to the use of accumulation curves to quantify endemism (Von Der Heyden 2011) and anthropogenic impacts (Novaglio et al. 2016). Our results also confirm that higher taxa resolution is not an obstacle to the use of species accumulation curves (Bevilacqua et al. 2018) (**Supp. Fig 3.1**). Documenting the diversity and abundance of wildlife is critical to managers and decision-makers who need to determine whether protection has delivered conservation outcomes; our results demonstrate that this method can cost-effectively provide this information, including at a national level. Given that the Perth Canyon is but one MPA within Australia's network of 44 MPAs, such cost-effective methods are essential.

Second, our results suggest that patterns in wildlife are sufficiently stable at this canyon head to generate key conservation outcomes, if protected effectively. Such conservation outcomes, analogous to those from shallow-water MPAs include ecological benefits such as increased diversity, fish abundance and biomass, ecological resilience as well as economic benefits to fisheries and tourism. That the Australian government suspended the network in 2013 such that the IUCN II zones were unprotected means a delay of six years in delivery of conservation benefits. Moreover, as we observed an approximate halving of fish abundance between 2013 and the three later surveys in 2016 and 2018, we may have witnessed a "race to fish" prior to implementation of protection. The placement of an IUCN II zone at the head of the canyon caused considerable consternation among the recreational and charter fishing sectors. The risk of an ultimate "lock up" may have encouraged some fishers to take advantage of the six-year review process undertaken by the Government to access the resources. Indeed, the charter sector catch in the Western Bioregion has increased by 12% between 2012/2013 and 2015/2016 (Fletcher et al. 2017, 2018), suggestive of such behavior.

There is growing evidence that the health of many of its fisheries is deteriorating, with, for example, Edgar et al. (2018) concluding that catches from wild fisheries have decreased 36% in the last decade. Moreover, Roff et al. (2018) reported declines of 74 to 92% in catch per unit effort for the hammerhead, tiger *Galeocerdo cuvier* and white *Carcharodon carcharias* shark in Queensland over the past five decades. Given these alarming statistics, it could be argued that more rather than less protection should be given to Australia's habitats and species of conservation concern (Barr and Possingham, 2013). Yet, in 2018, the decision was taken to remove 400,000 km² of IUCN II protection from its national network, downgrade the Perth Canyon's most land-ward canyon head, which has the strongest upwelling and known aggregations of wildlife, from IUCN II to IUCN IV, and move

the IUCN II zone to a smaller canyon head approximately 40 km further offshore. The report noted that the zoning changes would “improve access to important fishing areas” for both recreational and charter fishers (Director of National Parks 2018), suggesting that this may be an example of “residualism” (Devillers et al. 2015). Moreover, the conservation equivalency of these two areas remains unknown, confirming both the need for the science community to demonstrate the outcomes of protection in offshore MPAs and better understanding of the distribution of wildlife throughout these MPAs. Marine protected areas (MPAs) are increasingly being established as part of global efforts to halt and reverse observed declines in ocean wildlife (Koldewey et al. 2010, O’Leary et al. 2011) and build resilience (Roberts et al. 2017), with acceptance in the recreational and commercial fishing communities is also on the rise (Wolff 2015, Navarro et al. 2018). The Australian Marine Park network extends over approximately 30% of Australia’s EEZ and is a mosaic of highly protected (IUCN II), habitat management (IUCN IV), and multiple use (IUCN VI) areas with the latter two zones allowing a range of extractive activities such as fishing and oil and gas exploration. However, IUCN II zones comprise less than 10% of Australia’s EEZ, with many of these zones likely “residual” in nature (Devillers et al. 2015), despite comprehensive, adequate and representative protection being a key policy plank of the Australian government (NSES:AGPS 1992). As Australia’s marine park network is reviewed, as is required under the *Environmental Protection and Biodiversity Conservation Act 1999*, research such as that presented here should help inform future decisions to support more effective conservation outcomes.

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3.9 SUPPLEMENTARY MATERIALS

Supplementary Table 3. 1 Metadata for the mid-water stereo-BRUVS seasonal surveys at the head of the Perth Canyon.

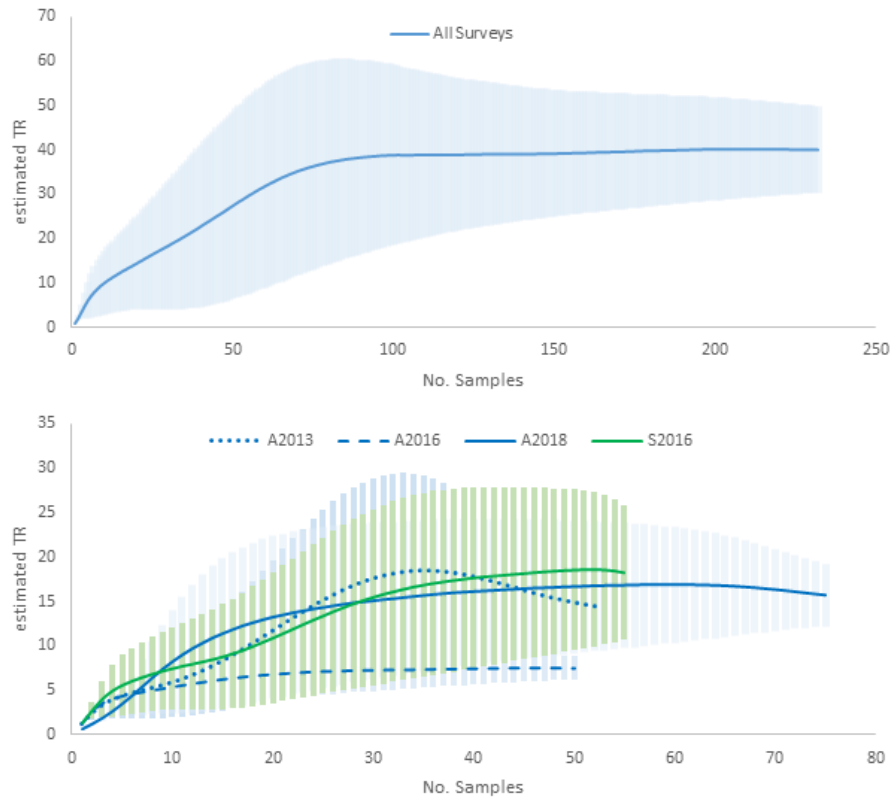
Survey	Start Date	End Date	Number of LL†	Number of drops‡
Autumn 2013	30/03/2013	17/05/2013	10	52
Autumn 2016	27/04/2016	4/05/2016	10	50
Autumn 2018	17/04/2018	12/05/2018	15	75
Spring 2016	25/11/2016	2/12/2016	11	55
			46	232

† Longlines of mid-water stereo-BRUVS.

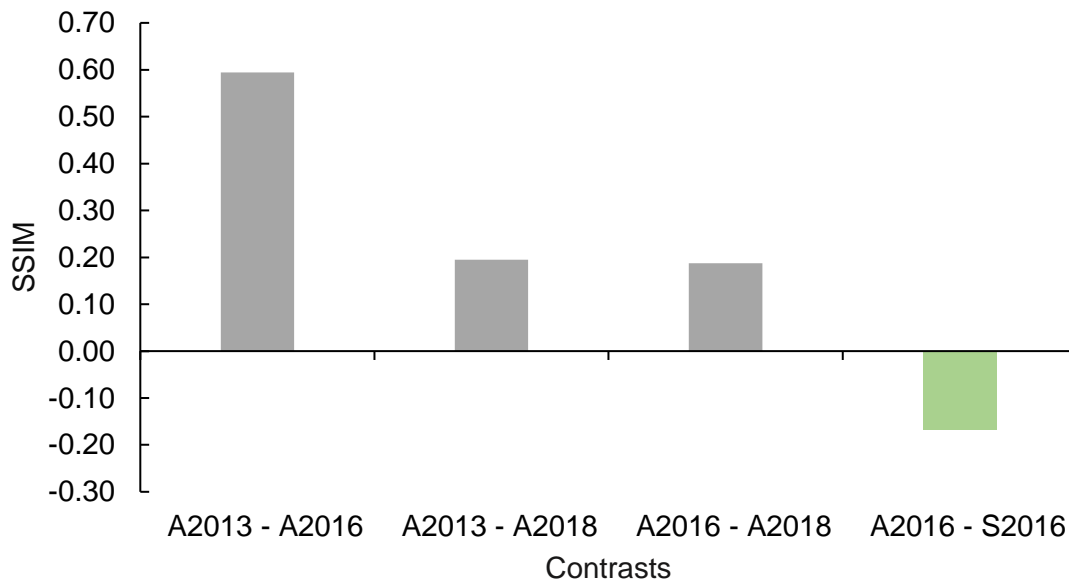
‡ Individual camera units deployed.

Supplementary Table 3. 2 Number of longlines on which a taxon was observed on a given number of rigs.

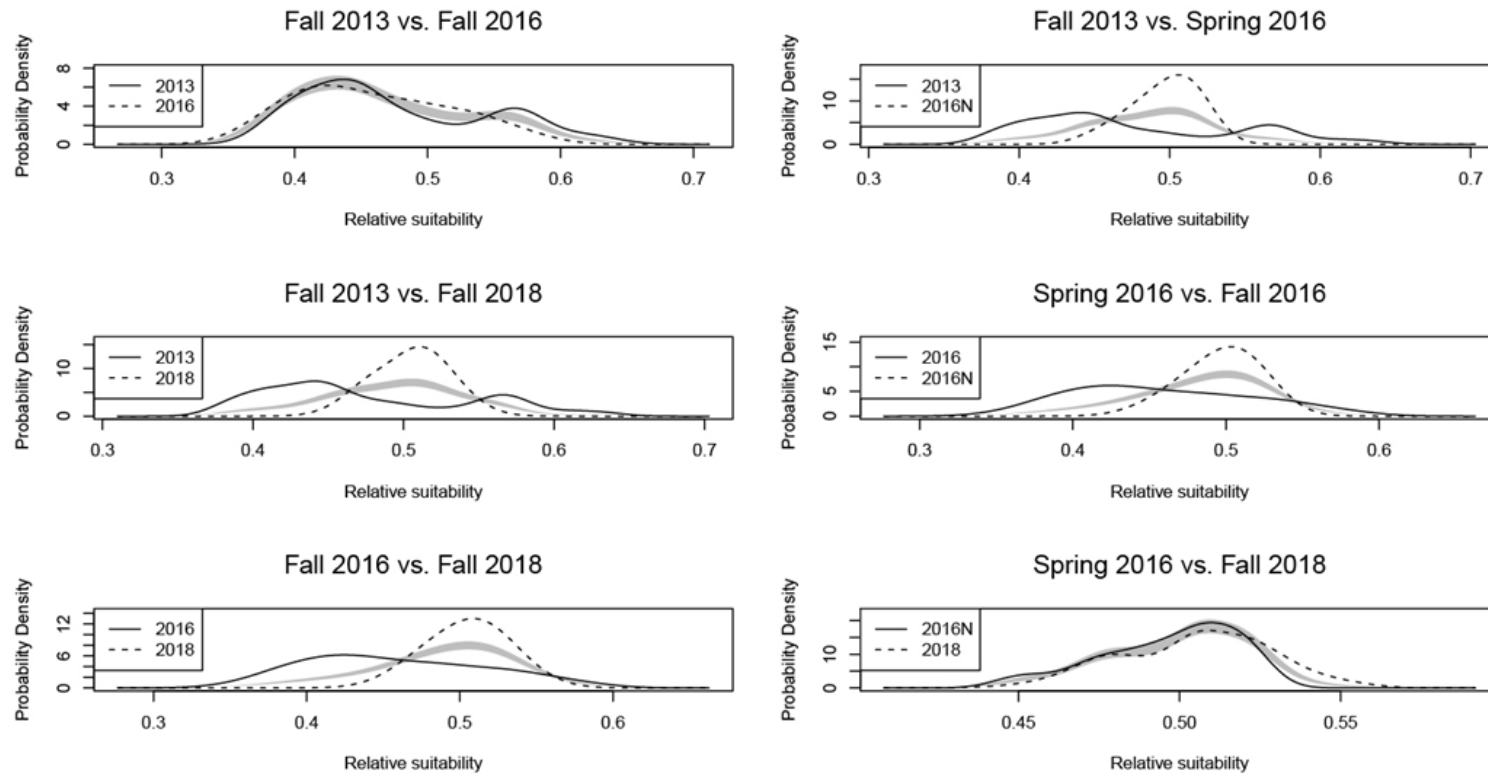
Scientific name	Common name	Number of rigs				
		1	2	3	4	5/6†
<i>Balaenoptera acutorostrata</i>	dwarf minke whale	1
<i>Atule mate</i>	yellowtail scad	1
Carangidae	jacks	8	2	3	1	2
<i>Decapterus muroadsi</i>	amberstripe scad	3	1
<i>Decapterus</i> sp.	scads	6	6	8	3	4
<i>Naucrates ductor</i>	pilotfish	9	...	3	2	...
<i>Carcharhinus brachyurus</i>	copper shark	...	1
<i>Prionace glauca</i>	blue shark	1
<i>Coryphaena equiselis</i>	pompano dolphinfish	...	1
<i>Coryphaena hippurus</i>	common dolphinfish	8	3	1	1	...
<i>Echeneis naucrates</i>	live sharksucker	1
<i>Remora remora</i>	shark sucker	2
<i>Fistularia</i> sp.	cornetfishes	1	1
<i>Istiompax indica</i>	black marlin	1
<i>Isurus oxyrinchus</i>	shortfin mako	4
<i>Mola mola</i>	ocean sunfish	...	1
<i>Aluterus monoceros</i>	unicorn leatherjacket filefish	1
Monacanthidae	leatherjackets	1	1
<i>Mobula</i> sp.	mobula rays	1
<i>Psenes cyanophrys</i>	freckled driftfish	2	1
<i>Acanthocybium solandri</i>	wahoo	2
<i>Katsuwonus pelamis</i>	skipjack tuna	1
<i>Thunnus maccoyii</i>	southern bluefin tuna	...	2
<i>Thunnus</i> sp.	tunas	1
<i>Sphyrna</i> sp.	barracudas	1
<i>Sphyrna mokarran</i>	great hammerhead	1
Syngnathidae	pipefishes	1
-	larval fish	10	6	6



Supplementary Figure 3. 1 Species accumulation plot for all surveys (n=268 deployments) indicating the relationship between number of samples obtained, n, and the estimated taxonomic richness (TR) for a given number of samples, S(n).



Supplementary Figure 3. 2 Structural similarity (SSIM) index values for contrasts among years in autumn (2013, 2016, and 2018) and seasons (autumn 2016 and spring 2016). SSIM values vary between 1 and -1 with the values indicating perfect positive and negative correlation, respectively. Seasons for each year are indicated by “A” for autumn and “S” for spring



Supplementary Figure 3. 3 Distribution of suitability values for the Austral autumn 2013 and 2016 (left panels) and the Austral autumn and spring in 2016 (right panels); grey bands indicate null models of no difference between the pair of kernel density estimates.

CHAPTER 4: HUMAN IMPACTS OVERWHELM NATURAL PATTERNS IN PELAGIC VERTEBRATE BIOGEOGRAPHY

Key words: macroecology, Indian Ocean, Latitudinal Diversity Gradient, Bergmann's Rule, pelagic, marine protected areas

4.1 ABSTRACT

Our oceans are being transformed at a staggering rate, with “peak” fish reached in 1996 and global catches since declining. Pelagic sharks and bony fishes (henceforth pelagic fishes) are amongst the world's most exploited vertebrates, with marine mammals and marine reptiles as unintended bycatch. These declines are occurring whilst we are still learning about the biogeography of pelagic wildlife and the degree to which, for instance, ecological theory grounded in the Latitudinal Diversity Gradient and Bergmann's Rule apply to these taxa. As the eastern Indian Ocean is considered to be one of the least exploited oceanic regions, we documented the diversity, abundance, size and biomass of pelagic vertebrates at 15 locations along nearly 23° of latitude of the coastline of Western Australia. We also collated data on seabed morphometry, water column characteristics and human proximity. With 57,634 individuals recorded from 147 taxa, the most important predictor of pelagic wildlife was the size of the nearest human population centre, with the expected negative correlation. Whilst geomorphology and environmental characteristics were present in the model, they explained relatively little of the variance. We suggest that pelagic wildlife may well follow expected biogeographical patterns as predicted by the Latitudinal Diversity Gradient and Bergmann's Rule, but that the human footprint effectively flattens expected latitudinal trends and drives down wildlife abundance where humans are most numerous. That pelagic fish assemblages associated with a relatively “pristine” coastline clearly show a human signature argues for expansion of the partially-protected MPAs currently in place throughout Australia's western EEZ into larger, highly protected MPAs.

4.2 INTRODUCTION

Human activities in the Anthropocene are transforming oceans at unprecedented rates and at global scales (McCauley et al. 2015, Payne et al. 2016). A combination of the presentist bias (Zeller & Pauly 2018) and shifting baselines make understanding this transformation challenging. Reconstructed fisheries catch shows that peak fish catch was reached in 1996, with global marine catches now declining annually by approximately 920 thousand tonnes, or ~0.8% (Pauly & Zeller 2016). In Australia, debate continues over the state of its oceans with, for instance, Edgar et al. (2018) concluding that the biomass of large (≥ 20 cm total length) fishes declined 36% in the last decade on Australian reefs. While some studies then questioned the relevance of ecological field surveys to stock assessments (Gaughan et al. 2019, Little et al. 2019), Edgar et al. (2018) responded that even the best stock assessments are often underpinned by limited data and characterised by large error margins. Similar debates play out between survey- and fisheries-based scientists globally and, importantly, these differences then hamper decision-makers from justifying the conservation of declining fish populations.

Decision-making in marine conservation is underpinned by understanding of large-scale biogeographical patterns in fish assemblages (Stuart-Smith et al. 2013). This is particularly true in the case of MPAs, where a knowledge of biogeography helps not only to ensure that all habitats and species are represented within protected area networks (Kendall et al. 2015) but also to prevent “residual” MPAs, areas of the ocean that are simply unattractive to extraction-based businesses and not necessarily in need of protection, from being established (Devillers et al. 2015). A key driver of biogeographical patterns in species distributions is latitude, with species richness generally increasing towards the equator, a pattern known as the “Latitudinal Diversity Gradient” (LDG) (Forster 1778). Although the drivers of the LDG remain debated (Brown 2014), its generality has been confirmed in both terrestrial and marine systems (Hillebrand 2004, Kinlock et al. 2018), with Menegotto and Rangel (2018) demonstrating that documented “dips” in marine species richness near the equator (e.g. Chaudhary et al. 2016) appear to be due to sampling bias.

Bergmann’s Rule (Bergmann 1848) is another well-established biogeographical “rule of thumb.” It was originally formulated to reflect the underlying physiology which dictates that endothermic vertebrates increase in size towards the poles (Van Voorhies 1996, Fortes & Absalão 2004). It has since been observed that ectotherms including worms, molluscs and over 6000 species of marine fish show patterns in their size distributions consistent with Bergmann’s Rule (Van Voorhies 1996, Fortes & Absalão 2004, Fisher et al. 2010a, Fernández-Torres et al. 2018) although this does not reflect the original mechanism

espoused by Bergmann. Rather, latitudinal trends in fish body size are likely driven by increased metabolic demand of ectotherms in, and the lower oxygen solubility of warmer water (Pauly 1981, Pauly 2010), of interest given the overarching effects of a warming ocean on fish biodiversity and productivity (Cheung et al. 2009, Cheung et al. 2010, Lotze et al. 2019).

The pelagic realm, which contains 99% of the volume of the biosphere or over one billion km³ (Angel 1993, Webb et al. 2010), is arguably the largest ecosystem on the planet. Defined as that part of the ocean that is not directly in contact with a shore or the sea floor, the pelagic zone contains a diversity of marine mammals, sharks, tunas, billfishes and the small forage fishes that support to predators. Pelagic ecosystems provide over 80% of the fishes eaten by humans (Pauly et al. 2002) and are at significant risk from anthropogenic activities (Halpern et al. 2008, Williams et al. 2011, Tournadre 2014, McCauley et al. 2015). Yet pelagic animals are also notoriously difficult to study due to the vastness of the three-dimensional pelagic zone, the high mobility of many pelagic species and their relative scarcity within the environment (Game et al. 2009). One solution to these challenges has recently been provided by innovations in small “action” camera technology (Letessier et al. 2015) which, due to their cost-effectiveness and ability to sample non-destructively, have enabled sampling programs at scales and locations that were previously unachievable (Letessier et al. 2017). When combined with environmental datasets derived from remote sensing and marine habitat mapping, this approach can accelerate our understanding of elusive pelagic species.

To understand the biogeography of pelagic species and the degree to which it is consistent with the LDG and Bergmann’s Rule, we used mid-water stereo Baited Remote Underwater Video Systems (stereo-BRUVS) to document species diversity, abundance, size and biomass, with 29 surveys conducted across 15 locations in the Indo-Pacific and covering 23° of latitude. This study extends on the analyses of Tittensor et al. (2010) that found patterns in species richness of sharks and tunas/billfishes consistent with the LDG and Worm et al. (2003) that largely found peaks in pelagic species richness at mid-latitudes. We predicted that pelagic species diversity would increase at lower latitudes, whereas body sizes would decline. The potential influence of seabed and water column characteristics, which can influence fish assemblages (Bouchet et al. 2015), and anthropogenic factors (Letessier et al. 2019) were also considered.

4.3 METHODS

4.3.1 MID-WATER STEREO-BRUVS

Mid-water stereo-BRUVS are an adaptation of seabed stereo-BRUVS, a well-established method used to document the structure of reef fish assemblages (Cappo et al. 2006) and represent a standardised and non-destructive method to characterise the abundance (Bouchet & Meeuwig 2015, Letessier et al. 2017) and behaviour (Turner 2016) of highly mobile pelagic species. Mid-water stereo-BRUVS rigs consist of a 1.45 m-long stainless steel frame that supports two high-definition action cameras. The frame is mounted on a cross bar 95 cm in length, perpendicular to a 180 cm-long metal rod that supports a 45 cm long perforated PVC pipe “bait chamber” containing 1 kg of the pilchard *Sardinops sagax*. The container ensures the slow release and diffusion of bait into the water column throughout the duration of the deployment and acts as a rudder to minimise rotational movement of the unit, such that a down-stream field of view is maintained. The cameras on the frame are 80 cm apart, inwardly converging at an angle of 8 degrees, with each camera set to record in medium field of view to maximise the area captured in the video frame and improve detection rates up to a distance of ca. 10 m. Individual stereo-BRUVS were deployed in long-line “strings” of five rigs or in moored sets of five, depending on the vessel size and conditions, at a depth of 10 m, with each rig separated by 200 m of line. Rigs were deployed for a minimum of two hours soak time.

4.3.2 DATA COLLECTION

We deployed 2,760 individual stereo-BRUVS rigs between 2012 and 2018. All surveys occurred in the eastern Indian Ocean and the Southern Ocean, between the latitudes -12.00° and -34.84° (Fig. 4.1). All sampling was undertaken during daylight hours between 7:00 and 17:00 to minimise any effects of crepuscular behaviour in fish (Axenrot et al. 2004, Birt et al. 2012). Sampling was stratified across target features such as islands (the Cocos (Keeling) Islands, Ashmore Reef, Long Reef, the Rowley Shoals, the Montebello Islands, and the Recherche Archipelago), canyons (Ningaloo, Perth Canyon and Bremer Canyon), shelf breaks (the Argo-Rowley Terrace and Gracetown) and coastal bays (Shark Bay, Geographe Bay). Sampling took place within or adjacent to Australian Marine Parks with the exception of Long Reef, which is in a Western Australian state marine park. Within each MPA, sampling occurred in a range of zones from IUCN Ia, IUCN II, IUCN IV and IUCN VI. Surveys were conducted from multiple vessels under UWA ethics permit RA/3/100/1484 and in the case of private vessels, under exemptions from the Australian Marine Safety Authority (EX2016/A185; EX2017/A007A, AU-COM2012-170, AU-COM2018-426, PA2018-00036-1, PA2018-00091-1, PA2018-00091-2, PA2018-00079). Research permits were also obtained (DPAW 01-000049-4, DPAW 01-000049-7, DPAW 01-000049-8, CMR-17-000526, CMR-16-

000426, CMR-18-000550 and Fisheries Exemption Numbers 2853 and 3172).

4.3.3 DATA PROCESSING AND TREATMENT

Stereo-BRUVS were calibrated prior to field work in an enclosed pool, using software CAL (SeaGIS Pty Ltd) and following standard protocols that allow length measurements (Harvey & Shortis 1998). Videos were imported into the EventMeasure software package (SeaGIS Pty Ltd <http://www.seagis.com.au/>) for processing and synchronised in the lab based on a slow hand clap recorded immediately before deployment of each rig in the field. Processing commenced once the stereo-BRUVS had stabilised at a depth of 10 m and ran for a total of 120 minutes. All observed animals were identified to the lowest taxonomic level possible, with relative abundance estimated as the maximum number of individuals of a given species in any frame (Cappo et al. 2006). Fork length (cm) was measured in stereo.

The video analysis generated taxonomic identifications, abundances and length measurements for the observed animals for each rig deployment. Because the stereo-BRUVS were deployed in longline formation, all attributes were reported as the mean value for the five rigs in a given longline, or “string”. Taxonomic Richness (TR), the number of taxa on each deployment, summed and averaged for the string, was calculated. Total Abundance (TA) was similarly calculated as the sum of all abundances across all taxa on a deployment and then averaged for the string. Mean fork length (FL) was calculated as the sum of all FL divided by the number of individuals per deployment and then averaged for a given string. Where no FL was available for a given deployment due to obstructed view of the fish or where the fish was too far away from the cameras to allow an accurate estimate, the FL for that taxa measured elsewhere on the string was used. Failing this, a FL was used from the nearest string or the survey as a whole. To calculate total biomass, taxon-specific length weight relationships (LWR) based on FL were sourced from FishBase (Froese & Pauly 2019; Appendix 4) to estimate mean fish weight by taxa. Where a LWR based on FL was not available, the relationship based on total length (TL) was used, with species-specific TL:FL conversions applied. Where individuals were identified to genus or family only, the Bayesian LWR based on TL was used as provided by FishBase. The mean weights for each taxa were then multiplied by the abundance of that taxa to estimate taxa-specific biomass. Biomass was based only on the weights of fishes and marine reptiles as marine mammals were orders of magnitude heavier than even the largest of fish and would have skewed the results.

To understand the relationships between attributes of the fish assemblage and environmental variables, a database of seabed morphometry and water column chemistry was compiled, following Bouchet et al. (2018). A large number of geophysical and

oceanographic variables (**Supplementary Table 4.8.3**) have been extracted for each sampling location and used in the statistical analysis of this study. Each such sampling location is represented as a rectangle which is a 200 m buffer around the minimum rectangle covering the five rigs of the mid-water stereo-BRUVS deployment. The majority of these habitat variables were derived from the following six main datasets:

- Geoscience Australia (GA) 250 m bathymetry (Whiteway 2009);
- GA Australian submarine canyons (Huang et al. 2014);
- GA marine sediment prediction (Li et al. 2010, Li et al. 2012);
- CSIRO Atlas of Regional Seas (CARS; (Dunn & Ridgway 2002, Ridgway et al. 2002);
- Australia's Integrated Marine Observing System (IMOS) Moderate Resolution Imaging Spectroradiometer (MODIS); and
- Hybrid Coordinate Ocean Model (HYCOM) (Bleck 2002).

Geoscience Australia's 250 m bathymetry data and its morphometric derivatives represent the water depth, the topographic complexity and the habitat heterogeneity of the benthic environment. These variables have been identified as important predictors of pelagic species (Bouchet et al. 2015). Submarine canyons are often recognised as biodiversity hotspots (Huang et al. 2018 and references therein). The metrics generated from GA's Australian submarine canyons dataset have been shown to be suitable surrogates and important predictors for pelagic species (Bouchet et al. 2017, Huang et al. (2018)). The several variables derived from the CARS dataset represent the water chemistry and nutrient state. These variables such as temperature, oxygen and salinity were considered as potentially important drivers of pelagic species (Pauly 1981, Pauly 2010, Cheung et al. 2013, Bouchet et al. 2017). The satellite-derived Chlorophyll-a variable from the IMOS MODIS data is an indicator of ocean primary productivity and as such represents an important predictor of pelagic species (Platt and Sathyendranath 2008, Bouchet et al. 2017). The water temperature, represented by the satellite-derived SST variable from the IMOS MODIS data, is known to strongly influence the distribution and migration patterns of pelagic species (Cheung et al. 2013, Klemas 2013, Bouchet et al. 2017, Payne et al. 2017). Bouchet et al. (2017) also identified ocean energy and current regimes as important predictors of pelagic fishes. In this study, several of these oceanographic variables were derived from the Hycom model outputs over the time period during which the survey occurred. In addition, sediment variables are useful surrogates of benthic species (McArthur et al. 2010, Huang et al. 2018). The trophic relationship between the benthic and pelagic species may render sediment

variables potential predictors of pelagic species. Therefore, several of these sediment variables were obtained for this study.

Distance to the source of material flux could be a suitable surrogate for the availability of the amount of nutrients and organic matter (Huang et al. 2018). Several such variables including 'distance to coast', 'distance to shelf break' and 'shelf or slope' were thus calculated for this study. Human impact on the pelagic species could not be ignored (Dunn et al. 2017, Juhel et al. 2019, Letessier et al. 2019). Therefore, attributes of human impact were used for this study, including distance to port and population size of aforementioned port at the nearest census before the survey (**Supplementary Table 4.8.3**). These variables are a proxy for fishing effort, noting that accurate estimates of vessels and efforts are not publically available for Western Australian fisheries.

4.3.4 STATISTICAL ANALYSIS

The first objective of the statistical analysis was to build a predictive model that relates univariate attributes of the fish assemblages with physical, environmental and human drivers at the level of "survey". Survey was chosen as the analytical unit because it indicates location but also provides information about variation in time when multiple surveys occurred at a given location. To look at large-scale biogeographical patterns across the study area, surveys were assigned to one of four regions, roughly based on definitions provided by the WA Department of Primary Industries and Regional Development (DPIRD) (Gaughan et al. 2019). The four regions were: (1) North (N; with latitudes of less than 17.5°), corresponding to DPIRD's "North Coast", with the exception of the Montebello Islands which was allocated to (2) Central (C; -17.5° to -30°), corresponding to DPIRD's "Gascoyne Coast", (3) Southwest (SW; -32° to -35°), corresponds to "West Coast", and (4) South (S; >33°), corresponding to the Southern Ocean or DPIRD's "South Coast". A one-way ANOVA was used to assess variation among all surveys and then among the four regions.

Predictive models were built using linear regression based on mean values for each survey. Linear variables were log₁₀ transformed to ensure homoscedastic variation (Zar 1999). Circular variables, such as aspect, were transformed to radians. Collinearity between independent variables was assessed, such that no two variables with a correlation coefficient >0.6 were included in the same model. Variables were grouped on the basis of whether they were physical, environmental or human, to test for these collinearities. The "BEST" model was used in the R package VEGAN (Oksanen et al. 2007) to determine variables combinations for models with 1 to 3 independent variables, noting that, with 29 surveys and using the guidance of 1:10 variables per sample (Harrell et al. 1984), the final models would be restricted to 3 independent variables. On identification of a suite of key

variables from the BEST model, we then used stepwise regression to determine the most parsimonious model. To ensure no “anchor” effect, whereby the first variable chosen might exclude other potentially explanatory variables, we also removed the first entered variable from the full suite of explanatory variables and reran the BEST process. The final model was then chosen on the basis of the maximum amount of variation in the dependent variable being explained (R²) and that any additional variables added significant improvement to prediction, based on Aikaki’s Information Criterion (AIC). Outliers in the regressions were identified as those data points that departed from the trend line by > 2 SD (Cousineau & Chartier 2010).

To understand how the taxonomic assemblage varied across this latitudinal gradient and which explanatory variables were important, multivariate analyses were completed on abundance and biomass of each taxa. Abundance and biomass were square-root transformed to minimise the influence of abundant taxa or those with high biomass. We calculated the Bray Curtis resemblance matrix as the basis for our analysis. Distance-based linear models were used to associate assemblage data with explanatory variables and these relationships were visualised using distance-based redundancy analysis (dbRDA).

4.4 RESULTS

Across the 29 surveys from 15 locations across 4 bioregions and two oceans that were included in this analysis (**Fig. 4.1**), we counted 57,634 individuals representing 147 taxa and 44 families, comprising 88.7 tonnes of marine wildlife, excluding marine mammals (**Supplementary Table 4.8.1**). There were 67 individual marine mammals from 7 taxa that were not included in this estimate of biomass and subsequent biomass analyses, observed at 7 of the 15 locations (**Table 4.1**).

Family	Binomial	Common name	COC	MNT	NIN	SHK	PC	GCT	BRE
Balaenopteridae	<i>Balaenoptera acutorostrata</i>	northern minke whale	-	-	2	1	3	2	-
	<i>Balaenoptera</i> sp.	rorquals	-	-	1	-	-	-	-
	<i>Megaptera novaeangliae</i>	humpback whale	-	4	2	-	-	-	-
Delphinidae	<i>Delphinus delphis</i>	common dolphin	1	-	-	-	3	-	-
	<i>Globicephala macrorhynchus</i>	short-finned pilot whale	-	-	-	-	-	-	2
	<i>Orcinus orca</i>	killer whale	-	-	-	-	-	-	1
Ziphiidae	Ziphiidae sp.	beaked whales	2	-	-	-	-	-	-

Table 4. 1 List of families and species (scientific and common names) and numbers of individual marine mammals observed at Cocos (Keeling) Islands (COC), Montebello Islands (MNT), Ningaloo (NIN), Shark Bay (SHK), Perth Canyon (PC), Gracetown (GCT) and Bremer Canyon (BRE).

Records ranged from a 0.4 cm driftfish *Psenes* sp to a 10.2 m humpback whale *Megaptera novaeangliae*, with the largest fish being a 3.7 m tiger shark *Galeocerdo cuvier*. Six families accounted for 94.2% of all fish recorded: the leatherjackets (Monacanthidae; 38%), jacks (Carangidae; 33.4%), herrings (Clupeidae; 10.1%), driftfishes (Nomeidae; 6.6%), mackerels (Scombridae; 3.9%) and requiem sharks (Carcharhinidae; 1.8%). The 15 locations varied in physical and environmental characteristics and their level of remoteness from humans, both in terms of distance from and the size of the nearest population centres.

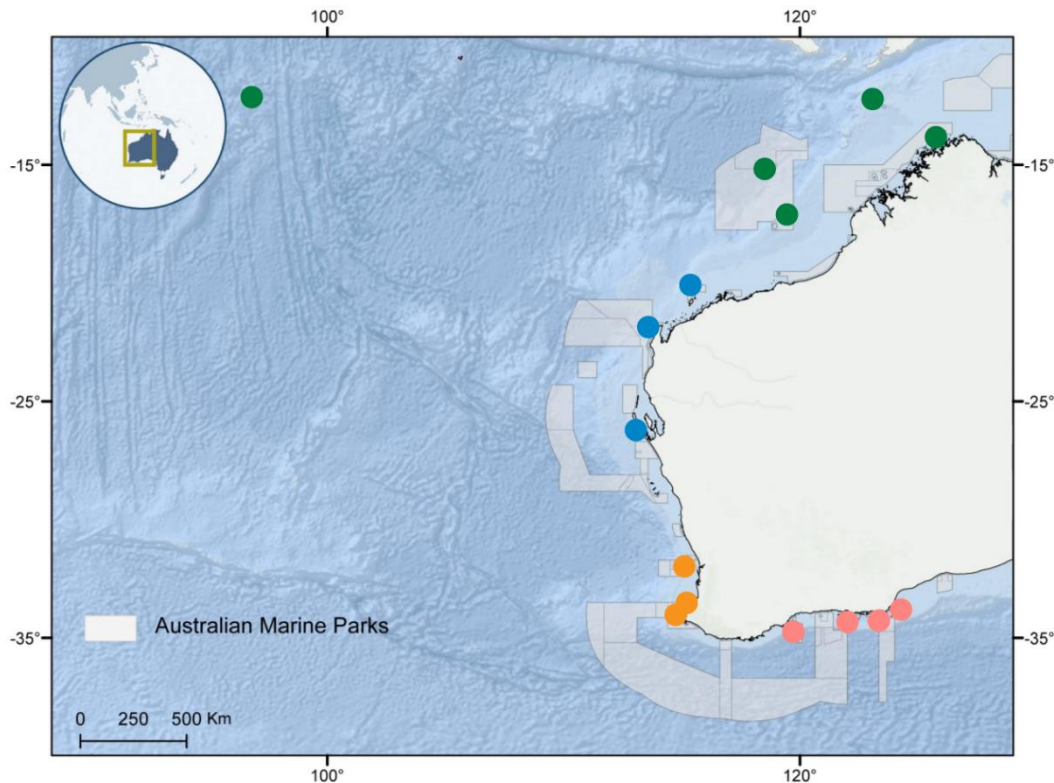


Figure 4. 1 Locations of surveys, ranging from the Cocos (Keeling) Islands in the far north to the Recherche Archipelago in the south. Northern sampling locations are indicated in green, central coast locations in blue, orange represents Perth-metro and the south-west, whilst locations bordering the Southern Ocean are indicated in pink.

Surveys differed significantly in TR, logTA, logTB ($p < 0.0001$ in all cases; **Fig. 4.2 left**). There were no clear latitudinal gradients from north to south in any of the variables. An analysis of surveys by region showed that, typically, the southwest region had lower TR, TA and TB values than adjacent regions (**Fig. 4.2 right**). Size of the nearest human population was the most significant factor affecting all attributes of the fish assemblage, explaining between 37-57% of the variation in the attributes of the fish assemblages (**Table 4.2; Fig. 4.3**). All relationships were negative, such that fish diversity, abundance and biomass decreased with increasing human population. Additional variation in TR and TB were explained by salinity and inorganic phosphate (**Supplementary Table 4.8.1**) but neither of these additional variables were significant without log human population. In the case of TA, depth and

inorganic nitrate were also significant once the effect of human population was included but again, model improvement was negligible (**Supplementary Table 4.8.2**). These relationships held, albeit were weaker, when those points visually identified as outliers were included. In the case of TA, two locations from the Recherche Archipelago had higher than expected abundances, whilst Ashmore Reef and the Rowley Shoals had much lower abundances than expected. In the case of biomass, the two Argo-Rowley Terrace surveys had lower than expected biomass as did the first Ningaloo 2016 survey.

We analysed the taxonomic assemblage in terms of both abundance (**Fig. 4.4**) and biomass by taxa (**Fig. 4.5**). In both cases, we saw very strong separation of assemblages by region. In the case of abundance, the most important separators of taxonomic assemblage were sea surface temperature and chlorophyll. No seabed morphometric characteristics was important. Distance to the nearest port, an indicator of anthropogenic impact, was important in both cases. The taxa that distinguished the regions were generally large, high trophic level predators, with the south coast characterised by the copper shark *Carcharhinus brachyurus* and the southern blue fin tuna *Thunnus maccoyii*, whilst the north was characterised by billfishes (Istiophoridae) and the tiger shark. In contrast, the central coast was dominated by forage species such as scads *Decapterus* sp. and trevallies *Caranx* sp., as well as very small fishes such as the driftfishes (Nomeidae).

Table 4. 2 Regression equations for log taxonomic richness (LTR), log total abundance (LTA) and log total biomass (LTB) indicating that, in each case, the log of population size for the nearest community (LPOP), was the best predictor variable. Regression statistics included number of locations included in the model (n), the coefficient of determination (R^2), model standard error (SEE), p-value, intercept (B_0) and slope (B_1) with their associated standard errors (SE)

DV	n	R^2	SEE	p	B_0 (SE)	B_1 (SE)
LTR	29	0.37	0.094	0.000458	0.672 (0.067)	-0.058 (0.014)
LTA	25	0.43	0.199	0.000397	1.398 (0.145)	-0.128 (0.031)
LTB	26	0.57	0.630	7.48E-06	5.995 (0.489)	-0.590 (0.103)

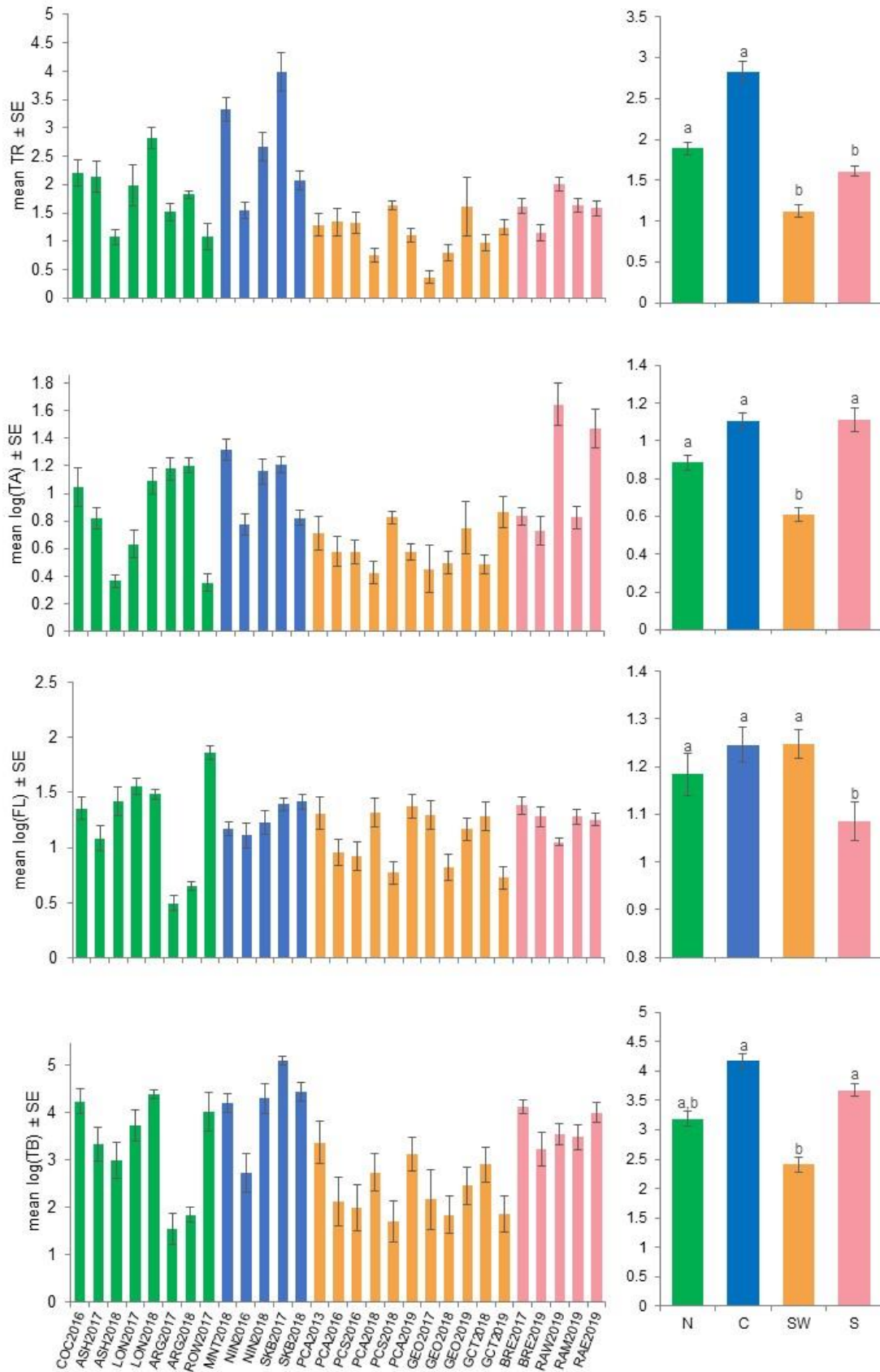


Figure 4. 2 Mean values with standard errors (SE) for taxonomic richness (TR), and logged values of total abundance (TA), fork length (FL) and total biomass (TB) by survey (left) and by region (right).

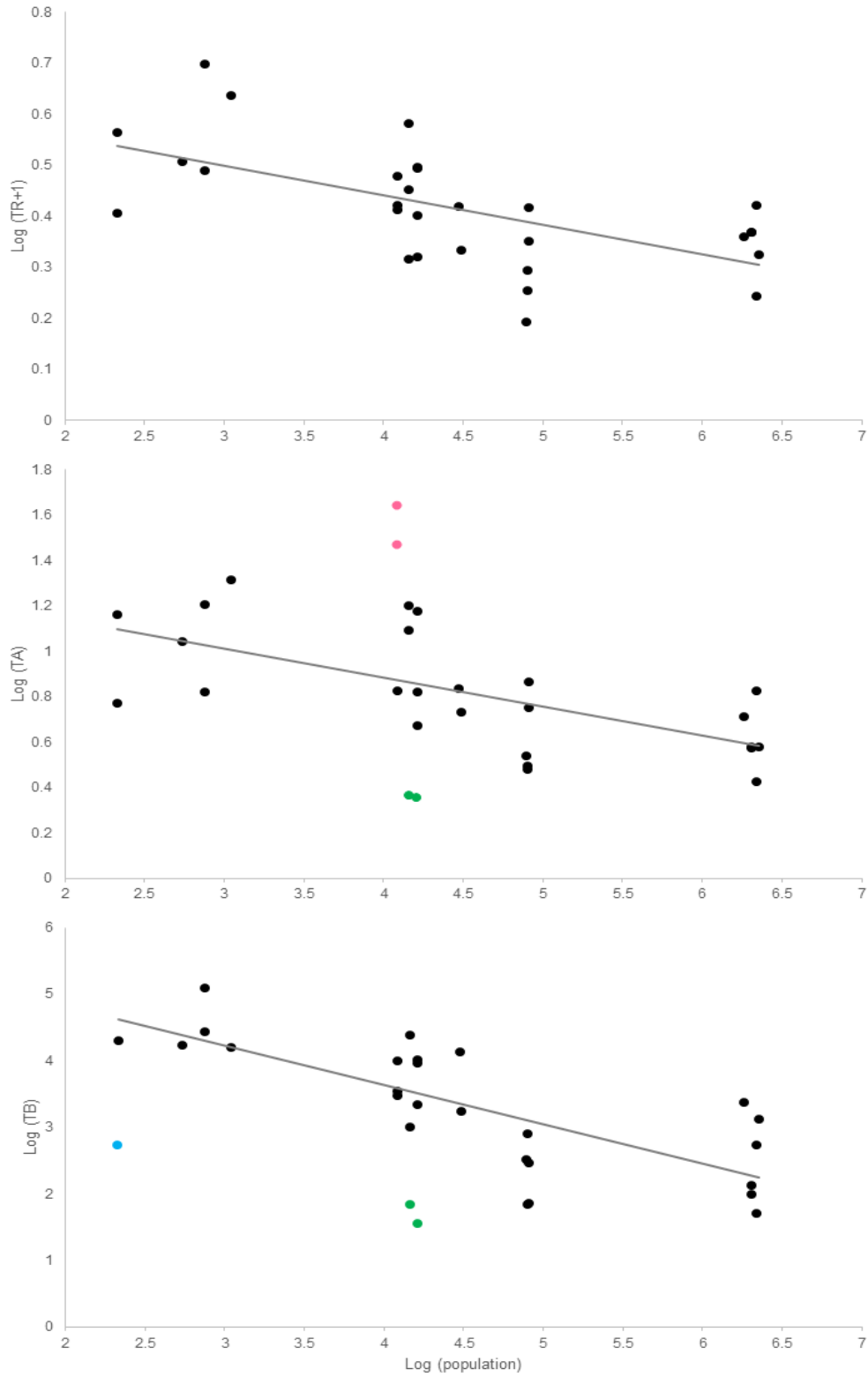


Figure 4. 3 Linear regressions for taxonomic richness (TR), log₁₀ total abundance (TA) and log₁₀ total biomass (TB) as a function of the log₁₀ of size of the nearest human population centre. Outliers are colour coded according to region: North (blue), Centre (green), South (pink).

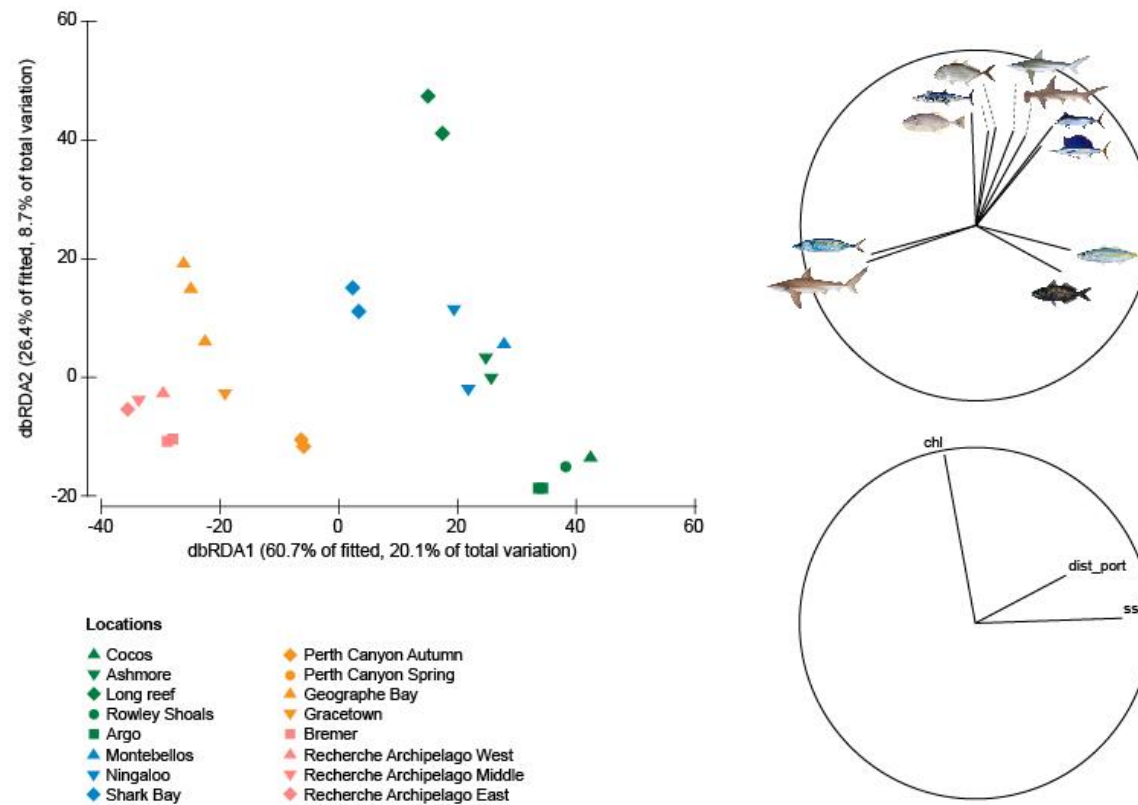


Figure 4. 4 Distance-based redundancy analysis (dbRDA) plot for total abundance (left) and vector overlays with taxon (top right), and environmental variables chl, distance to the nearest port (dist_port), and sea surface temperature (sst) (bottom right). Taxon in clockwise order are as follows: leatherjackets *Monacanthidae* sp., mackerels *Scomber* sp., giant trevally *Caranx ignobilis*, requiem sharks *Carcharhinidae* sp., great hammerheads *Sphyrna mokarran*, billfishes *Istiophoridae* sp., Indo-Pacific sailfish *Makaira mazara*, scads *Decapterus* sp., driftfishes *Nomeidae* sp., bronze whaler *Carcharhinus brachyurus*, blue mackerel *Scomber australisicus* (Illustrations courtesy of anima.net.au).

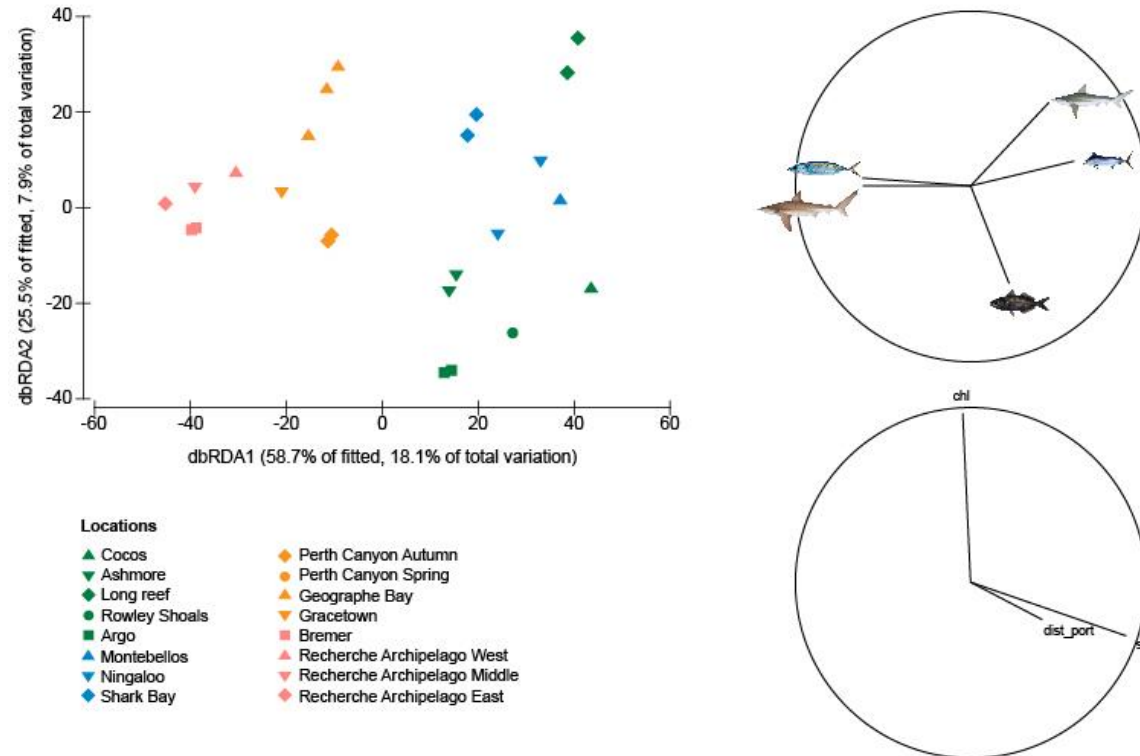


Figure 4. 5 Distance-based redundancy analysis (dbRDA) plot for total biomass (left) and vector overlays with taxon (top right), and environmental variables chloryphyll-a (chl), sea surface temperature (sst), and distance to the nearest port (dist_port) (bottom right). Taxon in clockwise order are as follows: blacktip shark *Carcharhinus limbatus*, billfishes *Istiophoridae* sp., driftfishes *Nomeidae* sp., bronze whaler *Carcharhinus brachyurus*, blue mackerel *Scomber australisicus*. (Illustrations courtesy of anima.net.au).

4.5 DISCUSSION

Our analysis of nearly 23° of latitude and two oceans, the Indian and the Southern, suggests that human activity is the best predictor of taxonomic diversity, abundance and biomass in pelagic wildlife assemblages. Size of human population centres nearest to each survey location predicted these characteristics better than any other variable included in the study; of importance was the lack of a correlation between latitude and attributes of human population, with both large and small populations found in the south of WA. Moreover, contrary to our predictions based on the Latitudinal Diversity Gradient and Bergmann's Rule, there was no difference in species diversity and mean fork length values between the tropical northern and temperate southern locations. These results were consistent with neither Tittensor et al. (2010) nor Worm et al. (2003) where some patterns were found with latitude. This divergence may reflect the different datasets underpinning the respective analyses. In the case of the earlier studies, the analyses were based on predicted distributions, whereas our analysis was based on empirical observations of animal abundance. In this sense, the inconsistency may reflect the absence of these animals from where they should be and provides strength to the argument that human presence is overwhelming any natural underlying patterns in biogeography that may be present. These results are surprising given that Western Australia is the largest state in the country yet has the lowest human population size, approximately 2.6 million people. In theory, therefore, the effect of humans on pelagic wildlife in this state should be relatively low, yet our results suggest the opposite.

Pelagic wildlife assemblages were particularly depauperate in the southwest region of Western Australia, with species richness, total abundance and biomass values here consistently lower than they were in the other studied regions, despite the existence of a diversity of productive habitats that included the pelagic hotspot, the Perth Canyon, the seagrass meadows of Geographe Bay and a west facing shelf break at Gracetown. We note, however, that the southwest region contains the vast majority of the state's population, with the state capital, Perth, and its suburbs, home to over two million people and the towns of Bunbury (31.9K) and Busselton (36.3K) respectively the second and eighth most populous settlements in the state (ABS 2018). A plausible explanation for our finding is that Geographe Bay and Gracetown are subject to relatively high recreational fishing pressure that includes 'weekenders' from Perth (Ryan et al. 2017), in addition to being a major centre of gillnet fishing for sharks (Gaughan & Santoro 2019). Moreover, although the regional population is just over one order of magnitude lower than that of Perth, diversity, abundance and biomass are comparable to those for the Perth Canyon, with a lower biomass than

predicted based on nearest population. This is likely because the effective population is much larger than that used in the model.

While the negative impact of human population density on the health of coastal and/or reef fish assemblages is well established (Stallings 2009, Mora & Sale 2011, Bellwood et al. 2012, Cinner et al. 2013, Juhel et al. 2019), evidence for a human “footprint” among pelagic assemblages is less developed. Both human activity (fishing mortality, sub-lethal disturbance and displacement) and distance to human settlements have been identified as important predictors of both the abundance of pelagic fishes and the location of pelagic “hotspots” (Bouchet et al. 2017, Letessier et al. 2019). The true effects of humans on fish assemblages are, however, likely best understood as a function of multiple aspects of human activity, including population size, remoteness and distance/time to market, a concept referred to as “gravity” by Cinner et al. (2016). Moreover, similar to our study, the aptly named “Breaking Bergmann’s Rule” analysis by Fisher et al. (2010a) concluded that underlying biogeographical patterns in the size of pelagic fishes have been erased in the Northwest Atlantic by the effects of fishing. Indeed, although the “Father of International Law”, Hugo Grotius, once characterized the oceans as an infinite source of fish and encouraged “open access” for all, this is clearly no longer the case (Russ & Zeller 2003).

Environmental variables also explained some of the variation in pelagic fish assemblage traits in the univariate analyses, however they were only significant after the negative effect of the size of the nearest human population centre was included. We found a negative relationship between inorganic nutrient concentrations and fish abundance and biomass, which is to be expected as the high phytoplankton concentrations are needed to support large fish populations (Selvin Pitchaikani & Lipton 2012, Bai et al. 2015). Contrary to our expectations, temperature, oxygen, productivity, depth and rugosity (Cheung et al. 2013, Bouchet et al. 2017) were not important factors in our study. In contrast to the univariate analyses, the multivariate analyses showed stronger associations with environmental variables, with sea surface temperature and chlorophyll-a as the most important determinants of fish abundance and biomass, in addition to distance to nearest port. Large-scale biogeographical patterns were largely consistent with the LDG and Bergmann’s rule, however we note that distance to port was still an important variable, with apex predators such as the copper shark *Carcharhinus brachyurus* and billfishes (Istiophoridae) more common at locations distant from humans, regardless of their affinity for temperate or tropical environments, consistent with the results of Bouchet et al. (2017).

Whilst we have observed a strong relationship between attributes of pelagic fish assemblages and the human footprint, this study is, nonetheless, correlative. Indeed, it is

possible that pelagic fish assemblages do not adhere to the LDG and Bergmann's Rule for reasons that are not anthropogenic, with some studies suggesting that there are exceptions to these biogeographic rules (Meiri et al. 2004, Adams & Church 2008). We note, however, that there is good evidence for Bergmann's Rule in fishes, including mesopelagic species (Edeline et al. 2013, Fernández-Torres et al. 2018, Saunders & Tarling 2018). Moreover, lack of biogeographic trends in size and diversity have been attributed to insufficient sampling, both in terms of number of samples and extent of a species' range, while species that have relatively restricted ranges are also less likely to display size clines (Palmer 1999, Meiri et al. 2004). However, we note that our analysis included relatively large sample sizes and that many of the species are highly mobile, with large ranges, thus these are unlikely explanations for our results. Lastly, it has been shown that adherence to ecogeographic rules is sometimes intra- rather than inter-specific (Meiri et al. 2004), thus it is also possible that if our analysis was conducted by species, we would find large-scale trends in size that are not otherwise apparent.

According to the most recent "State of the Fisheries" report by the Department of Primary Industries and Regional Development, fisheries management in WA currently ranks as "outstanding," with "97% of our fish stocks assessed as not being at risk or vulnerable through exploitation" (Gaughan & Santoro 2019). Moreover, the report categorises the threats posed by climate change, bycatch and depredation to large, pelagic species as "low risk." In contrast, our findings suggest that offshore fish assemblages in WA may, in fact, be vulnerable to anthropogenic impacts and suggest an urgent need for improved ocean management via large marine protected areas, consistent with a number of other studies (Edgar et al. 2014, Roberts et al. 2017, O'Leary & Roberts 2018). This is of particular concern given the Australian government's decision in 2018 to substantially cut the area of its Marine Parks protected by "no take" zoning and increase partial protection, triggering claims that Australia had gone from "leader" to "laggard" in terms of ocean conservation (Readfearn 2018). Partially protected marine parks are not only more expensive to administer than no-take parks but also less effective at delivering conservation outcomes (Ban et al. 2011, Sciberras et al. 2015, Giakoumi et al. 2017, Turnbull et al 2018). Moreover, some Australian marine parks have been declared "residual", i.e. purposefully located in areas that are undesirable to extractive businesses (Devillers et al. 2015). Our results, in conjunction with growing evidence that pelagic fish assemblages demonstrate striking spatial and temporal stability and can therefore benefit from "static" marine parks (Jorgensen et al. 2010, Forrest et al. 2019a, Forrest et al. 2019b), suggest that a dramatic increase in no-take MPAs may be necessary to at least slow declines in offshore fish assemblages vulnerable to human impacts.

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4.8 SUPPLEMENTARY MATERIALS

Supplementary Table 4.8. 1 Metrics of each survey from the Great West Ozzie Transect ordered from north to south, including location (decimal degrees), start and end dates and the number of longline samples (n).

Survey	Start date	End date	n	LAT*	LONG*	TR* ± SE	TA* ± SE	TB* ± SE (kg)
COC†2016	11/10/2016	21/11/2016	22	-12.12	96.82	2.21 ± 0.230	30.18 ± 9.508	81,000 ± 26,200
ASH2017	14/07/2017	21/07/2017	16	-12.21	123.06	2.14 ± 0.283	7.71 ± 1.908	14,600 ± 5,000
ASH2018	2/10/2018	7/10/2018	24	-12.22	123.05	1.07 ± 0.138	1.61 ± 0.307	39,900 ± 27,000
LON2017	30/06/2017	13/07/2017	16	-13.82	125.74	2.12 ± 0.369	5.69 ± 1.594	26,400 ± 9,800
LON2018	18/09/2018	23/09/2018	24	-13.88	125.74	2.83 ± 0.185	21.88 ± 5.995	43,200 ± 12,700
ARG2017	16/11/2017	18/11/2017	12	-15.15	118.50	1.52 ± 0.149	17.47 ± 3.925	2,200 ± 2,100
ARG2018	4/08/2018	10/08/2018	36	-15.45	118.53	1.83 ± 0.063	20.26 ± 3.225	900 ± 400
ROW2017	19/11/2017	22/11/2017	11	-17.10	119.42	1.09 ± 0.238	1.49 ± 0.369	26,500 ± 4,500
MNT2018	15/08/2018	23/08/2018	40	-20.08	115.36	3.34 ± 0.208	35.15 ± 6.571	71,900 ± 13,500
NIN2016	15/09/2016	22/09/2016	16	-21.84	113.58	1.55 ± 0.143	6.53 ± 1.367	23,000 ± 10,700
NIN2018	24/07/2018	30/07/2018	20	-21.84	113.88	2.67 ± 0.246	24.09 ± 8.517	94,700 ± 19,400
SKB2017	15/09/2017	21/09/2017	15	-26.19	113.05	3.99 ± 0.350	17.05 ± 2.246	161,200 ± 29,200
SKB2018	6/08/2018	11/08/2018	20	-26.17	113.07	2.08 ± 0.164	6.7 ± 0.941	72,500 ± 18,400
PCA2013	30/03/2013	17/05/2013	9	-31.95	115.11	1.29 ± 0.208	6 ± 2.175	25,800 ± 18,100
PCA2016	27/04/2016	4/05/2016	10	-31.96	115.10	1.34 ± 0.244	3.81 ± 0.977	3,600 ± 2,100
PCS2016	25/11/2016	2/12/2016	11	-31.95	115.10	0.75 ± 0.117	3.07 ± 1.191	8,400 ± 2,900
PCA2018	17/04/2018	12/05/2018	20	-31.98	115.12	1.11 ± 0.120	3.44 ± 0.529	14,800 ± 3,800
PCS2018	6/12/2018	14/12/2018	12	-31.99	115.11	1.33 ± 0.187	3.46 ± 0.739	3,700 ± 2,100
PCA2019	9/04/2019	12/04/2019	20	-31.96	115.12	1.63 ± 0.077	6.05 ± 0.740	4,800 ± 3,400
GEO2017	4/02/2017	8/02/2017	10	-33.54	115.21	0.56 ± 0.188	6.12 ± 2.940	18,100 ± 13,700
GEO2018	9/02/2018	13/02/2018	20	-33.45	115.26	0.80 ± 0.140	3.35 ± 0.809	5,100 ± 2,800
GEO2019	7/03/2019	10/03/2019	20	-33.49	115.23	1.61 ± 0.515	40.48 ± 19.705	10,000 ± 3,800
GCT2018	21/05/2018	2/06/2018	20	-34.02	114.78	0.97 ± 0.145	2.87 ± 0.670	11,700 ± 3,800
GCT2019	28/02/2019	6/03/2019	20	-34.03	114.77	1.24 ± 0.134	19.85 ± 12.367	10,200 ± 6,900
BRE2017	27/02/2017	8/03/2017	20	-34.71	119.71	1.62 ± 0.123	7.82 ± 1.847	24,200 ± 4,500
BRE2019	6/03/2019	10/03/2019	20	-34.72	119.73	1.15 ± 0.139	8.36 ± 2.589	23,000 ± 7,200
RAW2019	12/02/2019	17/02/2019	22	-34.29	122.02	2.01 ± 0.119	132.61 ± 35.755	14,500 ± 4,800
RAM2019	4/02/2019	9/02/2019	22	-34.23	123.35	1.64 ± 0.122	10.33 ± 3.984	20,200 ± 10,800
RAE2019	28/01/2019	2/02/2019	22	-33.82	124.28	1.59 ± 0.132	75.45 ± 22.861	32,800 ± 9,800

Survey	POP.*	PORT± SE (km)	CAN ± SE (km)	CANU ± SE (km)	CANH ± SE (km)	SHELF± SE (km)	COAST± SE (km)
COC2016	544	8 ± 0.7	3 ± 0.8	3 ± 0.8	4 ± 0.8	2105 ± 1.6	2117 ± 1.6
ASH2017	16,222	647 ± 2.5	66 ± 2.6	66 ± 2.6	66 ± 2.6	24 ± 2.5	353 ± 2.6
ASH2018	14,445	644 ± 2.1	64 ± 2.1	64 ± 2.1	64 ± 2.1	23 ± 2.5	352 ± 2.4
LON2017	16,222	595 ± 3.9	357 ± 2.7	357 ± 2.7	359 ± 2.7	217 ± 3.2	30 ± 3.0
LON2018	14,445	591 ± 3.5	359 ± 4.2	359 ± 4.2	361 ± 4.2	220 ± 4.5	23 ± 3.3
ARG2017	16,222	507 ± 8.1	29 ± 8.8	33 ± 9.4	38 ± 9.6	278 ± 8.1	458 ± 8.5
ARG2018	14,445	484 ± 7.1	24 ± 4.4	28 ± 4.8	32 ± 4.9	257 ± 6.4	439 ± 6.5
ROW2017	16,222	311 ± 4.5	125 ± 2.8	125 ± 2.8	126 ± 2.8	82 ± 2.3	287 ± 4.4
MNT2018	1,104	148 ± 3.9	45 ± 3.9	45 ± 3.9	46 ± 3.8	25 ± 2.8	122 ± 3.8
NIN2016	214	159 ± 5.7	8 ± 3.1	9 ± 3.1	18 ± 3	29 ± 5.5	40 ± 5.5
NIN2018	214	128 ± 4.4	13 ± 4.0	13 ± 3.9	15 ± 3.7	6 ± 0.9	17 ± 1.0
SKB2017	754	39 ± 2.2	153 ± 2.6	153 ± 2.6	154 ± 2.6	57 ± 1.7	15 ± 2.5
SKB2018	754	42 ± 2.3	154 ± 1.9	154 ± 1.9	154 ± 1.9	58 ± 1.5	17 ± 3.1
PCA2013	1,830,000	54 ± 1.0	0 ± 0.0	0 ± 0.0	8 ± 0.9	6 ± 0.8	57 ± 1.0
PCA2016	2,040,000	56 ± 1.1	0 ± 0.0	0 ± 0.0	10 ± 1.1	7 ± 0.9	59 ± 1.1
PCS2016	2,192,000	54 ± 1.5	1 ± 0.3	1 ± 0.3	9 ± 1.2	6 ± 1.2	56 ± 1.5
PCA2018	2,280,000	55 ± 1.4	0 ± 0.2	0 ± 0.2	10 ± 1.3	6 ± 1	58 ± 1.4
PCS2018	2,040,000	56 ± 1.1	0 ± 0.0	0 ± 0.0	10 ± 1.1	7 ± 0.9	59 ± 1.1
PCA2019	2,200,000	55 ± 1.7	0 ± 0.2	0 ± 0.2	10 ± 1.7	9 ± 1.4	58 ± 1.7
GEO2017	78,000	48 ± 2.1	113 ± 2.0	113 ± 2.0	113 ± 2	64 ± 2.2	9 ± 1.8
GEO2018	79,800	40 ± 2.4	111 ± 2.3	111 ± 2.3	111 ± 2.3	67 ± 2.5	15 ± 1.2
GEO2019	81,389	44 ± 1.4	112 ± 2.2	112 ± 2.2	112 ± 2.2	65 ± 2.1	13 ± 1.3
GCT2018	79,800	113 ± 1.5	104 ± 1.2	104 ± 1.2	104 ± 1.2	25 ± 1.5	19 ± 1.4
GCT2019	81,389	113 ± 1.7	106 ± 1.1	106 ± 1.1	106 ± 1.1	26 ± 1.6	19 ± 1.5
BRE2017	29,890	165 ± 4.5	1 ± 0.2	1 ± 0.2	13 ± 1.9	2 ± 0.4	35 ± 1.3
BRE2019	30,479	168 ± 4.2	0 ± 0.2	0 ± 0.2	15 ± 2	4 ± 0.7	39 ± 1.4
RAW2019	12,145	50 ± 5.6	35 ± 5.8	35 ± 5.8	42 ± 5.2	35 ± 5.3	33 ± 6.0
RAM2019	12,145	139 ± 3.7	49 ± 2.4	49 ± 2.4	51 ± 2.6	42 ± 2.1	28 ± 2.5
RAE2019	12,145	220 ± 1.5	57 ± 1.7	57 ± 1.7	57 ± 1.7	48 ± 1.6	40 ± 1.2

Survey	depth \pm SE (m)	slope \pm SE (m)	relief \pm SE (m)	SA \pm SE (m ²)
COC2016	1110 \pm 122	16.33 \pm 1.221	221.43 \pm 16.935	82000 \pm 700
ASH2017	210 \pm 16	1.97 \pm 0.334	27.53 \pm 4.674	77000 \pm 0
ASH2018	230 \pm 17	1.51 \pm 0.207	21.38 \pm 2.966	77000 \pm 0
LON2017	40 \pm 1	0.15 \pm 0.029	2.24 \pm 0.419	76000 \pm 0
LON2018	50 \pm 1	0.17 \pm 0.027	2.41 \pm 0.413	76000 \pm 0
ARG2017	5200 \pm 161	3.58 \pm 1.321	48.27 \pm 17.543	77000 \pm 300
ARG2018	4680 \pm 195	3.64 \pm 0.751	48.79 \pm 9.817	77000 \pm 200
ROW2017	430 \pm 3	0.26 \pm 0.025	3.41 \pm 0.345	76000 \pm 0
MNT2018	230 \pm 46	0.94 \pm 0.194	12.98 \pm 2.641	77000 \pm 0
NIN2016	1120 \pm 149	3.31 \pm 0.801	41.68 \pm 10.419	77000 \pm 200
NIN2018	400 \pm 51	2.60 \pm 0.243	32.20 \pm 2.997	77000 \pm 0
SKB2017	100 \pm 4	0.22 \pm 0.057	2.74 \pm 0.731	76000 \pm 0
SKB2018	90 \pm 4	0.22 \pm 0.047	2.62 \pm 0.557	76000 \pm 0
PCA2013	1000 \pm 101	8.49 \pm 0.748	112.77 \pm 9.828	78000 \pm 200
PCA2016	950 \pm 90	10.32 \pm 1.174	137.13 \pm 15.271	79000 \pm 400
PCS2016	700 \pm 89	6.97 \pm 1.013	90.37 \pm 13.023	78000 \pm 300
PCA2018	780 \pm 89	7.63 \pm 0.785	101.04 \pm 10.697	78000 \pm 300
PCS2018	910 \pm 96	7.92 \pm 0.897	104.73 \pm 11.562	78000 \pm 300
PCA2019	850 \pm 105	8.33 \pm 1.271	108.48 \pm 16.995	78000 \pm 600
GEO2017	30 \pm 2	0.12 \pm 0.021	1.60 \pm 0.235	76000 \pm 0
GEO2018	30 \pm 2	0.08 \pm 0.009	1.18 \pm 0.126	76000 \pm 0
GEO2019	30 \pm 2	0.10 \pm 0.010	1.27 \pm 0.117	76000 \pm 0
GCT2018	80 \pm 8	0.36 \pm 0.072	4.00 \pm 0.812	76000 \pm 0
GCT2019	80 \pm 9	0.31 \pm 0.074	3.30 \pm 0.758	76000 \pm 0
BRE2017	580 \pm 114	8.61 \pm 1.134	114.90 \pm 16.187	79000 \pm 600
BRE2019	970 \pm 122	10.91 \pm 0.937	149.51 \pm 15.131	80000 \pm 1000
RAW2019	330 \pm 109	3.89 \pm 1.249	48.43 \pm 15.646	77000 \pm 400
RAM2019	60 \pm 4	0.19 \pm 0.026	2.36 \pm 0.340	76000 \pm 0
RAE2019	60 \pm 2	0.08 \pm 0.014	1.00 \pm 0.186	76000 \pm 0

Survey	lmi* ± SE (m)	tpi* ± SE (m)	Northness ± SE	Eastness ± SE
COC2016	18.20 ± 1.549	-3.17 ± 1.150	-0.053 ± 0.103	-0.332 ± 0.144
ASH2017	31.88 ± 0.293	-0.30 ± 0.312	0.233 ± 0.193	-0.153 ± 0.099
ASH2018	31.45 ± 0.309	-0.13 ± 0.182	0.197 ± 0.145	-0.185 ± 0.077
LON2017	35.12 ± 0.024	-0.05 ± 0.017	0.075 ± 0.063	0.114 ± 0.118
LON2018	35.01 ± 0.029	0.02 ± 0.029	0.146 ± 0.096	0.025 ± 0.088
ARG2017	26.51 ± 1.28	-0.03 ± 0.03	0.054 ± 0.197	-0.243 ± 0.187
ARG2018	6.95 ± 0.69	0.39 ± 0.46	0.132 ± 0.104	-0.326 ± 0.100
ROW2017	7.47 ± 1.22	1.97 ± 1.25	0.732 ± 0.068	-0.185 ± 0.105
MNT2018	31.69 ± 0.794	0.07 ± 0.076	0.424 ± 0.059	-0.338 ± 0.069
NIN2016	18.00 ± 2.074	-0.25 ± 0.238	0.114 ± 0.121	-0.675 ± 0.077
NIN2018	28.55 ± 0.912	-0.13 ± 0.161	0.420 ± 0.062	-0.685 ± 0.075
SKB2017	33.95 ± 0.08	-0.07 ± 0.06	-0.222 ± 0.083	-0.678 ± 0.098
SKB2018	34.09 ± 0.8	-0.02 ± 0.02	-0.172 ± 0.098	-0.663 ± 0.073
PCA2013	18.99 ± 1.463	-2.16 ± 0.972	0.166 ± 0.181	-0.215 ± 0.095
PCA2016	19.77 ± 1.341	-1.78 ± 0.719	-0.113 ± 0.224	-0.067 ± 0.158
PCS2016	23.80 ± 1.395	-0.74 ± 0.492	0.121 ± 0.143	-0.421 ± 0.089
PCA2018	22.52 ± 1.373	0.19 ± 0.790	-0.022 ± 0.145	-0.377 ± 0.092
PCS2018	20.31 ± 1.376	-1.12 ± 0.839	0.078 ± 0.203	-0.105 ± 0.130
PCA2019	21.35 ± 1.535	-0.73 ± 0.803	0.301 ± 0.193	-0.212 ± 0.100
GEO2017	35.44 ± 0.043	-0.01 ± 0.025	0.670 ± 0.067	-0.067 ± 0.138
GEO2018	35.31 ± 0.032	-0.02 ± 0.013	0.484 ± 0.070	-0.264 ± 0.067
GEO2019	35.37 ± 0.047	-0.01 ± 0.017	0.615 ± 0.056	-0.185 ± 0.064
GCT2018	34.35 ± 0.163	0.03 ± 0.021	0.063 ± 0.040	-0.678 ± 0.111
GCT2019	34.29 ± 0.178	-0.02 ± 0.019	0.108 ± 0.051	-0.700 ± 0.096
BRE2017	26.01 ± 1.764	0.70 ± 0.424	-0.844 ± 0.040	0.066 ± 0.043
BRE2019	19.96 ± 1.730	0.67 ± 1.169	-0.632 ± 0.094	0.068 ± 0.076
RAW2019	28.89 ± 1.85	0.3 ± 0.3	-0.008 ± 0.175	-0.162 ± 0.090
RAM2019	34.66 ± 0.070	-0.01 ± 0.012	-0.190 ± 0.125	-0.106 ± 0.117
RAE2019	34.66 ± 0.030	0.00 ± 0.011	-0.125 ± 0.096	0.136 ± 0.080

Survey	O ₂ ± SE (mmol/L)	PO ₄ ± SE (mmol/L)	NO ₃ ± SE (mmol/L)	S ± SE (PSU)	Chl-a* ± SE (mg/m ³)	SST* ± SE (°C)
COC2016	3.484 ± 0.045	2.452 ± 0.009	34.94 ± 0.076	34.740 ± 0.001	0.250 ± 0.008	27.49 ± 0.010
ASH2017	2.605 ± 0.112	1.499 ± 0.101	21.57 ± 1.676	34.533 ± 0.004	0.358 ± 0.015	28.96 ± 0.012
ASH2018	2.554 ± 0.107	1.595 ± 0.097	23.05 ± 1.587	34.536 ± 0.004	0.335 ± 0.004	28.96 ± 0.007
LON2017	4.273 ± 0.010	0.315 ± 0.003	0.80 ± 0.042	34.798 ± 0.010	0.671 ± 0.033	28.94 ± 0.044
LON2018	4.217 ± 0.009	0.341 ± 0.006	1.10 ± 0.078	34.780 ± 0.012	0.624 ± 0.034	29.00 ± 0.039
ARG2017	4.390 ± 0.017	2.331 ± 0.005	34.48 ± 0.160	34.713 ± 0.000	0.206 ± 0.002	28.67 ± 0.025
ARG2018	4.248 ± 0.056	2.369 ± 0.014	34.60 ± 0.166	34.715 ± 0.001	0.204 ± 0.001	28.63 ± 0.015
ROW2017	2.667 ± 0.022	2.076 ± 0.019	31.35 ± 0.304	34.651 ± 0.004	0.231 ± 0.002	28.39 ± 0.016
MNT2018	3.997 ± 0.119	0.811 ± 0.127	9.71 ± 1.999	34.989 ± 0.029	0.395 ± 0.011	26.90 ± 0.032
NIN2016	3.139 ± 0.217	2.132 ± 0.227	30.63 ± 3.360	34.828 ± 0.090	0.357 ± 0.020	25.90 ± 0.031
NIN2018	4.179 ± 0.126	0.993 ± 0.164	14.24 ± 2.683	35.035 ± 0.091	0.452 ± 0.011	25.89 ± 0.031
SKB2017	4.667 ± 0.008	0.200 ± 0.005	1.21 ± 0.071	35.431 ± 0.009	0.486 ± 0.017	23.30 ± 0.023
SKB2018	4.671 ± 0.007	0.196 ± 0.005	1.16 ± 0.066	35.426 ± 0.011	0.515 ± 0.018	23.30 ± 0.025
PCA2013	4.073 ± 0.212	1.981 ± 0.109	31.15 ± 1.900	34.526 ± 0.021	0.356 ± 0.003	21.33 ± 0.003
PCA2016	4.106 ± 0.215	1.981 ± 0.132	30.60 ± 2.211	34.531 ± 0.038	0.354 ± 0.004	21.33 ± 0.003
PCS2016	4.653 ± 0.168	1.406 ± 0.165	21.07 ± 2.868	34.825 ± 0.105	0.363 ± 0.005	21.32 ± 0.002
PCA2018	4.501 ± 0.170	1.599 ± 0.150	24.26 ± 2.645	34.723 ± 0.090	0.362 ± 0.005	21.33 ± 0.003
PCS2018	4.285 ± 0.192	1.876 ± 0.097	29.52 ± 1.714	34.528 ± 0.022	0.354 ± 0.003	21.33 ± 0.003
PCA2019	4.475 ± 0.222	1.713 ± 0.130	26.78 ± 2.431	34.592 ± 0.048	0.360 ± 0.005	21.32 ± 0.004
GEO2017	5.232 ± 0.010	0.164 ± 0.001	0.30 ± 0.003	35.749 ± 0.002	0.568 ± 0.041	19.66 ± 0.036
GEO2018	5.228 ± 0.008	0.167 ± 0.001	0.31 ± 0.008	35.742 ± 0.003	0.505 ± 0.016	19.65 ± 0.031
GEO2019	5.241 ± 0.013	0.167 ± 0.001	0.32 ± 0.008	35.744 ± 0.003	0.599 ± 0.051	19.64 ± 0.035
GCT2018	5.175 ± 0.015	0.180 ± 0.007	0.50 ± 0.036	35.732 ± 0.003	0.437 ± 0.017	20.18 ± 0.035
GCT2019	5.184 ± 0.016	0.184 ± 0.007	0.51 ± 0.038	35.732 ± 0.003	0.435 ± 0.018	20.17 ± 0.038
BRE2017	4.896 ± 0.130	1.247 ± 0.194	17.45 ± 3.134	34.940 ± 0.104	0.400 ± 0.002	18.97 ± 0.013
BRE2019	4.517 ± 0.139	1.867 ± 0.150	27.30 ± 2.357	34.644 ± 0.058	0.400 ± 0.004	18.88 ± 0.026
RAW2019	5.130 ± 0.096	0.722 ± 0.174	9.07 ± 2.716	35.380 ± 0.132	0.461 ± 0.011	18.45 ± 0.017
RAM2019	5.417 ± 0.009	0.203 ± 0.003	0.54 ± 0.020	35.787 ± 0.011	0.456 ± 0.009	18.32 ± 0.015
RAE2019	5.339 ± 0.004	0.190 ± 0.004	0.51 ± 0.012	35.813 ± 0.011	0.445 ± 0.004	18.41 ± 0.004

*Latitude (LAT) and longitude (LONG) described in decimal degrees, TR = taxonomic richness, TA = total abundance, TB = total biomass, Pop.= population of the nearest port at closest census date before the survey, source: population.net.au. lmi = the local Moran's I value, tpi= the mean topographic position index value, Chl-a = chlorophyll-a, SST = sea surface temperature (for all variable descriptions see **Supplementary Table 4.8.3**). Make sure all variables included.

†COC = Cocos (Keeling) Islands, ASH = Ashmore Reef, LON = Long Reef, ARG = Argo-Rowley Terrace, ROW = The Rowley Shoals, MNT = Montebello Islands, NIN = Ningaloo Reef, SKB = Shark Bay, PCA = Perth Canyon Autumn, PCS = Perth Canyon Spring, GEO = Geographe Bay, GCT = Gracetown, BRE = Bremer Canyon, RAM = Recherche Archipelago Middle, RAW = Recherche Archipelago West, RAE = Recherche Archipelago East.

Supplementary Table 4.8. 2 Statistics associated with 3, 2 and 1 variable(s) models for taxonomic richness, total abundance and total biomass, including the coefficient of determination (R^2), adjusted R^2 , and p values associated with each of the variables including \log_{10} population (Lpop), salinity, inorganic phosphate (PO_4), depth and inorganic nitrate (NO_3)

Taxonomic richness	R^2	adj R^2	p	Variable	p	Variable	p	Variable	p
3 variables	0.53	0.4783	0.0002	Lpop	7.51E-05	salinity	0.02017	PO_4	0.004831
2 variables	0.42	0.3748	0.0009	Lpop	0.000243	salinity	0.1	-	-
2 variables	0.36	0.3065	0.0033	Lpop	0.000859	PO_4	0.9	-	-
2 variables	0.1174	0.0495	0.1900	salinity	-	PO_4	-	-	-
1 variable	0.3556	0.3317	0.0006	Lpop	0.00064	-	-	-	-
Total abundance									
3 variables	0.3607	0.2840	0.0098	Lpop	0.013577	depth	0.041349	NO_3	0.129571
2 variables	0.2978	0.2438	0.0101	Lpop	0.005646	depth	0.16	-	-
2 variables	0.2424	0.1841	0.3000	Lpop	-	NO_3	-	-	-
2 variables	0.1803	0.1173	0.0754	depth	-	NO_3	-	-	-
1 variable	0.2404	0.2123	0.0069	Lpop	0.006929	-	-	-	-
Total biomass									
3 variables	0.5207	0.4632	0.0003	Lpop	0.000271	salinity	0.00774	PO_4	0.004156
2 variables	0.3300	0.2785	0.0055	Lpop	0.001432	salinity	0.67	-	-
2 variables	0.3599	0.3107	0.0030	Lpop	0.002106	PO_4	0.25	-	-
2 variables	0.1770	0.1136	0.0800	salinity	-	PO_4	-	-	-
1 variable	0.3252	0.3002	0.0012	Lpop	0.001238	-	-	-	-

Supplementary Table 4.8. 3 Environmental variables included in the analysis, in addition to data source, description of the variable, the type of variable and extra information.

Variable Name	Data Source	Description	Type	Additional description
dist_canyon	GA Australian Submarine Canyons	distance to the nearest submarine canyon	numeric	
shelf_incising_canyon	GA Australian Submarine Canyons	1: the nearest canyon is a shelf-incising canyon; 0: the nearest canyon is a slope-confined canyon	categorical	Values were assigned only for the sample units with dist_canyon < 10 km.
dist_canyon_upper	GA Australian Submarine Canyons	distance to the upper reach of the nearest canyon	numeric	
dist_canyon_head	GA Australian Submarine Canyons	distance to the nearest canyon head	numeric	
O ₂	CSIRO CARS	The mean bottom water o ₂ (oxygen) value within the sample unit	numeric	The o ₂ values were derived from the CARS 2006 dataset. The values for the sample units at Cocos Islands were from CARS 2009. The metadata is available in Huang et al. (2010).
PO ₄	CSIRO CARS	The mean bottom water po ₄ (phosphate) value within the sample unit	numeric	The po ₄ values were derived from the CARS 2006 dataset. The values for the sample units at Cocos Islands were from CARS 2009. The metadata is available in Huang et al. (2010).
NO ₃	CSIRO CARS	The mean bottom water no ₃ (nitrate) value within the sample unit	numeric	The no ₃ values were derived from the CARS 2006 dataset. The values for the sample units at Cocos Islands were from CARS 2009. The metadata is available in Huang et al. (2010).
S	CSIRO CARS	The mean bottom water s (salinity) value within the sample unit	numeric	The s values were derived from the CARS 2006 dataset. The values for the sample units at Cocos Islands were from CARS 2009. The metadata is available in Huang et al. (2010).
bathy	GA 250 m bathymetry	the mean bathymetry value within the sample unit	numeric	
bathy_std	GA 250 m bathymetry	the standard deviation of the bathymetry values within the sample unit	numeric	
slope	GA 250 m bathymetry	the mean slope (gradient) value within the sample unit	numeric	the slope was generated from GA 250 m bathymetry grid with a window size of 3 by 3
slope_std	GA 250 m bathymetry	the standard deviation of the slope values within the sample unit	numeric	
relief	GA 250 m bathymetry	the mean topographic relief value within the sample unit	numeric	the topographic relief was generated from GA 250 m bathymetry grid with a window size of 3 by 3

Variable Name	Data Source	Description	Type	Additional description
relief_std	GA 250 m bathymetry	the standard deviation of the topographic relief values within the sample unit	numeric	
SurArea	GA 250 m bathymetry	the mean surface area value within the sample unit	numeric	The surface area indicates the seabed rugosity (Jenness, 2004). The surface area was generated from GA 250 m bathymetry grid with a window size of 3 by 3
SurArea_std	GA 250 m bathymetry	the standard deviation of the surface area values within the sample unit	numeric	
Imi	GA 250 m bathymetry	the mean local moran's I value within the sample unit	numeric	The local moran's I (Imi) indicates the local spatial-autocorrelation (Moran, 1950). The Imi was generated from GA 250 m bathymetry grid with a window size of 3 by 3.
tpi	GA 250 m bathymetry	the mean topographic position index value within the sample unit	numeric	The topographic position index (tpi) indicates slope position (Weiss, 2011). The tpi was generated from GA 250 m bathymetry grid with a window size of 3 by 3.
northness	GA 250 m bathymetry	the mean northness value within the sample unit	numeric	The northness indicates the northness aspect of the location; A value of 1 indicates that the location faces north; a value of -1 indicates that the location faces south. The northness was converted from the aspect data which is derived from the GA 250 m bathymetry grid with a window size of 3 by 3.
eastness	GA 250 m bathymetry	the mean eastness value within the sample unit	numeric	The eastness indicates the eastness aspect of the location; A value of 1 indicates that the location faces east a value of -1 indicates that the location faces west The eastness was converted from the aspect data which was derived from the GA 250 m bathymetry grid with a window size of 3 by 3.
aspect_majority	GA 250 m bathymetry	The dominant aspect direction of the sample unit. 0: flat, 1: north, 2: northeast, 3: east, 4: southeast, 5: south, 6, southwest, 7: west, 8: northwest.	categorical	The aspect data was derived from the GA 250 m bathymetry grid with a window size of 3 by 3.
aspect_variety	GA 250 m bathymetry	The number of different aspect values within the sample unit	numeric	

Variable Name	Data Source	Description	Type	Additional description
geofeature_majority	GA 250 m bathymetry	The dominant geofeature type within the sample unit	categorical	The geofeature data was from Heap and Harris (2008).
geofeature_variety	GA 250 m bathymetry	The number of different geofeature types within the sample unit	numeric	The geofeature data was from Heap and Harris (2008).
gravel	GA Marine Sediment Prediction	The mean seabed gravel content within the sample unit	numeric	The metadata is available in Huang et al. (2010).
mud	GA Marine Sediment Prediction	The mean mud gravel content within the sample unit	numeric	The metadata is available in Huang et al. (2010).
sand	GA Marine Sediment Prediction	The mean seabed sand content within the sample unit	numeric	The metadata is available in Huang et al. (2010).
ew_velocity	HYCOM	The mean East-west current velocity within the sample unit	numeric	The metadata is available in Huang et al. (2013).
ns_velocity	HYCOM	The mean north-south current velocity within the sample unit	numeric	The metadata is available in Huang et al. (2013).
v_velocity	HYCOM	The mean vertical current velocity within the sample unit	numeric	The metadata is available in Huang et al. (2013).
mld1	HYCOM	The mean mixed layer depth (definition 1) within the sample unit	numeric	The metadata is available in Huang et al. (2013).
mld2	HYCOM	The mean mixed layer depth (definition 2) within the sample unit	numeric	The metadata is available in Huang et al. (2013).
chl	IMOS MODIS	The mean Chlorophyll-a value within the sample unit	numeric	The chl data was derived from long-term mean MODIS chlorophyll-a data (2002-2017).
sst	IMOS MODIS	The mean SST value within the sample unit	numeric	The sst data was derived from long-term mean MODIS SST data (2002-2017).
monthly_chl	IMOS MODIS	The mean monthly Chlorophyll-a value within the sample unit	numeric	The monthly chl data was extracted from monthly MODIS Chl data, matching the sampling date to the corresponding month.
monthly_sst	IMOS MODIS	The mean monthly SST value within the sample unit	numeric	The monthly sst data was extracted from monthly MODIS SST data, matching the sampling date to the corresponding month.
dist_coast		distance to the mainland coast	numeric	
dis_shelf_break		distance to the shelf break line	numeric	

Variable Name	Data Source	Description	Type	Additional description
shelf_slope		shelf: the sample unit is either completely or partially within the continental shelf; slope: the sample unit is completely within the continental slope	categorical	
dist_port		distance to the nearest port	numeric	
population_port		population size of the port	numeric	using the nearest census data before the survey

CHAPTER 5: THE REVOLUTIONISING OCEAN MEASUREMENT PROJECT DRIVING A RETURN TO FLOURISHING OCEANS

Keywords: BRUVS, eDNA, eCell, genomics, autonomous vehicles, drones, multibeam, research vessels, artificial intelligence, machine learning

5.1 ABSTRACT

In December 2017, UNESCO announced a new initiative: ‘a decade of ocean science,’ which advocates for a renewed focus on ocean research:

“Nearly 3 billion people depend on marine and coastal biodiversity to meet their needs ... [The ocean] absorbs around a third of the CO₂ produced by humans and reduces the impact of climate change. However, science has not yet managed to fully evaluate the cumulative effects of human activities on the ocean, including the impact of pollution, warming and acidification, which threaten this environment, which is vital for our survival”.

One of the greatest threats to the ocean is a global reliance on seafood as a primary source of protein, which has led to ongoing overexploitation of declining fish populations, compounded by the effects of a warming climate and ocean plastification. Despite these growing threats, it remains challenging to efficiently quantify human impacts on ocean habitats and wildlife at large-scales. Clearly, approaches that map and predict how our oceans’ biota is changing are vital to its future management. Here, we propose a technology initiative, the Revolutionising Ocean Measurement Project (ROMP), which seeks to provide a conceptual framework for holistic ecosystem-based monitoring of oceans at the necessary speed and scale.

5.2 INTRODUCTION

“And to the question asked by Ecclesiastes six thousand years ago, ‘That which is far off and exceeding deep, who can find it out?’”

-Jules Verne, 20,000 Leagues Under the Sea

Up to forty percent of the total animal protein consumed by some small-island and/or low-income countries is comprised of seafood (FAO 2014) and the livelihoods of over 200 million people rely on ocean fisheries (Teh and Sumaila 2011). Yet our oceans are experiencing dramatic declines in wildlife populations that will alter many of the ecosystem services that are fundamental to human civilization (Ceballos et al. 2017). Threats to the oceans are well-documented and include overexploitation (Dulvy et al. 2008, Pauly and Zeller 2016a), industrialisation (McCauley et al. 2017), climate change (Cheung et al. 2009) and plastification (Jambeck et al. 2015). Indeed, ocean ecosystems are already displaying irreversible, catastrophic regime shifts (Hare and Mantua 2000, Scheffer and Carpenter 2003) yet a minority of scientists continue to “fiddle while Rome burns,” questioning the scientific consensus that ocean wildlife has undergone an alarming decline in recent decades and, more broadly, generating an exaggerated air of debate that fuels inaction by policy-makers (e.g. Hilborn et al. 2003, Hilborn 2007, Gaughan et al. 2019, Hilborn and Hilborn 2019). While scientific reticence is key to the scientific method, excessive caution prevents the public from understanding the potentially catastrophic implications of human activity (Hansen 2007). “We may rue reticence, if it serves to lock in future disasters,” Hansen (2007) notes. In the case of ocean conservation, the dangers of scientific reticence are further compounded by presentist bias, which can lead to misplaced optimism about global catch data (Zeller and Pauly 2018) and datasets that are often poor and incomplete (Carruthers et al. 2014).

A disturbing example of how poor data, combined with scientific reticence, misinterpretation of fisheries data and mismatches between international, best-practice recommendations and national policy can lead to catastrophic environmental outcomes is provided by the shortfin mako shark *Isurus oxyrinchus*. The shortfin mako shark was listed as “Vulnerable” in 2009 by the International Union for Conservation of Nature (IUCN) (Cailliet et al. 2009), and, after a ten year deterioration in its population status, upgraded to “Endangered” in 2019 (Rigby et al. 2019). Finally, in 2019, trade in the species was restricted by the listing of the shortfin mako on Schedule II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), a move that was described as a “last ditch” attempt to reverse population collapse (Fobar 2019). Within Australia, this was the first protection the shortfin mako received as the nation is a CITES member, despite the government having

judged there to be a threat to the species from bycatch many years prior (Listing Advice 2014). The downwards spiral towards extinction of species like the shortfin mako raises important questions about how a lack of data, together with incoherence between international and national policy, can facilitate inaction, particularly in countries such as Australia where recent reports show declines of up to 92% in other shark species (Roff et al. 2018).

Globally, progress on conservation is either paralysed or occurring too slowly to adequately address the threats that are facing the earth's "life support systems" (Scharlin 1982, Ricketts and Hildebrand 2011). This is partly due to lack of funding: for example, Australia has the world's third largest Exclusive Economic Zone (EEZ) yet invested less than one percent of the estimated value of its "blue economy", A\$47.2 billion, in research in 2015 (National Marine Science Committee 2015). Reflecting this lack of investment, national "State of the Environment" reporting on oceans is, at best, meagre (Mackay and Wienecke 2011). Another roadblock to progress is that the tools available for monitoring ocean health are not fit for purpose: they are relatively slow, expensive, lacking resolution at ecosystem scales (e.g. satellite tagging, acoustic receiver arrays, the use of trawling by fisheries departments to estimate biodiversity) and often lethal. This situation is compounded by the very limited availability of vessels suitable for large-scale ocean ecology research, lengthy gaps between expeditions due to lack of funding, low incentives to innovate and a lack of *in situ* technologies allowing immediate analysis (Hand and German 2018).

Here, we present the Revolutionising Ocean Measurement Project (ROMP), a vision for a next-generation research ship that will integrate and scale existing technologies in addition to adopting ideas from other scientific realms. Rapid technological developments from areas as diverse as renewable power generation, automation, artificial intelligence and machine learning, cloud computing, genomics and image analysis can be harnessed to disrupt conventional approaches to ocean research. As a platform to mobilise more efficient biomonitoring tools that can holistically quantify marine ecosystem health, we propose that ROMP will provide the catalyst for more informed decision-making in marine conservation.

5.3 THE STATUS OF OCEAN WILDLIFE

We suggest that one of the most fundamental challenges to our oceans is the declining status of ocean wildlife. We identify complementary and non-destructive methods to document the status of wildlife across depths as the centrepiece of ROMP: automation of sampling of the atmosphere, sea surface and submarine vessels to cover the entire water column and gaseous exchanges between air and sea, including stereo Baited Remote Underwater Video Systems (stereo-BRUVs), genomics (eDNA and eCell) and sonar.

Stereo-BRUVS are based on the principle of two cameras that share a field of view centred on a bait canister, collecting video imagery of the animals within the field of view to quantify diversity, relative abundance and size (Whitmarsh et al. 2017). The method has been used for over two decades (Harvey and Shortis 1998b) to document the status of demersal species in both shallow (Langlois et al. 2010) and deep (Priede et al. 1994) habitats, from the tropics (Andradi-Brown et al. 2016) to the poles (Devine et al. 2019). More recently, the method has been adapted to pelagic habitats (Heagney et al. 2007) resulting in, for instance, global maps of oceanic hotspots for sharks (Letessier et al. 2019), identification of pelagic shark pupping grounds (Forrest et al. 2019d), in addition to long-term, fine-scale distributions of pelagic assemblages that demonstrate spatial and temporal stability (Bouchet et al. 2015, Forrest et al. 2019a).

Environmental DNA is defined as genetic material that can be recovered from an environmental sample and typically originates from multiple organisms and sources, including eggs, shed skin and biological secretions (Valentini et al. 2016). The role of eDNA in aquatic research is expanding rapidly (Kelly et al. 2017) as it has the potential to provide more accurate assessments of species diversity than traditional surveys (Jerde et al. 2019), potentially estimate biomass (Takahara et al. 2012) and identify cryptic and rare species (Jerde et al. 2011), and was even recently applied to the mystery of the Loch Ness monster (Greshko 2019). Environmental DNA techniques are yet to be widely implemented in routine ocean monitoring, even though the installation of rapid and portable DNA sequencers and cell-sorters on sea-going vessels is already a reality (Truelove et al. 2019; Forrest, unpubl. data). A possible extension of eDNA is the concept of obtaining whole cells from the oceans. The “eCell” approach, as I have termed it, could be adopted from cancer research (Hanash et al. 2002) and would provide information on a wider range of individual attributes than the ‘barcode’ fragments used in eDNA. Specifically, eCell harvests entire cells, and thus an organism’s entire genome. As such, eCell sampling has the potential to provide data on relative abundance, sex, age and other demographic variables such as kinship that, over time, can be used to evaluate the status and trends of ocean wildlife populations.

Multibeam sonar techniques have long been used to locate and generate relatively low-resolution estimates of the abundance of marine animals (Fernandes et al. 2005). However, significant progress has been made in assigning “acoustic signatures” to species that allow species-specific quantification of abundance (Letessier et al. 2016). Multibeam sonar technologies, by acquiring both individual targets and the size of the targets, also allow quantification of biomass on large scales relative to other sampling methods (Boersch-Supan et al. 2017, Proud et al. 2017). Individually, stereo-BRUVS, genomics and sonar have ‘blind spots’, which is why some studies have attempted to combine stereo-BRUVS with either genomics (Stat et al. 2018) or sonar (Letessier et al. 2016). ROMP will combine all three methods, however, with the aim of extending their use to understudied ecosystems ranging from the mesopelagic to the hadopelagic (Friedlander et al. 2019, Giddens et al. 2019)

5.4 THE ROMP – GENERATING OCEAN KNOWLEDGE AT PACE AND SCALE

Fundamental to collecting data about ocean wildlife at an accelerated pace and scale is a need to radically rethink the platforms that we use to conduct marine life surveys. Publicly available information on national research vessels (www.researchvessels.org) suggests that only 21% of coastal countries (32 of 151) have dedicated vessels for ocean research, with these countries accounting for approximately 30% of the world’s EEZs. The philanthropic sector is increasingly providing use of private vessels for ocean research, but the cumulative capacity of this privately-owned and operated fleet is small compared to publicly-funded vessels.

Core to ROMP is the development of a carbon-neutral, circular ecosystem “home ship” that will support multiple autonomous sampling platforms that can cover large swathes of the ocean and sample across a wide range of habitats, environmental parameters and marine life (**Fig 5.1**). We identify five major attributes that are necessary for this “home ship”:

- (1) 24-7 sampling with adequate levels of redundancy;
- (2) Ability to sample over a linear distance of 200 nautical miles (nm) per day, both while transiting and on-station (**Fig 5.2**);
- (3) Autonomous sampling from sky (drones) to seabed;
- (4) High Performance Computing that harnesses on-board data management, cloud computing, machine learning and AI; and
- (5) Telepresence that allows real-time engagement with the broader community.

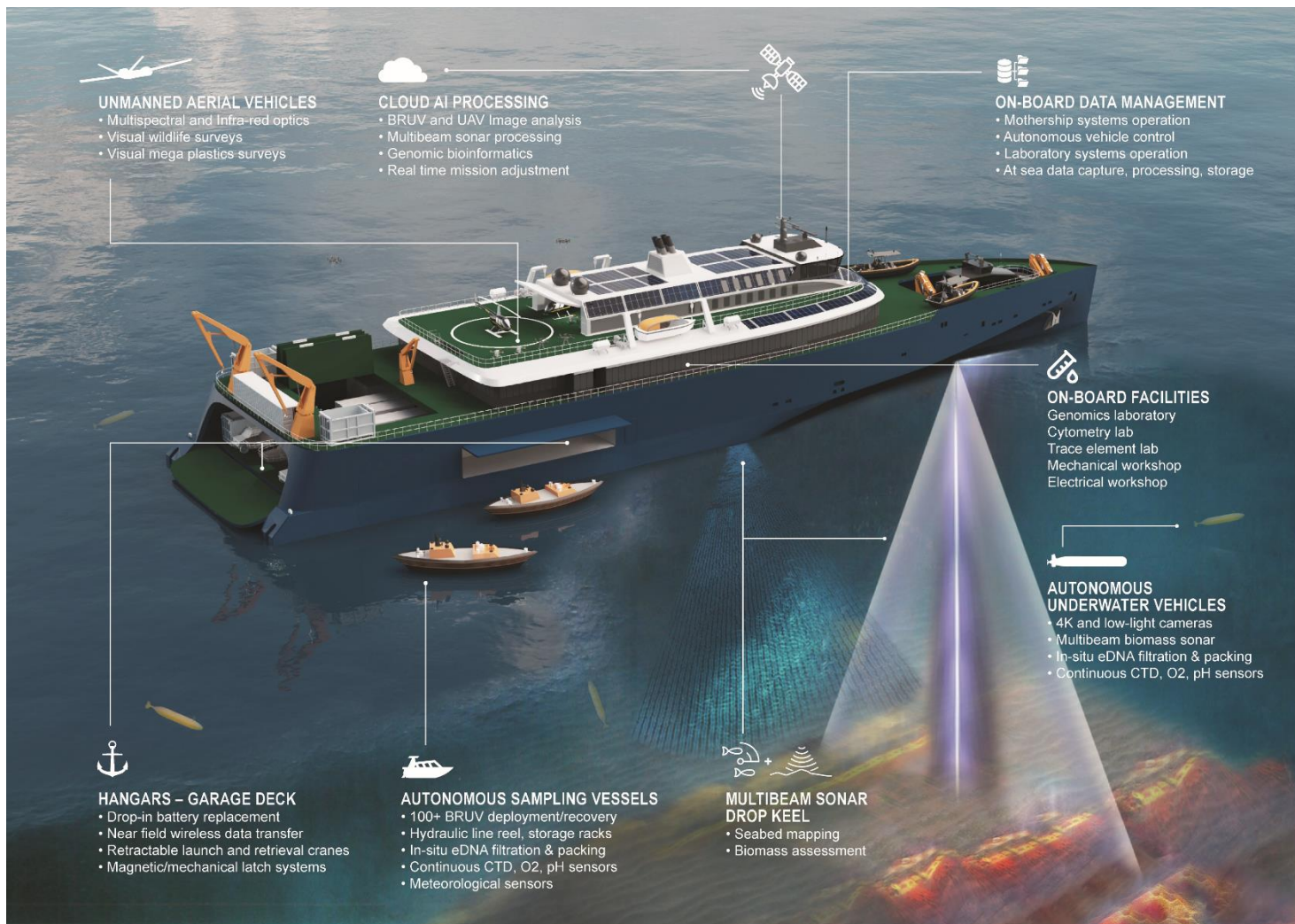


Figure 5. 1 Schematic of home ship indicative of scale and on-board capacity.

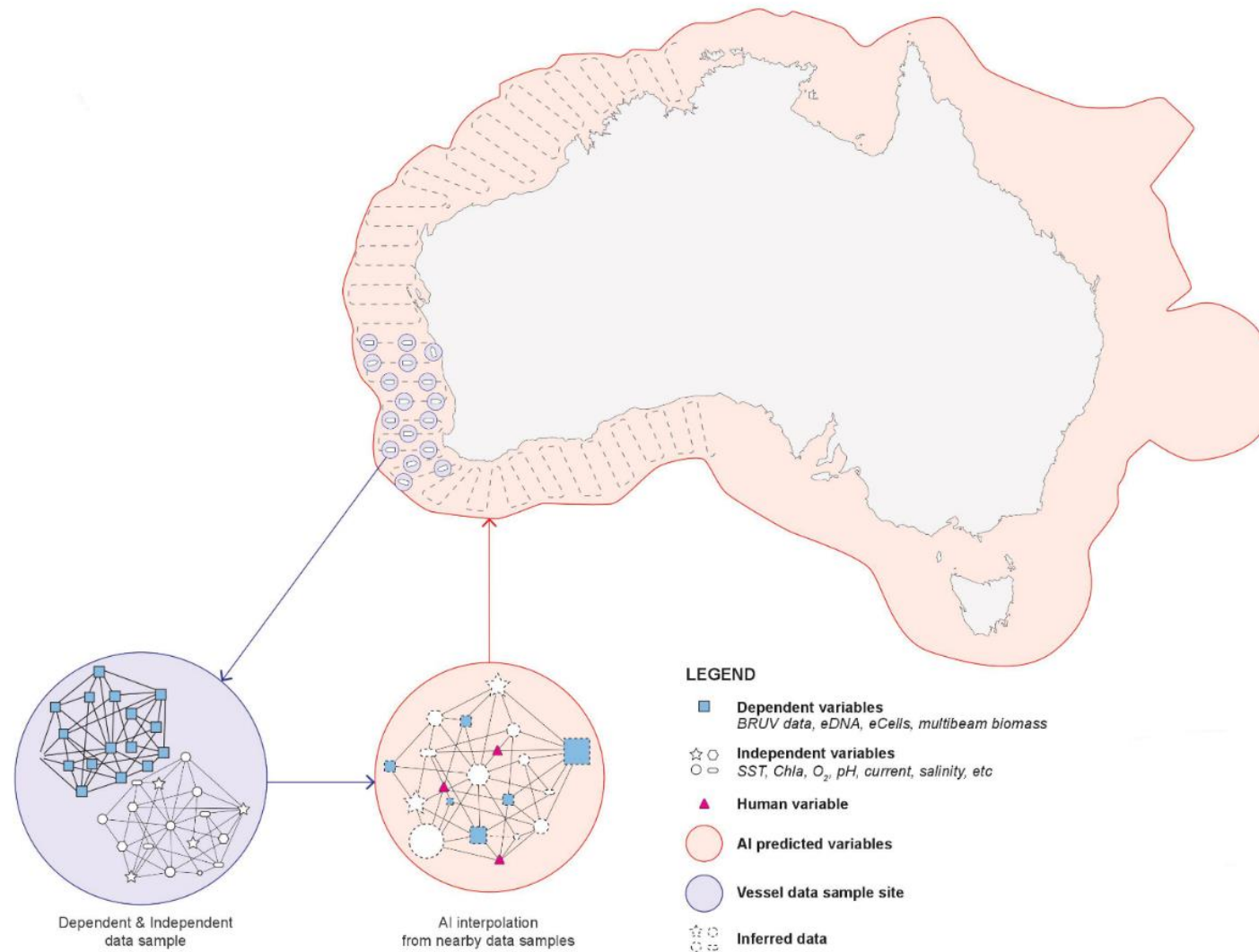


Figure 5. 2 ROMP – Survey transit that allows integration of data on ocean wildlife and environmental data, harnessing Artificial Intelligence (AI) to predict status of ocean wildlife on ocean basin scales. Transit shown as dashed line.

The home ship will be capable of surveying large distances on a daily basis and spending long periods at sea. It will be powered by emerging renewable energy, with a hydrogen production plant providing the most likely and consistent source of energy (Bicer and Dincer 2018). A multihull configuration will offer the advantage of maximal and most versatile deck area relative to overall length. For example, a 138 m long trimaran with a ~25 m beam can comfortably provide sufficient hangar capacity for up to twelve Autonomous Surface Vehicles (ASVs) and equivalent contingents of Autonomous Aerial Vehicles (AAVs) and Autonomous Underwater Vehicles (AUVs), plus ancillary launch, retrieval and garaging machinery (e.g. extendable boom cranes, underwater docking carriages, elevators, etc.). Redundancy will be critical to ensure that sampling targets are achieved in remote and often challenging environmental conditions, with 50% redundancy in AAVs, ASVs and AUVs, a percentage representative of other remote activities occurring for extended periods over lengthy periods (JAH Forrest, Fortescue Metals Group, 2019, pers. obs.). A trimaran hull also provides the optimal combination of a stable and safe platform relative to monohulls, speed and sufficient space for large wet and dry laboratories and a lecture theatre that will support global communications for educational purposes. Laboratories will include genetic, cell-sorting, trace element and image analysis equipment, with substantial -80°C freezer facilities for archiving the large numbers of environmental samples such as filtered water, tissues and sediment collected during extended surveys (Jarman et al. 2018a). Additionally, given long periods at sea, adequate space for mechanical and electrical workshops and single quarters for scientific and vessel crew are essential.

The home ship will rapidly transport fleets of autonomous sampling platforms (ASVs, AAVs and AUVs) between sampling stations, with a cumulative transit distance of 200 nm per day. During inter-site transits, continuous quantification of fish biomass and mapping of the seafloor with wide band and multibeam sonars will be undertaken (Vatnehol et al. 2017, Lacharité et al. 2018). Masthead sensors will collect a comprehensive suite of meteorological data while sea water samples are collected through sterilised intakes in the home ship's hull and filtered subsamples of seawater are robotically packed in sterile sealed containers for subsequent genetic analyses (Thomsen and Willerslev 2015). Flow-through fluorometers and Continuous Particle Image Classification Systems will analyse other subsamples of water to produce high resolution maps of chlorophyll and zooplankton composition (Davies et al. 2018), while Raman spectrography, electrochemical techniques and X-ray fluorescence will be used to assess distributions and composition of microplastics and trace elements.

While on station, the home ship will deploy the autonomous sampling platforms and continue to sample with large tethered equipment such as deep-water rosettes to collect suspended

plastic and genetic samples, conductivity temperature depth (CTD) profilers and tethered Remotely Operated Vehicles (ROVs), which can perform targeted visual census, physical collection of geological and biological specimens, and perform tasks using specialised arms (Phillips et al. 2018). Observations of previously inaccessible deep-water habitats have provided important insights into the diversity and ecology of the ocean depths, promoting protection of these under studied areas of the ocean (DeVogelaere et al. 2006, Trotter et al. 2019).

A return to flourishing oceans makes efficient data generation essential. In addition to high levels of automation throughout the vessel's navigational, sampling, laboratory processing and data analytics functions, a key operational principle of the home ship is to balance the amount of data processed on board vs. that transmitted. Automated on-board sample and data processing systems, such as video file conversion, will consolidate the volume of data that are transmitted ashore for post-processing analyses, with the vessel having substantial on-board storage to securely manage non-time critical data until it can be physically offloaded at expedition end. Transmitted data take advantage of latent capacity of shore-based facilities and capacity of cloud-based Artificial Intelligence applications, which is critical for extracting the full potential of data collected at such an envisaged scale and complexity.

5.5 AUTONOMY FROM SKY TO SEABED

The home ship will be the launch pad for multiple fleets of AAVs, ASVs and AUVs that autonomously sample a range of depths from the surface to bathypelagic depths. The fleets will employ high levels of automated navigation to undertake sampling missions away from the home ship before being recovered.

Long-range AAVs, fitted with multispectral and infra-red optics, will be sent ahead to rendezvous with the home ship at the next sampling station, conducting visual surveys for wildlife, mega plastics and evidence of illegal fishing activities in transit. AAVs are increasingly preferred over direct, often intrusive interactions which pose a risk for the observer (Lea et al. 2019) or incur behavioural bias (Coghlan et al. 2017). Developments in AAVs have increased the capacity for longer and faster transects owing to greater battery life and more powerful motors (Johnston 2018): transects can be predetermined and analysis automated (Gonzalez et al. 2016). Higher resolution cameras and sensors, satellite communication and flight-plan software advancements generate real-time, large-scale, comparable data (Chirayath and Earle 2016). These AAVs represent a versatile airborne sampling tool with the ability to simultaneously collect multiple streams of data, with recent uses including observing at-risk shark and turtle populations (Kiszka et al. 2016, Rees et al.

2018) and collecting microbiome samples from cetacean blow to assess pulmonary health (Aprill et al. 2017, Pirota et al. 2017). Their applications to marine protected areas include shallow reef habitat mapping and reef health assessment (Kobryn et al. 2013), detection of large marine debris (Veenstra and Churnside 2012), counting megafauna such as whales and seals whilst simultaneously mapping surface chlorophyll and temperature (Elarab et al. 2015), all with low emissions (Bryson and Williams 2015) and minimal noise pollution (McIntosh et al. 2018).

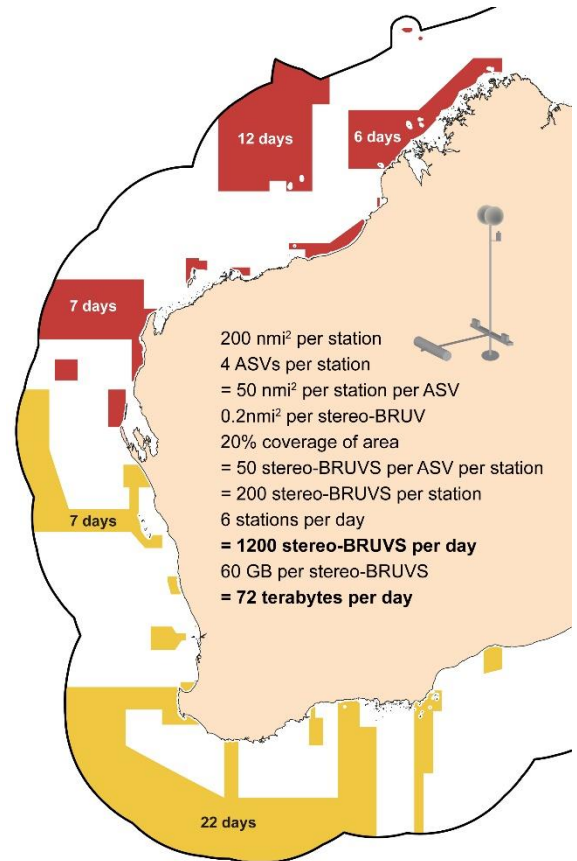
Although ASVs have yet to be applied to marine ecological applications, they are well advanced for military applications (Kitts et al. 2011). The ASV component of the ROMP will be designed specifically for triple sampling by stereo-BRUVS, water sample collection for eDNA and eCell analyses and concurrent environmental sensing. We envision scaling stereo-BRUVS data collection by an order of magnitude, from 20 to 200 samples per day, primarily by automating the deployment and retrieval of miniaturised stereo-BRUVS. New generation frames for BRUVS will be constructed of contemporary lightweight synthetic materials to minimise on-board storage constraints. High-definition BRUVS imagery will be collected from dual daylight and lowlight cameras attached to frames via quick-release docking brackets that are hard-wired to a combination battery charging and data transfer docking port on the frame. Innovative modification of auto-longline fishing clipping/de-clipping and baiting machinery will allow ASVs to rapidly deploy and recover hundreds of individual BRUVS during each sampling period. During deployment, BRUVS will be clipped to a single high-tensile mainline and shot over the bow-shaped stern as the ASVs travel ahead. During deployment, environmental data such as SST, DO, fluorescence, pH and meteorological data are measured at stereo-BRUVS sampling locations through hull-mounted and mast-head sensors on the ASV and water samples will be pumped on board and automatically filtered and/or packed for (1) quantification of microplastics and (2) eDNA and eCell analyses on the home ship/on shore. On completion of sampling, the ASV's azimuth propeller will be reversed to propel the ASV along the mainline while reeling the line aboard on a self-spooling drum reel. As they are brought aboard, the rigs will be secured in racks that connect each docking port to a central power and data transfer port on the ASV's exterior. Upon retrieval to the home ship hangar, ASVs will be connected to the ship's power supply for recharging and data are transferred to on-board storage servers.

On station, a fleet of four AUVs will be deployed from the home ship to sample depths below the euphotic zone using chemical and thermal sensors, biomass-quantifying wide band and multibeam sonars and water collection systems for *in-situ* Raman spectrographic analyses of microplastics, filtration of eDNA and eCell samples and trace element analyses. Vehicles will also be fitted with stereoscopic lowlight and illuminated 4K cameras to survey fish and

invertebrate diversity and abundance and on-board sensors record continuous high-resolution vertical profiles of the physical and chemical characteristics (conductivity, temperature, current, O₂, pH, etc.) during ascents and descents through the water column.

5.6 FROM EXABYTES TO KNOWLEDGE

The magnitude of data collected through ROMP's "disruptive" approach to science will be in the order of exabytes per year and will comprise a diversity of data streams from video imagery to genetic sequencing to multibeam sonar. ROMP will be capable of completing transects of 200 nm per day based on a maximum speed of 40 knots and 24 hour operations with, for example three hours spent on station deploying, operating and recovering sampling equipment before moving to the next station. (**Fig 5.3**) Such an operational strategy results in sampling at six stations per 24 hour period, with ~30 nm between stations. Using stereo-BRUVS as an example, such a programme would be sufficient to sample the largest marine parks in Western Australia, covering over 660,000 km², in 53 days, generating nearly 64,000 samples and 3,800 terabytes of data. With advances in image analysis automation, it is now entirely feasible to process such vast quantities of data at a similarly rapid pace (Villon et al. 2018), contributing to and building on existing large databases such as FishBase (Froese and Pauly 2019), SeaLifeBase (Palomares and Pauly 2019) and the Ocean Biogeographic Information System (www.obis.org).



5.3 Indicative scale of sampling for the major marine parks off of Western Australia

5.7 DISRUPTION

We have outlined a major disruption to how research can be conducted on global scales and in remote areas, with the aim of informing best-practice ocean conservation decision-making. The ROMP step-change responds to UNESCO's "decade-of-ocean-science" and will help ensure that humankind meets multiple Sustainable Development Goals (SDGs) relating to human well-being and sustainability (SDGs 1, 2, 3, 12 and 14; UNDP 2018). This approach is currently being ocean-tested with many of the methods being integrated on an existing research vessel (RV), the Pangaea Ocean Explorer, in order to evaluate how integrating these methods should best proceed.

Evidence is not sufficient to transform political decision-making, as demonstrated by the resilience of climate change policy to scientific consensus (Lewandowsky et al. 2012, 2015). However, in its absence, civil society struggles to shift the debate from short-term self-interest to long-term preservation. We must transform the “wicked problem” of ocean health to a tractable one. We propose that by harnessing a broad suite of recent technological developments in an advanced large high-speed vessel, it is not too late to gather knowledge about how to conserve and restore ocean health at a pace and scale exceeding the rate of anthropogenic damage. And of course, as technology continues to exponentially evolve, so will our capacity to unveil, in time, “that which is far off and exceeding deep”.

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CHAPTER 6: ELIMINATING PLASTIC POLLUTION: HOW AN INDUSTRY-LED VOLUNTARY CONTRIBUTION WILL DRIVE THE CIRCULAR PLASTICS ECONOMY

Keywords: oceans and marine, circular economy, voluntary contribution, technology, toxicity, waste, linear consumption model

6.1 ABSTRACT

Marine plastic pollution is a symptom of an inherently wasteful linear plastic economy, costing us more than US\$ 2.2 trillion per year. Of the 6.3 billion tonnes of fossil fuel-derived plastic (FFP) waste produced to date, only 9% has been recycled; the rest being incinerated (12%) or dumped into the environment (79%). FFPs take centuries to degrade, meaning five billion tonnes of increasingly fragmented and dangerous plastics have accumulated in our oceans, soil and air. Rates of FFP production and waste are growing rapidly, driven by increased demand and shifting strategies of oil and gas companies responding to slowing profit growth. Without effective recycling, the harm caused by FFP waste will keep increasing, jeopardising first marine life and ultimately humankind. In this Perspective article, we review the global costs of plastic pollution and explain why solving this is imperative for humanity's wellbeing. We show that FFP pollution is far beyond a marine environmental issue: it now invades our bodies, causing disease and dysfunction, while millions of adults and children work in conditions akin to slavery, picking through our waste. We argue that an integrated economic and technical solution, catalysed through a voluntary industry-led contribution from new FFP production, is central to arrest plastic waste flows by making used plastic a cashable commodity, incentivising recovery and accelerating industrialisation of polymer-to-polymer technologies. Without much-needed systematic transformation, driven by a contribution from FFP production, humanity and the oceans face a troubling future.

6.2 INTRODUCTION

Plastics are a ubiquitous and persistent form of marine pollution (Angiolillo 2019, Barboza et al. 2019a, 2019b, Gago et al. 2018) with contamination levels rising drastically on beaches (Barnes 2005, Kako et al. 2014, Lavers and Bond 2017), the seafloor (Matsuguma et al. 2017) and coastal and oceanic waters (Lebreton et al. 2018, Ostle et al. 2019). While our knowledge of ocean plastics is far from complete, the best available predictions suggest that with increasing mismanagement of fossil fuel-derived plastics (FFP) (Lebreton and Andrady, 2019), there will be one tonne of ocean plastic for every three tonnes of fish by 2025 (Ellen MacArthur Foundation 2017), the equivalent of 600 plastic bags for every ten-kilogram fish. The most visible impact of marine plastic pollution is its harm to marine megafauna (Galgani et al. 2019) including turtles (Lynch 2018, Wilcox et al. 2018), mammals (Panti et al. 2019), birds (Wilcox et al. 2015) and sharks (Parton et al. 2019), which ingest and become entangled in FFP with fatal consequences (Provencher et al. 2017, Stelfox et al. 2016). Millions of marine animals of at least 260 species die annually due to interactions with FFP (Claro et al. 2019, Moore 2008, Thompson et al. 2009). These numbers are likely to increase as smaller and more elusive organisms are examined (Jamieson et al. 2019). Additionally, plastics transport invasive species and pathogens between marine regions (Lamb et al. 2018, Rech et al. 2016), inhibit gas exchange between sea water and seafloor sediments (Goldberg 1997), and smother fragile seafloor inhabitants (Gregory 2009).

Marine plastic pollution not only comprises visible items, such as single-use packaging and fishing gear, but also microplastics, particles <5 mm (Hidalgo-Ruz et al. 2012), and nanoplastics, <1 µm (Gigault et al. 2018), released directly into the environment or created by the fragmentation of larger items. The legacy and reach of FFP is strikingly demonstrated by its impact on the most remote and inaccessible marine ecosystems. Jamieson et al. (2019) recently detected synthetic particles in the hindguts of the majority of crustaceans sampled in deep ocean trenches around the Pacific Rim, at depths from 7,000 m to 10,890 m, the latter being the deepest point of the ocean. Over 72% of specimens contained at least one synthetic item in their digestive systems. Concerningly, preliminary Fourier-transform infrared spectroscopy (FTIR) analyses suggest that most of these synthetic materials were produced before the 1970s (Jamieson, New Castle University, 2019, pers. obs.), implying that they have taken 50 years to reach their current resting place and that the far vaster quantity of plastic pollution generated since is still working its way through the marine ecosystem. Plastics' invasion of natural environments now appears complete, with contamination found from mountain tops and polar extremes (Bergmann et al. 2019) to the remote depths of our oceans.

Plastics' fragmentation and dispersal as micro-particles may be the most insidious property of marine plastic pollution. FFPs contain dyes, flame retardants and plasticizers, some of which are persistent, bio-accumulative toxins (Rani et al. 2015). These compounds, along with water-borne pollutants absorbed into nano- and microplastics (Engler 2012), can be transferred to organisms upon ingestion (Cole et al. 2011, Neves et al. 2015), contaminating them, their predators and potentially accumulating up food chains to human consumers of seafood (Bouwmeester et al. 2015, Lusher et al. 2017, Revel et al. 2018, Rist et al. 2018, Rochman et al. 2015b, Santillo et al. 2017, Vethaak and Leslie 2016). Nanoplastics, while understudied at present, may pose the greatest ecotoxicological risk (Haegerbaeumer et al. 2019, Koelmans et al. 2015) because their concentrations at sea are likely higher than for microplastics (Andrady 2011), they have a proportionally larger surface area for the absorption of toxic chemicals (Koelmans et al. 2015; Mattsson et al. 2015) and, critically, they can penetrate living tissues (Kashiwada 2006, Mattsson et al. 2017, Rossi et al. 2014) causing intracellular damage (Brown et al. 2001, Haegerbaeumer et al. 2019). Observed effects of nano- and microplastics on experimental organisms include increased mortality (e.g. Lee et al. 2013, Ziajahromi et al. 2018), disease (e.g. Chae et al. 2018, Vasseur and Cossu-Leguille 2006), inflammation in digestive and respiratory systems (e.g. Jin et al. 2018, Murray and Cowie 2011), increased oxidative stress (e.g. Bhattacharya et al. 2010, Gomiero et al. 2018), disturbed feeding behaviour (e.g. Cedervall et al. 2012, Nasser and Lynch 2016) and compromised fecundity and reproduction (e.g. Bergmann et al. 2015, Sussarellu et al. 2016).

In this Perspective article, we argue that ocean plastic pollution is a symptom of a far broader issue, rooted in the way we place little to no value on a versatile material made from a finite resource, whose environmental legacy lasts decades following its brief usefulness to us. The solution to plastic pollution cannot rely solely on more ocean research (Borja and Elliott 2019), education (Uyarra and Borja 2016) and clean-up technologies (Sherman and van Sebille 2016), nor in phasing out or replacing all FFPs (Rochman et al. 2015a). It requires us to shift the economics of FFPs from a linear to a circular model. We first review the cost of the plastic pollution problem and then propose a pragmatic solution to fix it, which builds beyond current proposals.

6.3 THE PRICE OF PLASTIC

Plastic pollution results from our failure to account for the full economic cost of FFP's manufacture and disposal, and its impacts on ecosystem services and human health. While our knowledge is incomplete, best estimates suggest that plastic costs humanity over US\$ 2.2 trillion per year in environmental and social damage (Beaumont et al. 2019, Ricke et al.

2018, UNEP 2014, Zheng and Suh 2019). This is a consequence of a linear economic model in which resources flow unidirectionally from fossil fuels, are cracked into monomers (the building blocks of plastic ‘resins’), extruded into a final product, used, often briefly and then discarded (Geyer et al. 2017, Hundertmark et al. 2018; **Figure 6.1a**). It is estimated that nearly 60% of this plastic waste is dumped into landfill and the environment, with at least 10% entering the oceans (**Figure 6.1b**; Supplementary Materials ‘6.12.1. Plastic leakage to oceans’).

Most of the costs of plastic pollution damage results from impacts on our oceans (**Figure 6.1c**; Supplementary Materials ‘6.12.2. Plastic pollution damage costs’). An estimated ~US\$ 1.5 trillion per year is lost through reductions in the oceans’ capacity to provide seafood, genetic resources, oxygen, clean water and recreational and cultural value, as well as critical regulation of Earth’s climate (Beaumont et al. 2019). An additional ~US\$ 730 billion per year in losses occurs during FFPs’ upstream lifecycle (UNEP 2014, Zheng and Suh 2019), due to a model of plastic waste management that is more ‘burn, bury, befoul’ than ‘reduce, reuse, recycle’. These costs include ~US\$ 700 billion per year from the release of greenhouse gases during FFP production and waste incineration (Ricke et al. 2018, Zheng and Suh, 2019), the release of toxic chemicals from plastics buried in landfill to soils and water sources (~US\$ 25 billion per year); water usage during plastic production and manufacturing (~US\$ 4.5 billion per year); release of plastic-associated pollutants into the air (~US\$ 1.3 billion per year); and land value loss due to littering or proximity to waste disposal sites (~US\$ 875 million per year; UNEP 2014).

The estimated total cost of plastic pollution is likely conservative, as several important impacts are yet to be quantified, particularly those related to human health. Plastics can harm us both through the interaction of nanoplastics with human cells and our exposure to harmful additives in plastic products (Hermabessiere et al. 2017, Revel et al. 2018). Both nanoplastics and harmful additives occur in food packaging, household items and even medical equipment, entering the body via ingestion, inhalation and skin contact.

Nanoplastics have been shown to cause damage and inflammation in human skin, lung and brain cells (Lehner et al. 2019) and may be linked to cancers (e.g. Mastrangelo 2003; Pauly et al. 1998). Plastics also leach harmful endocrine-disrupting chemicals (Meeker et al. 2009, Talsness et al. 2009) which have been linked to:

- Cancer (Brophy et al. 2012, DeMatteo et al. 2013, Ohlson and Hardell 2000)
- Obesity (Angel Nadal 2012, Manikkam et al. 2013)
- Diabetes (Lang et al. 2008, Shankar and Teppala 2011)

- Endocrine system disorders (Andra and Makris 2012, Brophy et al. 2012)
- Thyroid dysfunction (Ahmed 2016)
- Reproductive impairment (Kabir et al. 2015)

Infants and children are the most vulnerable groups, due to their greater sensitivity and higher exposure to plastic-associated chemicals via baby food packaging (Fantoni and Simoneau 2003), children's toys (Turner 2018, Xie et al. 2015) and breast milk (Tanabe and Kunisue 2007). Plastic contamination in humans has been detected globally (Barboza et al. 2018, Koch and Calafat 2009), with the average US citizen consuming more than 74,000 microplastic particles annually (Cox et al. 2019) and an unknown but likely larger number of nanoplastics (Triebkorn et al. 2019). Further research is urgently required into the human health impacts and associated health-care costs of plastics and their ingredients.

A further unevaluated cost of plastic is the prevalence of acute labour issues in the waste management systems of many low and middle-income nations, where collection, recycling and disposal of domestic and imported waste are largely unregulated. The informal recycling sector employs an estimated 15-20 million workers globally (UNEP 2015) and often creates abusive and hazardous conditions for a meagre but crucial income (GAIA 2019; Walk Free 2018, Wilson et al. 2006). This highlights a moral dimension of plastic pollution: profits from fossil fuel extraction and plastic production typically accrue to a small number of companies headquartered in high-income nations, while waste disposal, burning and dumping, including of imported waste from wealthy countries, are usually shifted to low to middle-income nations (GAIA 2019). This disconnect between production and disposal also weakens the impetus for consumers in rich, high consumption countries to shift behaviour, since they are insulated from the consequences of their plastic habit (Torras and Boyce 1998).

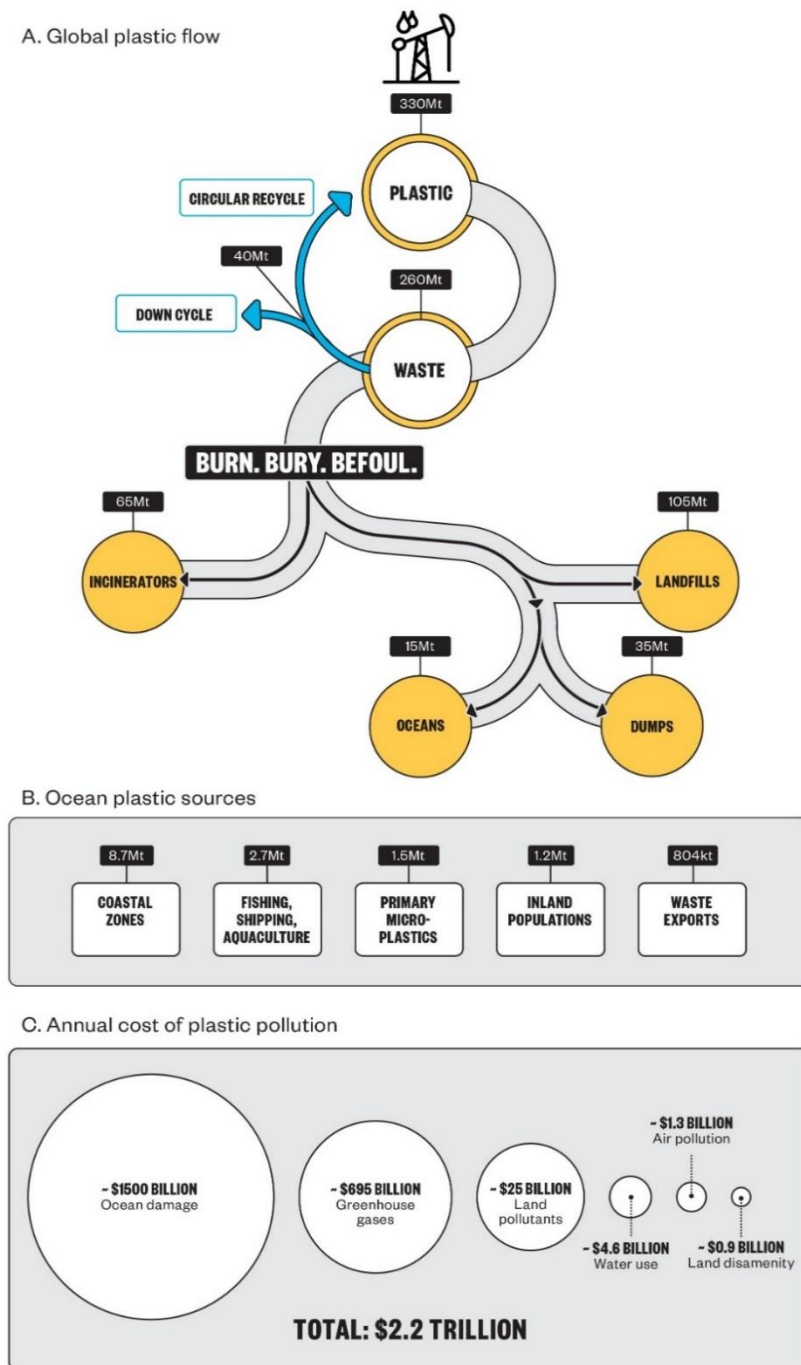


Figure 6. 1 The plastic problem. **(a)** Global plastic flow in 2016, as estimated by Hundertmark et al. (2018). Numbers within black boxes represent amounts (million tonnes per year) of plastics produced, wasted, recycled, incinerated, landfilled, dumped and leaked to oceans. **(b)** Break-down of the different sources of ocean plastics, in million tonnes per year: coastal zones (Jambeck et al. 2015), fishing (Watson et al. 2013, Bell et al. 2017, Lebreton et al. 2018), shipping (Halpern et al. 2008; Lebreton et al. 2018), aquaculture (FAO 2016a, Lebreton et al. 2018), primary microplastics (Boucher and Friot 2017), inland populations (Lebreton et al. 2017, 2018), and waste exports (UN trade statistics 2004). **(c)** Break-down of the different costs associated with plastic pollution in US\$ per year: ocean damage (Beaumont et al., 2019), greenhouse gases (Ricke et al. 2018; Zheng and Suh 2019), land pollutants, water usage, air pollution and land disamenity (UNEP 2014). The data, limitations and assumptions used for the plastic mass flow and environmental economic analysis are described in Supplementary Material 6.12.1 'Plastic leakage to oceans' and 6.12.2 'Plastic pollution damage costs'.

6.4 PRINCIPLES OF A SOLUTION

The US\$ 2.2 trillion annual external cost of plastic pollution is not captured in the production costs of our linear plastics economy, representing a major market failure (Laffont 2008). Whilst initiatives such as Extended Producer Responsibility (EPR) and plastics-related legislation have made some positive inroads to reduce plastic pollution, a system-wide change is needed to rectify this market flaw. A pragmatic solution is to apply an appropriate voluntary contribution to FFP at first production, whereby the supply chain passes on this price premium on raw FFP resin equitably through to the trillions of plastic items purchased each year by end consumers, converting plastic waste into a cashable commodity. This voluntary contribution, promoted publicly as the “Sea The Future” initiative, but referred to herein as “the Contribution”, will also generate considerable funds to tackle the plastic issue, via investment in transformative technologies and by funding environmental remediation. Such a voluntary, industry-led contribution for FFP, applied at the resin production level, has the greatest potential to drive global manufacturing towards a circular economy (**Figure 6.2**; Schepel 2005).

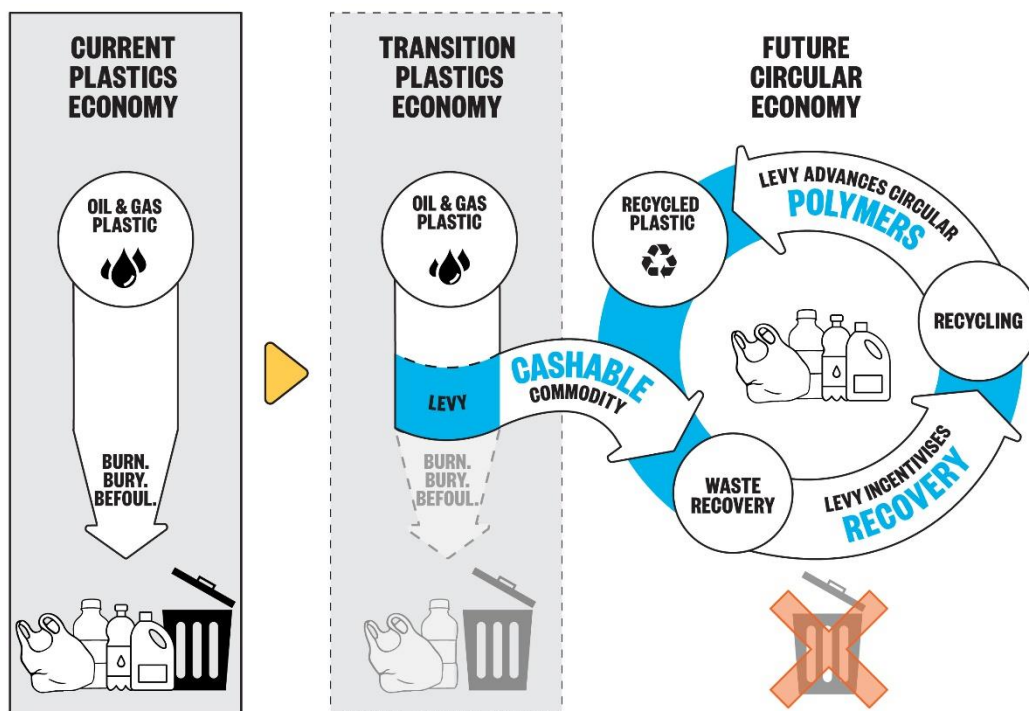


Figure 6. 2 Illustrations of the linear plastics economy (left), circular polymers economy (right), and transition plastic economy (centre). Our proposed voluntary contribution will re-direct the flow of fossil fuel-derived plastics away from the environment and into the economy.

A circular plastics economy has remained elusive despite decades of concerted advocacy and public outcry. The key barrier to its realisation has been the inability of circular recycling technologies to compete with the extremely low direct cost of producing FFPs. This perverse market price signal has meant that emerging technologies which can infinitely recycle most used plastic into high-purity polymers (e.g. Ragaert et al. 2017, Rahimi and García 2017) have failed to achieve global-scale commercialisation. A higher cost of FFPs, applied via the Contribution, levels the playing field and should drive plastic producers to rapidly seek out lower cost feedstocks. Demand for recycled polymers will ignite, transforming plastic waste from toxic and destructive into a cashable commodity, incentivising recovery and recycling rates. This will have a transformative effect on the recycling industry as the next generation of polymer-to-polymer technologies are modularised, thereby enabling both an extremely low capital cost compared with traditional large scale refineries and petrochemical plants (IHS Chemical 2015) and placement at aggregation points of plastic waste (e.g. rivers, garbage depots, mine sites). This powerful competitive advantage, together with a level playing field, will energize new entrepreneurs and recycling businesses to contribute to the circular economy in both developed and developing markets (Baechler et al. 2013). Concurrently, mobile applications and artificial intelligence (AI) are set to remodel and decentralise waste collection services (Adams 2018, Coelho et al. 2019). These peer-to-peer technologies have tremendous potential to connect billions of people currently not serviced by formal waste collection systems, increasing recovery rates without the need for expensive waste collection infrastructure. Access to this technology may also contribute to improving the livelihood of millions of disenfranchised waste-pickers through improved transparency, security and compensation (Walk Free 2018).

The Contribution would also directly (via investment) and indirectly (via demand) support advances in renewable, compostable biomaterials including 'edibles' and polyhydroxyalkanoates (PHAs) (Dilkes-Hoffman et al. 2019a, Shit and Shah 2014) derived from sustainable sources such as seaweed (Rajendran et al. 2012) and biomass residues (FAO 2016b). These are ideal materials for problematic applications such as sachets and agricultural films (Dilkes-Hoffman et al. 2019b) as well as aquaculture and fishing gear likely to be lost at sea (Bilkovic et al. 2012, Bugnicourt et al. 2014, Kim et al. 2016, Park et al. 2010, UNEP 2018).

In summary, the Contribution achieves the following:

- Transforms plastic waste into a cashable commodity, rewarding recovery and increasing recycling rates.
- Incentivises the plastic industry to reduce its use of fossil fuel feedstocks and seek out recycled and degradable alternatives.
- Makes a material contribution to reducing climate change in the context of the global community's targeted temperature increase range, when considered on an accumulative basis to 2050 (Zheng and Suh 2019).
- Halts the rising health impacts of plastics on both humans and other species and ensures viable ecosystem services.
- Materially improves the profitability of polymer-to-polymer technologies and other supporting industries.
- Is complementary to and supportive of the 'Three Rs' philosophy, 'Reduce, Reuse, Recycle', promoted by the circular economy community (Ellen MacArthur Foundation 2019).
- Prioritises technologies which keep plastics within the economy, e.g. purification, depolymerization and pyrolysis technologies (Sardon and Dove 2018, The Center for the Circular Economy 2019).

How will the Contribution be implemented?

The most effective point to apply and collect the Contribution is at resin production, ensuring that it can be practically implemented and administered. This concentrated point in the supply chain, comprised of only a small group of producers (American Chemistry Council 2013), facilitates the application of the Contribution on a simple, equitable and transparent per-weight basis, streamlining stakeholder participation and industry-wide application. As the cost of the Contribution is passed through intact from the base of the supply chain to the point of final consumption, it is divided out via manufacturing intermediaries, packaging companies and retailers (UNEP 2014), to the end consumers, who each experience only a small price increase. To anticipate potential anti-trust concerns regarding the implementation of the Contribution, the lead author has engaged with global law firms to investigate the issue and believes that an initiative in the public good can comfortably operate within the law in countries across the world.

How will the Contribution be governed?

Strict governance of the Contribution is paramount to avoid issues such as corruption and gaming of the system and to ensure that intra- and international value transfer is equitable

and efficient, as the revenues from FFP production and the costs from its polluting effects often accrue in different regions (Abbott and Sumaila 2019). Extended Producer Responsibility (EPR) policies (European Commission 2019, Hanisch 2000, Hilton et al. 2019; Spicer and Johnson 2004) provide a template for a global scheme. EPRs often rely on a Producer Responsibility Organisation (PRO) to act on behalf of the stakeholders and operate under clearly defined governance arrangements (Börner and Hegger 2018, Fleckinger and Glachant 2010, Park et al. 2018). Here, a global audit system will play a critical role and to circumvent concerns regarding governance costs the Minderoo Foundation has committed to underwrite five years of audit fees for a total cost of US\$ 260 million, plus US\$ 40 million in establishment costs, subject to appropriate conditions. As a further protection mechanism against fraud, a certification process would be implemented that could leverage blockchain technologies to track provenance (Kim and Laskowski 2018), supported by technologies allowing identification of plastics via their chemical signatures (Primpke et al. 2018, Serranti et al. 2018). This potentially facilitates producer-specific labelling of plastics, informs consumer purchasing decisions and further drives industry adoption. A global PRO would allow funds raised from the Contribution to be aggregated and distributed according to a constitution that ensures efficient governance and management, timely allocation of funds and maximises environmental, social and health benefits. Four constitutional pillars will be prioritised as follows:

1. Drive the global deployment and industrialisation of polymer-to-polymer technologies and associated infrastructure as well as support the segments of the incumbent supply chain vulnerable to the impact of the Contribution, during the transition from FFP to polymer-to-polymer production
2. Provide pollution remediation where a market solution is not feasible, such as sedimentary build-up of plastics in rivers and landfills
3. Recovery of oceanic plastics
4. Other environmental policies as agreed, which may include mitigating climate change.

The first author has reached a preliminary agreement on these four pillars with a major consumer-facing organisation. With respect to (1), the consensus reached was that the petrochemical businesses are the most vulnerable to the impact of the Contribution, but conversely have a key role to play in achieving a circular plastics economy (Hundertmark et al. 2018). It was agreed, under an initiative known as the Bridging Scheme for Industry (BSI), that part of the Contribution proceeds will be used to facilitate the industry-wide transition from fossil fuels to plastic waste, which eventually becomes its sole feedstock.

As the capital raised by the Contribution is estimated to reach at least US\$ 20 billion per annum, these funds may eventually exceed the quanta required for priorities (1) to (3). While this excess could be fully refunded, it may instead be attractive to apply the balance to the mitigation of anthropogenic climate change, largely a by-product of the fossil fuel industry, under pillar (4). Given consumers have ultimately paid the cost of the Contribution, this would be an attractive outcome for the fossil fuel industry.

If not a voluntary, industry-led contribution, then what?

Should industry fail to act, then alternatives to the Contribution include government-led taxes, heightened industry regulation or joining international treaties with similar force. However, each of these interventions has drawbacks in the context of the complexity of plastics. Global treaties like the Montreal Protocol on Substances that Deplete the Ozone Layer, and the Stockholm Convention on Persistent Organic Pollutants, regulate the manufacture and application of specific chemicals with relatively narrow use cases, like chlorofluorocarbons and the pesticide dichlorodiphenyltrichloroethane. While both are supported by multilateral compensation funds and have successfully assisted countries to develop and promote safer alternatives to these niche chemicals, they do not address a problem as pervasive and embedded in every part of the consumer economy as plastics. Our most visible attempt to solve such a problem with an international treaty, the Kyoto Protocol and subsequent Paris Agreement, continues to attract criticism from environmental groups, academics and governments for the range of exemptions and trade-offs it offers to individual countries and industries, and its failure to achieve its central goal of reducing carbon emissions. Consequently, the US has yet to ratify the treaty and Canada has withdrawn from the protocol.

Compared with a global voluntary industry contribution, taxes and regulation in individual jurisdictions open the door for regulatory arbitrage and the requirement for border taxes on plastics imports. Trillions of plastics items are traded globally daily in varying different forms, from resin pellets to finished products, and often contain compositions of many different plastic types and other materials. As a result, the task for governments of administering taxes and regulations on goods, both domestically and across borders, is onerous and complicated, potentially compromising their effectiveness. Local governments may also invest tax revenue in programs unrelated to resolving the plastic problem, compromise on policies unpopular with voters and be unwilling to redistribute proceeds to other countries, making it difficult to deal with the issues in many impoverished nations subject to exported waste.

The Contribution is a global solution with the potential to overcome many of these shortcomings by transcending the compliance issues related to a mosaic of national taxes, laws and treaties. Self-imposed industry levies have also previously been applied successfully to challenges of cost recovery in the agriculture (OECD 2017), fisheries (Townsend et al. 2008) and media industries (Leeds 2006).

Ultimately, the Contribution and other interventions should not be seen as mutually exclusive and, in fact, will likely play complementary roles. For example, regulations can and should be used to tackle specific situations such as environmental leakage of nano- and microplastics and use of harmful additives in plastic products. In these cases, immediate alternatives are available, including compulsory filters on washing machines (McIlwraith et al. 2019), upgrades in wastewater treatment plants (Talvitie et al. 2017), regulatory frameworks to better prevent plastic pellet spills (Karlsson et al. 2018) and bans of both toxic plastic additives (Halden 2010, Kole et al. 2017, Lahimer et al. 2017) and nano- and microplastics as ingredients in products (Hernandez et al. 2017, Rochman et al. 2015a).

What should the quantum of the Contribution be?

An estimated contribution in the range of US\$ 200 to US\$ 5,000 per tonne is required to incentivise the collection and recycling of used FFP, with the quantum of the Contribution depending on the type of polymer. This equates to between ~20% and 500% of the cost of FFP resin (The Plastics Exchange 2019), translating into only a ~US\$ 1 to 3 cent increase in the cost of a take-away coffee, as an example. Assuming an average contribution of US\$ 500 per tonne of FFP resin, the total cost of the Contribution is less than 10% of the US\$ 2.2 trillion in damages currently caused by FFP pollution. Our suggested contribution range is based on anecdotal estimates collected during discussions with major stakeholders in the plastics supply chain, including oil and gas companies, resin producers, consumer goods brands, retailers and recycling businesses. As such this quantum is preliminary, with further economic modelling required to triangulate other factors, including the different incentive levels required to collect waste in different regions, capital market incentives and the price elasticity of plastics demand.

6.5 CONCLUSION

Plastic pollution damages societies, economies and natural environments, particularly the world's oceans. With plastic pollution's increasing visibility on land and at sea, pressure for action is mounting while an effective global solution to this 'wicked problem' remains elusive. While removing plastic litter from oceans (Sherman and van Sebille 2016) and replacing plastics with other materials (Dilkes-Hoffman et al. 2019b, Song et al. 2009) contribute to a

solution, we suggest that a far broader economic and technical approach is needed to catalyse change. The proposed initiative, a voluntary contribution on new FFP production led by the global plastics industry, should jump-start the circular economy by transforming plastic waste into a cashable commodity. The new economics will help catalyse global-scale commercialisation of polymer-to-polymer technologies capable of creating food grade polymers from plastic waste, while drawing plastic pollution from the environment back into the economy. If implemented successfully, the effects can be far-reaching: stopping the flow of plastics into oceans, giving economic opportunity to vulnerable people, funding remediation of contaminated ecosystems and protecting future generations from the toxicity of plastic waste.

6.6 CONFLICT OF INTEREST

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest. Minderoo Foundation has committed to underwrite five years of audit fees for a total cost of US\$ 260 million, plus US\$ 40 million in establishment costs, subject to appropriate conditions. The authors also declare that any existing and future investments that may form part of the FFP supply chain will be subject to the proposed initiative.

6.7 AUTHOR CONTRIBUTIONS

AF conceived the idea; AF, JR, DT, LG delineated the research; AF, JR, DT, LG collated data from the literature; AF, LG, DT, JR, JM wrote the manuscript; LG made the figures. All authors revised the manuscript.

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6.10 DATA AVAILABILITY STATEMENT

N/A

6.11 REFERENCES

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6.12 SUPPLEMENTARY MATERIALS

6.12.1 PLASTIC LEAKAGE TO OCEANS

Total plastic leakage to the ocean was estimated at 15 million tonnes per year by compiling previously published sources and combining their estimates for individual contributions (**Supplementary Table 6.1.1**). Sources of ocean plastics included macroplastics (greater than 5 mm) from coastal and inland populations (including mismanaged plastic from imported waste streams); macroplastics from ocean-based sources (fisheries, shipping and aquaculture); and primary microplastics (smaller than 5 mm).

Our estimate of the total plastic load going to oceans is the highest ever published in the peer-reviewed literature (~15 million tonnes per year). Nonetheless, we believe this is still likely a conservative estimate, based on our own unpublished surveys in SE Asia as well as the age of some of the available data.

Supplementary Table 6.1. 1 Estimated plastic leakage to oceans (tonnes per year)

Source category	Total (t)	Source
Macroplastic leakage from coastal zones	8,762,950	Jambeck et al (2015)
Macroplastic leakage from inland areas	1,200,000	Lebreton et al (2017)
Macroplastic leakage from waste exports	804,473	United Nations Commodity Trade Statistics Database (2018 data), Jambeck et al (2015) mismanagement and leakage rates
Macroplastic leakage from ocean-based sources	2,737,000	Lebreton et al (2018) based on data from Watson et al (2013), Bell et al (2017), Halpern et al (2008), FAO (2016), Ocean Conservancy (2011), Arcadis (2012) and Eunomia (2016).
Primary microplastics	1,500,000	Boucher & Friot (2017)
Total	15,004,423	tonnes per year plastic leakage to the ocean

Macroplastic leakage from coastal zones

Rates of waste mismanagement for 192 coastal nations were modelled by Jambeck et al. (2015). Jambeck et al. applied these mismanagement rates to the estimated volume of plastic produced by populations living within 50 km of the coast to calculate the volume of mismanaged plastic in coastal zones. To calculate the amount of plastic entering oceans, they then assumed that between 15% (low) and 40% (high) of mismanaged plastic waste was leaking to oceans. We took the mid-point (27.5%) of this range as our estimated leakage rate to the ocean: applied to the mismanaged plastic waste volumes estimated for each country's coastal population, provided in the supplementary materials of Jambeck et al. (2015), this yields an estimate of 8.76 million metric tonnes (Mt) of plastic per annum (range 4.78 - 12.75 Mt) entering oceans from coastal populations (**Supplementary Table 6.1.2**).

Supplementary Table 6.1. 2 Estimated plastic leakage to oceans (tonnes per year) from coastal populations of the top 20 polluting countries, as ranked by Jambeck et al., 2015. Mid point assumes 27.5% of mismanaged plastic waste enters oceans, lower bound assumed 15%, upper bound assumes 40%.

Country	Mid-Point (t)	Lower bound (t)	Higher bound (t)
1 China	2,425,422	1,322,958	3,527,887
2 Indonesia	884,635	482,528	1,286,742
3 Philippines	518,006	282,549	753,463
4 Vietnam	504,300	275,073	733,528
5 Sri Lanka	437,574	238,677	636,471
6 Thailand	282,628	154,161	411,096
7 Egypt	265,928	145,052	386,805
8 Malaysia	257,625	140,523	374,727
9 Nigeria	234,160	127,724	340,597
10 Bangladesh	216,515	118,099	314,931
11 South Africa	173,251	94,501	252,002
12 India	164,950	89,973	239,928
13 Algeria	143,153	78,083	208,222
14 Turkey	133,633	72,891	194,375
15 Pakistan	132,136	72,074	192,197
16 Brazil	129,636	70,711	188,562
17 Myanmar	126,024	68,740	183,308
18 Morocco	85,285	46,519	124,050
19 North Korea	83,690	45,649	121,731
20 United States	75,742	41,314	110,170
Sum	7,274,294	3,967,797	3,967,797
Sum of all 192 nations	8,762,950	4,779,791	12,746,109

Macroplastic leakage from inland areas

We used the Lebreton et al (2017) estimate of plastic inputs from inland areas (> 50km from coastline) to oceans. They estimated inland populations contributed between 0.79 and 1.52 Mt per year of plastic to oceans via river transport. We took the mid-point¹ calculated from the average of lower and upper range, 1.2 Mt per year.

Macroplastic leakage from waste exports

Per capita generation of plastic waste is much larger in high-income countries. For example, OECD countries are estimated to generate an average of 2.2 kg per capita per day *versus* sub-Saharan Africa at an average of 0.65 kg per capita per day (Hoorweg and Bhada-Tata, 2012). High labour costs and low profitability from plastic recycling has driven the growth of plastic waste exports from high-income nations to low and middle-income nations whose recycling industries operate at lower cost (Brooks et al., 2018). However, less-developed waste management infrastructure in these destination countries results in higher waste mismanagement rates than in the wealthier source countries (Jambeck et al., 2015). The practice of exporting waste therefore potentially leads to additional plastic leakage to oceans through the mismanagement of plastic waste in destination countries, and likely exacerbates other harmful impacts resulting from, e.g., the uncontrolled burning of waste (GAIA, 2019).

We estimated the leakage of exported plastic waste to oceans using trade data (UN Trade Statistics) and the estimates of waste mismanagement rates in destination countries of Jambeck et al. (2015). We extracted UN trade data for 2018 (imported and exported) under the commodity code 3915, "Waste, parings and scrap, of plastics"². We based the analysis on records from 2018 as it is the only year of data available since the implementation of strict new waste import restrictions in China, historically the largest importer of plastic waste (Brooks et al., 2018). We extracted official trade records for plastic imports and exports by both reporter and partner trade countries. We compared the volume of plastic traded from paired reporter-partner reports and took the higher of the two reported figures to limit underreporting and fill gaps where only one nation reported trade data. We calculated that in

¹ In the absence of information on the underlying distribution of source data published as ranges, we have assumed the midpoint as a proxy of median value.

² Available from: <https://www.taricsupport.com/nomenclatuur/3915000000.html>

2018, for all records available for extraction on 26th July 2019, an estimated 9.4 Mt of plastic waste was traded globally.

For each plastic importing nation, we applied estimated rates of waste mismanagement from Jambeck et al. (2015) to their imported plastic waste volumes, then calculated potential leakage to oceans using the mid-point and range of leakage rates used above (27.5%; range 15% - 40%). Landlocked countries were assumed to have a leakage rate equal to zero. The estimated volume of plastic waste potentially leaking to oceans was summed for each exporting nation, based on the countries to which waste was sent, to estimate their contribution to ocean plastic pollution through exported waste. By summing these estimates across all exporting countries, our analysis reveals an additional 0.80 Mt of macroplastic (range 0.44-1.18 Mt) potentially entering oceans every year due to plastic waste export and mismanagement (**Supplementary Table 6.1.3**).

We recognise there is a fluid international policy situation as other countries join China in banning some traded waste, and the assumption that mismanagement rates of imported plastic waste is equal to that occurring within domestic waste management systems is unverified. Therefore, we consider these figures as indicative of the potential for mismanagement to occur and stress that further research is required to quantify plastic leakage to oceans via the waste trade.

Supplementary Table 6.1. 3 Estimated plastic leakage to oceans (tonnes per year) from plastic exports for the top 20 countries as ranked by plastic mass exported (left) and leaked to oceans via exports (right). 'Kg per capita' columns show kg of exported plastic per capita (left) and kg of exported plastic per capita potentially leaking to oceans via waste exports (right). These estimations are based on data from UN trade statistics for the 2018 and Jambeck et al. (2015).

Top 20 exporters		Tonnes (2018)	Kg per capita (2018)	Top 20 polluters via leakage to oceans from exported plastic waste	Tonnes (2018)	Kg per capita (2018)
1	Germany	1,266,787	15	Japan	143,243	1.1
2	United States	1,217,145	4	United States	142,360	0.4
3	Japan	1,035,421	8	Germany	79,277	1.0
4	United Kingdom	756,106	11	United Kingdom	63,514	0.9
5	France	441,705	7	China - Hong Kong SAR	57,065	7.7
6	Netherlands	344,419	20	Thailand	28,415	0.4
7	Mexico	323,338	3	Marshall Islands ³	20,562	352.0
8	China - Hong Kong SAR	302,511	41	Australia	19,818	0.8
9	Italy	274,870	5	Spain	19,794	0.4
10	Poland	231,095	6	Netherlands	18,973	1.1
11	Austria	206,644	23	Indonesia	17,200	0.1
12	Canada	197,545	5	South Korea	15,477	0.3
13	Thailand	196,463	3	Saudi Arabia	12,665	0.4
14	Spain	187,411	4	Canada	10,428	0.3
15	Slovenia	128,308	62	Mexico	10,354	0.1
16	Australia	127,381	5	Poland	9,798	0.3
17	Czech Rep.	121,512	11	China	9,304	0.0
18	Indonesia	106,576	0	Italy	9,286	0.2
19	Switzerland	99,412	12	France	8,961	0.1
20	Sweden	96,215	10	Philippines	8,693	0.1
Global total traded		9,403,457		Global total leakage	804,473	
Total from the top 20		7,660,862		Total from the top 20	705,187	
Proportion of plastic waste exports generated from top 20 exporters		81.5%		Estimated proportion of plastic waste exports leaking to oceans	8.6%	

Macroplastic leakage from ocean-based sources

³ The per capita figure for the Marshall Islands is an outlier relative to larger countries due to the fact that their waste management, in common with most other small island countries, relies disproportionately on exports.

We used the estimations for ocean-based sources of marine plastic pollution estimated by Lebreton et al (2018). We took the mid-point calculated from the average of lower and upper ranges for fishing, aquaculture and shipping as outlined in Supplementary Table 6 of Lebreton et al (2018) (**Supplementary Table 6.1.4**).

Plastic waste inputs from fishing were derived from fishing effort hotspots using data from Bell et al (2017) and Watson et al (2013). Estimates for aquaculture were derived from the UN FAO database for mariculture (ocean-based aquaculture; <http://www.fao.org/aquastat/en/>). Finally, estimates for shipping were estimated using gridded shipping frequency data from Halpern et al (2008). For further details on how these estimates were calculated, see the supplementary materials from Lebreton et al (2018)⁴.

Supplementary Table 6.1. 4 Estimated plastic leakage to oceans (tonnes per year) from ocean-based sources, as estimated by Lebreton et al (2018).

Sources	Range	Mid-point
Fishing	290,000 – 3,500,000	1,895,000
Shipping	100,000 – 1,400,000	750,000
Aquaculture	14,000 – 170,000	92,000
Total		2,737,000

Primary microplastics

While the degradation of macroplastics into fragments is the major source of ocean microplastics, primary microplastics – those directly released into the environment as particles smaller than 5 mm – represent an additional source of debris. Boucher and Friot (2017) estimated 1.5 Mt of microplastics (range 0.8 Mt – 2.5 Mt) entering oceans per year from the laundering of synthetic textiles, abrasion of car tyres, fallout of city dust, abrasion of road markings, release of marine coatings, microbeads from cosmetics, and spills of plastic pellets.

We used the reported per cent contributions to estimate the volume for each of the major source categories reported (**Supplementary Table 6.1.5**). For further details on how estimates were calculated, see the original report by Boucher & Friot (2017).

⁴ Available from https://static-content.springer.com/esm/art%3A10.1038%2Fs41598-018-22939-w/MediaObjects/41598_2018_22939_MOESM1_ESM.pdf

Supplementary Table 6.1. 5 Primary microplastics leakage to oceans (tonnes per year), as estimated by Boucher & Friot (2017).

Sources	%	Total
Laundering of synthetic textiles	35.0	525,000
Abrasion of car tyres	28.0	420,000
Fallout of city dust	24.0	360,000
Abrasion of road markings	7.0	105,000
Release of marine coatings	3.7	55,500
Release of microbeads from cosmetics	2.0	30,000
Spills of plastic pellet	0.3	4,500
Total		1,500,000

6.12.2 PLASTIC POLLUTION DAMAGE COSTS

To calculate the economic value of the damage caused to the environment from the plastics industry, we combined information from three sources; UNEP (2014), Beaumont et al. (2019) and Zheng and Suh (2019). While UNEP considered the decline in natural capital as a result of plastics across their life cycle, Beaumont et al. (2019) assessed the impact of ocean plastics on marine ecosystem services specifically and Zheng and Suh (2019) assessed the contribution to greenhouse gases. The latter two resources were considered more recent and comprehensive estimates of specific aspects of environmental damage and were therefore chosen to supersede the UNEP life cycle analysis in their respective categories. The UNEP breakdown of costs by commercial sector and cost category was applied *pro rata* to the other data to assign aggregate costs to particular product streams.

Land and water pollutants / Additives, Water use, Air pollutants and Land disamenity

UNEP (2014) applied a natural capital approach to quantify the upstream (plastic inputs to manufacturing) and downstream (post manufacturing) impacts of plastic products across industries. We extracted the predicted contribution of natural capital costs from each sector from the figures provided in their Appendix 1: Sector Specific Results (UNEP, 2014, page 55-76) using a web based plot digitiser (<https://automeris.io/WebPlotDigitizer/>).

Attempts to quantify these upstream and downstream impacts are hindered by the absence of robust data across the complexity of global industries and the variable scales of environmental impacts. Therefore, where more comprehensive or recent damage estimates were available, we substituted these for the equivalent categories in UNEP (2014); 'Marine Impacts' was superseded by Beaumont et al.'s *Marine ecosystem services delivery* (2019); 'Greenhouse gases' was replaced by Zheng and Suh's greenhouse gas analysis (2019);

UNEP categories 'Terrestrial pollutants to land and water' and 'Additives' were summed for brevity; UNEP categories 'Water use', 'Air pollutants' and 'Land disamenity' were left unchanged.

Declining marine ecosystem services delivery

After Beaumont et al (2019), we assume marine ecosystem services provided benefits to society approximating US\$ 49 trillion per year (Costanza et al., 2014). Following their estimates of a 1 - 5% decline in marine ecosystem service delivery due to the stock of marine plastics in 2011, this decline equals an estimated US\$ 500 - 2,500 billion annual loss. We take the mid-point estimate of their calculations to assume an annual loss worth US\$ 1.5 trillion. To allocate the cost of lost marine ecosystem services to different commercial sectors, the sectoral allocation of costs in the 'maritime services' category in the UNEP report was applied *pro rata* to the US\$ 1.5 trillion total (see below for UNEP data extraction methods).

Greenhouse gas contributions

To estimate the environmental costs of greenhouse gases (GHG) emitted by the current plastics economy, we used published estimates of the total GHG emissions from the plastics industry and the estimated social costs of carbon. Zheng and Suh (2019) estimate that the entire lifecycle of conventional plastics generated an estimated 1.665 GtCO₂ of emissions in 2015, net of plastic recycling carbon credits (0.116 GtCO₂ in credits).

Ricke et al. (2018) estimate a global social cost of carbon of US\$ 417 per ton (median, range US\$ 177–805 per tCO₂). By multiplying the emissions by the global social cost of carbon, we estimated the expected economic damage associated with greenhouse gas emissions from the plastics economy to be US\$ 694.3 billion in 2015 (**Supplementary Table 6.2.1**). To allocate greenhouse gas emissions to different commercial sectors, the ratio of sectoral costs in the 'greenhouse gas emissions' category of the UNEP report (2014) was applied *pro rata* to the total social cost of carbon calculated above.

Supplementary Table 6.2. 1 Estimated damage cost of plastic pollution broken down by industry sector (US\$ million per year).

Sector	Total annual loss	Ocean damage	Greenhouse gases	Land pollutants	Water use	Air pollution	Land disamenity
Food	783,162	619,104	157,179	5,304	1,050	303	222
Soft drinks	355,848	270,440	81,845	2,767	548	152	96
Retail	232,780	177,673	52,828	1,769	348	99	63
Non-durable household goods	190,021	121,462	65,649	2,284	424	118	84
Personal products	87,883	68,200	18,869	631	125	35	24
Clothing and accessories	98,981	58,176	39,124	1,314	252	72	43
Toys	80,080	48,546	30,182	1,065	195	56	36
Footwear	79,114	43,829	33,839	1,133	217	64	32
Restaurants	51,769	42,060	9,304	313	62	17	14
Tobacco	27,856	22,209	5,426	171	34	9	7
Athletic goods	26,678	15,330	10,871	377	69	20	11
Medical and pharmaceutical products	16,717	12,775	3,778	127	25	7	5
Consumer electronics	14,131	197	13,323	487	86	24	15
Furniture	69,097	-	66,151	2,345	415	118	68
Automobiles	66,965	-	63,318	2,972	453	128	94
Durable household goods	44,698	-	42,620	1,653	281	83	61
Total	2,225,781	1,500,000	694,305	24,712	4,584	1,305	875

Data Source

Beaumont et al (2019)

Zheng and Suh (2018),
Ricke et al (2019)

UNEP (2014)

UNEP (2014)

UNEP (2014)

UNEP (2014)

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CHAPTER 7: GENERAL DISCUSSION

Our ocean is the most important and indeed the largest habitable part of our planet. It is also under immediate threat which, if not arrested, will cause irrevocable wildlife destruction globally (Pauly 2009, McCauley et al. 2015, Pauly and Zeller 2016a, Payne et al. 2016). Covering over 70% of the world's surface area and 99% of its habitable space, its healthy functioning supplies at least 50% of the world's oxygen with three billion people depending on seafood as their primary source of protein. Marine fisheries directly or indirectly employ over 200 million people and the market value of marine and coastal resources and industries is estimated at \$3 trillion per year or about 5% of global GDP (UNDP 2018).

This Dissertation focuses on solutions to the two immediate and existential threats to our oceans: overexploitation and plastification. These solutions are: (1) large highly protected (IUCN I and II) marine protected areas (MPAs) as a response to overexploitation and the need to build resilience in the face of climate change and ocean plastification; (2) accelerated research to provide scientific evidence to improve ocean decision-making; and (3) market-based interventions for public good that addresses the inability of humanity to place long term self-preservation over short-term self-interest.

7.1 THE DATA CHAPTERS

This Dissertation is comprised of five Chapters for publication (Data Chapters 2-6). I here highlight the key findings from these papers but also, as part of my journey as a late-to-formal-research PhD candidate, the impressions I have gathered on this journey.

7.1.1 THE PERTH CANYON MARINE PARK AS A SHORTFIN MAKO PUPPING GROUND

Chapter 2, Forrest et al. (2019d), identifies a potential shortfin mako *Isurus oxyrinchus* pupping ground at the Perth Canyon, Western Australia. The Perth Canyon lies 129 km west of the State's major capital, Perth, and exceeds the US Grand Canyon in size. Using a global data set based on mid-water stereo-BRUVS surveys at 23 locations and 3,605 deployments, I concluded that this submarine structure may be an important pupping ground for shortfin makos. The study is important for two reasons. First, it provides much-needed information about habitat use and reproductive behaviour of an understudied, IUCN "Endangered" species that is additionally characterised by the double jeopardy of slow reproduction and high commercial value (Collette et al. 2011). Second, this work confirms the relevance of mid-water BRUVS as a non-destructive sampling method that can unveil key conservation-relevant information about cryptic, threatened species – without killing them. Moreover, we can also use mid-water stereo-BRUVS to challenge assumptions that

shortfin makos only use continental shelves as pupping grounds (Bustamante and Bennett 2013) by expanding sampling into a wider range of habitats.

My foray into the ecology of shortfin makos exposed me to the vastness of pelagic ecology and fueled my curiosity about sharks.

“When I came at last to know them ... all these characteristics slipped away, and I saw them as they really are – indolent, awkward, chinless cowards”

- William Beebe, “The Arcturus Adventure” (1926), on the topic of sharks

Understanding that shortfin makos undertake ocean-basin scale movements with females also demonstrating a degree of philopatry (Rogers et al. 2015) underpins the need for protection of key places, what I term “roadhouses”, analogous to those important way stations that underpin long-distance road travel in Australia. I also considered, with some sense of wonder, how sharks can migrate long distances to roadhouses, yet, because there is no parental care, these migrations are not “taught”. Learning about the unconventional approach to reproduction of the shortfin mako (like many other sharks, it is ovoviviparous and oophagous which seems an extraordinary way of perpetuating their individual genes) furthered my understanding of why these top predators are at such risk: the double jeopardy of slow reproduction and high commercial value (Collette et al. 2011). The vastness of the ocean and the nature of species cross-basin transits also exposed me to the value of transnational institutions such as the International Union for the Conservation of Nature (IUCN) and the value of peer-reviewed assessments of global wildlife that serves to drive their protection. These have, for instance, been a core tool for assessing the overall status of sharks (Dulvy et al. 2008).

7.1.2 OFFSHORE ENVIRONMENTS STABLE IN TIME AND SPACE CAN BE PROTECTED

The third Chapter explored the question of whether ocean “roadhouses”, those locations that are in principle critical to species persistence, are stable in space and time. I found that over a period of five years and two seasons (Austral spring and autumn), ocean wildlife used the more landward canyon head in a consistent, predictable way: “hot spots” of wildlife abundance were reliably hot across time; “cold spots” of low abundance were dependably cold. This work is significant in that it addresses two criticisms of pelagic marine parks. First, it reinforces the presence of roadhouses in the ocean, with such areas important for population persistence. Second, it demonstrated that despite the dynamic nature of pelagic environments with respect to currents, fronts and other oceanographic features, population dynamics at some submarine structures are sufficiently stable in *time* as well as space to support the establishment of MPAs with fixed boundaries (Game et al. 2009). This is in

contrast to the arguments for dynamic MPAs (Grantham et al. 2011, Fulton et al. 2015) which are much more difficult to manage. This outcome means that establishing MPAs over such areas is highly likely to deliver conservation outcomes, both locally and globally given the large-scale migrations performed by some pelagic species. Finally, this Chapter also demonstrated the effectiveness of mid-water BRUVS for conducting both longitudinal and baseline surveys.

I would also note that the third Chapter was a data-rich effort that moved beyond conventional univariate and multivariate statistics to incorporate spatial modelling through time, using techniques such as MaxEnt (Elith et al. 2011). The complexity of the methods belies the simplicity of the outcomes, but these outcomes will underpin the decisions governments make in the near future in terms of MPAs as a solution to halt and reverse ongoing declines in ocean wildlife.

This five year journey, commenced before I enrolled in my PhD, was also plagued by the vagaries of government decision-making. The first survey in 2013 (Bouchet & Meeuwig 2015) was at the head of the landward-most canyon. The choice of this location reflected the 2012 intention of the Australian government to place a highly protected, IUCN II zone at this location. Following a change of federal government in 2013, the management plans were suspended and, after a rather cynical review of the “scientific evidence” for MPAs, the protection of the canyon’s most landward head was downgraded, in 2018, to IUCN IV – a partially protected zoning that allows pelagic fishing, whilst protecting the seabed at depths between 500 m and 3000 m. It is worth noting that on-water protection, such as it is, only commenced in late 2018, at which point we had completed five surveys.

Chapters two and three clearly indicate that the Perth Canyon deserves to be highly protected. Clear evidence is available on the failure of partial protection to deliver conservation outcomes as effectively as full protection (Shears et al. 2006, Lester and Halpern 2008b, Costello 2014, Edgar et al. 2014, Sciberras et al. 2015). Partial protection is also more expensive to deliver (Ban et al. 2011, 2014b), an issue that should be a concern to all of us who pay tax. Moreover, there is increasing evidence (Edgar et al. 2014, O’Leary et al. 2018) that, to provide conservation benefits, the Perth Canyon’s IUCN II MPA should run from the canyon head to the limit of Australia’s EEZ to meet the principles of comprehensive and representative (CAR) (Day et al. 2000) and size (Edgar et al. 2014). With the need for protection of Areas Beyond National Jurisdiction (ABNJs) clearly identified (Gjerde and Rulska-Domino 2012), increasing evidence for the value of such protection (Sumaila et al. 2015b), it would also be sensible to expand this protection beyond the boundaries of the canyon system.

7.1.3 UNDERSTANDING PELAGIC BIOGEOGRAPHY

The fourth Chapter documented large-scale patterns in pelagic biogeography in relation to ocean wildlife across nearly 23° of latitude. Much of what we know about ocean wildlife is a function of dead fish (and other animals) on the back deck of a fishing boat; this largely constrains our understanding to commercially targeted species and is often clouded by poor taxonomic resolution, as is the case for sharks (Cashion et al. 2019). Documenting wildlife patterns is critical as it creates a frame against which we can understand change.

Importantly, it also helps us determine where the strongholds of ocean wildlife persist (Letessier et al. 2019). General principles of biogeography indicate that tropical and temperate ecosystems have inverse patterns in terms of attributes such as species diversity (Hillebrand 2004, Tittensor et al. 2010), size (Fisher et al. 2010) and biomass (Lotze et al. 2019). The results from 29 surveys conducted at 15 locations across nearly 23° of latitude stunningly suggested that, despite a wide range of conditions in terms of seabed morphology (depth, slope, shelf position) and environment (sea surface temperature, chlorophyll-*a* and dissolved oxygen), the overwhelming determinant of taxonomic richness, fish abundance and fish biomass was simply the size of the human population in the nearest port. This is dishearteningly consistent with Juhel et al. (2019), Letessier et al. (2019) and Edgar et al. (2014) where remoteness constitutes effective protection rather than active management. This finding is a clarion call for more active management in terms of protection (O'Leary et al. 2011) and, in areas proximate to humans, will provide guidance on *where* to implement this protection.

My personal voyage on this Chapter was twofold. First, the most significant predictor of the status of ocean wildlife is us. That means the best levers to pull around ocean management is managing our behaviour. The failure of conventional fisheries management to halt, let alone reverse, declines in ocean wildlife has necessitated the establishment of large no-take MPAs. Evidence abounds on the benefits of highly protected MPAs and it behooves us to ensure that, as governments make decisions on ocean management, they rely on scientific evidence rather than allowing parties with a fiduciary interest to drive conservation outcomes. Second, this Chapter emphasised to me the sheer size of the datasets needed to shift how we understand and manage our oceans. Field work requires acquisition of a vessel, obtainment of health and safety plans and ethics approvals, and mobilisation and demobilisation of a capable team and needed equipment, typically on a constrained time table. Upon acquisition of the data, significant resources are then required to process the video as current image analysis remain time-consuming and expensive.

The general lack of resources for this data collection is a reality check on just how hard it is currently to sample our oceans to obtain the data needed to drive effective ocean

management. This led me to the depressing realisation that conservation research may not be able to move quickly enough to protect marine ecosystems given the far more rapid advances of some elements within the fishing industry, including the use of forced labour. I concluded that by the time we have generated a baseline survey of ocean wildlife, said wildlife may no longer be abundant. Thus, the measurement of ocean wildlife must leapfrog the subsidised advances of the fishing industry, which is partly fuelled by human rights infringements that equate to slavery (Tickler et al. 2018).

7.1.4 ACCELERATING MARINE RESEARCH FOR FLOURISHING OCEANS

The fifth Chapter (Forrest et al. 2019e) puts forward a realistic and achievable vision for a research vessel that, equipped with a range of existing and emerging technologies and harnessing the power of machine learning and artificial intelligence, will transform how we efficiently generate the information needed for effective and timely ocean management, recognising that this data will need to be transformed to knowledge as the next step. Key to this strategy is scale: the application of existing, proven methods that are combined uniquely to become the solution to a major problem.

This paper was an outcome of the fourth Chapter. The nearly four years of my PhD journey made me aware of how challenging it is to gather research momentum at large scales. Indeed, Chapters 2 to 4 dismayed me to the point of feeling as if there was an insurmountable wall in front of us: that we will never be able to measure wildlife at anything like the rate it is being destroyed. It made me think that the PhD I was doing was simply a vanity - shutting the stable door after the horse had bolted.

However, my research also made me extremely curious about the possibility of combining genomic techniques, such as environmental DNA (eDNA) (Taberlet 2012) with new genomic techniques based on whole cells (eCell), derived from cancer research (Hanash et al. 2002) and BRUVS sampling. Combining such techniques provides multiple windows onto the problem and is more informative and efficient than relying on a single method (Letessier et al. 2016, Stat et al. 2018). I also became interested in the challenge of extending this efficiency from the euphotic zone, where my research has focused, across the entire water column of the oceans at spatial scales relevant to the vastness of this environment.

My business background then led to peer-to-peer discussions with technology providers from diverse backgrounds across marine engineering, oncology and technology, including Google, Woods Hole Institute of Oceanography, CSIRO, Cambridge University, Turing Institute, Deep Mind, London School of Economics, Stanford University, University of California – Santa Barbara, New York University, York University and Harvard MIT. The vastness of an ocean sampling endeavour requires the application of the lightspeed data

interpretation provided by machine learning and artificial intelligence, automating marine life identification and measurement. Further analysis of emerging and frontier technologies indicated that the overnight revolutions in communication and marketing (with inherent invasions of privacy and evisceration of public trust) could also be applied for good purposes, such as the automation of oceanic wildlife measurement. While machine learning and automation are only in the very early days of their application to the marine environment, I note that they are already operating at industrial scale in the commercial sector: the extreme complexity and efficiency of the sourcing and distribution systems of JD.com and Alibaba.com warehouses, retrieval and distribution of goods systems with retrieval of purchases from hundreds of millions of choices instructed by the equally large number of customers to these organisations, during distribution and delivery is entirely driven by AI.

Further building on my business experience, I started with where the world needs to be to return its oceans to their once-flourishing state. Evidence-based recommendations indicate that at least 30% of all major marine habitats throughout the volume of the ocean should be highly protected (O'Leary et al. 2016) and that closing the High Seas to fishing would both increase catches and improve equity. I then reverse-calculated the speed that is required for full ocean measurement of just Australia's EEZ in order to compile the data to present evidence on where marine parks should be placed and what level of protection they require. Recognising that every other coastal nation and the high seas would also require rapid wildlife measurement, the solution was a highly automated vessel that could document the status of ocean life over areas covering a linear distance of not less than 200 nm per day.

7.1.5 MARKET INTERVENTIONS IN A SYSTEMIC GLOBAL PROBLEM

The sixth Chapter reviews the threats of plastic waste to our ocean ecosystems and provides up-to-date estimates of the scale of the problem, which costs us more than US\$2.2 trillion per year. It also presents a global-scale market-based solution to this systemic, pernicious problem. By making waste plastic a cashable commodity, not only will we transform the linear plastics economy into a circular one, we will reduce and ultimately halt the flow of waste plastic into our oceans whilst creating economic opportunity for some of the most vulnerable people in the world currently reliant on picking through waste as a livelihood. Significantly, this approach comes together by recognising that a market-based solution allows existing technology to survive and spread industrially and at commercial scale globally. It does this by sending a price signal that turns plastic waste into a cashable commodity.

The Chapter describes an intervention to allow the environment to recover by making linear fossil fuel-derived plastics more expensive than plastic-derived plastic. This price signal

delivered by the Sea the Future also incentivises the capacity to transform the linear plastic economy to a circular one. This intervention utilises the power of the global market place to apply an ecosystem-wide solution to an ecosystem wide “wicked problem” which also recognises that only the market place is powerful enough to arrest the global challenge of plastic pollution.

From the beginning of my PhD, although I was focused on the issue of overexploitation of ocean wildlife and MPAs, it was nevertheless obvious that plastics were overwhelming marine life. Whilst humans were obliterating top trophic levels through overfishing, we had also found a way to poison the base of the foodweb, the phytoplankton and zooplankton: nanoplastics. I recalled the comment my PhD supervisor made in one of my first ever tutorials, that every second breath we take relies on ocean health, and I started studying the effects of plastics on marine ecosystems furiously. I also engaged in peer-to-peer discussions on the subject of plastic waste with senior executives of fast-moving consumer goods companies such as Coca Cola, Nestle, Unilever, Tesco, Coles, Mars, Walmart, fossil-fuel producers including Chevron, Exxon Mobile, Shell, and resin producing companies such as Ineos, amongst others. I turned to a market-based solution after years of running down rabbit holes and blind gullies that led me to realise that all the ocean clean-up and corporate recycling initiatives had been largely ineffective.

This journey into market-based solutions reflects my broader financial experience. The 2008/2009 Global Financial Crisis (GFC) was, at the time, considered the largest example of the failure of an unfettered, self-regulatory market. In this crisis, the retail banking community bundled property-mortgages across the global developed economy and sold them to large investors like mutual funds and governments. The assumption that self-preservation would prevail over self-interest was the key error of the Bernanke-led United States Federal Reserve who allowed light regulation on the mortgage origination and whole-sale banking community which led, despite several warnings, to the near collapse of the monetary system (Fitwi et al. 2015). However, a larger example of where the market has also been allowed to self-regulate is the vertical industry of the production, fabrication, marketing, retailing and sale of plastics. While the lack of regulation and appropriate market interference led to the GFC, and the threatened destruction of the global economy, a similar event has emerged during the construction of this Dissertation: the destruction of the oceanic environment by plastics, which will have immediate repercussions for humanity.

7.2 SYNERGY AMONGST CHAPTERS

The five data Chapters share the theme of solutions to threats and reflect the tensions between research and political decision-making (Franks et al. 2014). They all reflect the presence of poorly or unregulated markets that have resulted in short-term self-interest taking precedence over long-term self-preservation (Dodd and Favaro 2006). These chapters are also synergistic through their description of local challenges with local solutions that lead directly to global solutions for systemic challenges through cooperation between business and science.

In considering Chapters 2 to 4 of the Dissertation, what became clear was the power of an increasing scale of research through space and time. I started with the study of the ecology and conservation of a single, IUCN “Endangered” species, the shortfin mako and a focus on a single MPA, the Perth Canyon (Forrest et al. 2019d). This work evolved into the consideration of the Perth Canyon MPA in terms of the spatial stability of ocean wildlife over a five year period (Forrest et al. 2019a). Finally, the documentation of the status of ocean wildlife was scaled across over 23° of latitude to understand patterns in the distribution and abundance of ocean wildlife and the human impacts thereon (Forrest et al. 2019c).

Combined, these papers make a strong scientific case, based on empirical data, for the establishment of highly protected marine parks to protect pelagic species. Specifically, in combination, they demonstrate that (1) critical habitat areas such as pupperies can be identified and thus protected, (2) the use by ocean wildlife of key locations, or roadhouses, is sufficiently stable and consistent such that static protection will generate conservation outcomes, and (3) expected biogeographical patterns have been sufficiently disrupted by human presence as to warrant protection that rebuilds this wildlife.

The role of poorly-managed and unregulated markets in driving ocean plastification is clear and laid out in Chapter 6 (Forrest et al. 2019b) where I argue that without proper market intervention into otherwise free markets, those markets will fail the environment, the economy and immediately thereafter humanity. What is worth recognising is that this is symptomatic of a broader problem that plagues the environment. In the case of MPAs, politically orientated decisions lead to the ecological compromise of partial rather than high protection in order to meet the demands of both commercial and recreational fishing. As with plastic, the incorrect application of non-market forces such as the subsidies on fishing fleet construction and fuel (Sumaila et al. 2012) and unpaid labour (Tickler et al. 2018a) leads to overexploitation of the ocean’s wildlife. The paper “Modern Slavery and The Race to Fish” (Tickler et al. 2018a) on which I am a co-author, is included as Appendix 1 in this

Dissertation and indicates that lack of regulation or intervention into the market place monstrously infringe upon human rights.

Further distortions occur as fishers, including recreational participants, serially exploit species both by fishing down the food-web (Pauly and Palomares 2005) and by expanding their activities into offshore areas (Tickler et al. 2018b). Self-regulation and third party certification, such as via the Marine Stewardship Council, conducted within a free-market context, further muddy the waters with respect to the status of ocean wildlife (Froese and Proelss 2012). As a result, decisions such as those that establish partially-protected, rather than highly protected MPAs, facilitated by the lack of scientific data, will continue to fail to achieve adequate conservation outcomes (Di Franco et al. 2009, Kelaher et al. 2014, Sciberras et al. 2015, Giakoumi et al. 2017, Claudet 2018) and, ironically, be more expensive to implement than full-protection (Ban et al. 2011, 2014b). I argue in Chapter 5 that a much greater understanding of the oceanic environment is needed urgently in order to assess the damage of poorly regulated or utterly free market activity and consequently shore up political decision-making for the public good and the environment.

The human character, which appears unable to place self-preservation above short-term profit, is further exacerbated by the uneasy relationship between the business and scientific communities (Parsons et al. 2015). These two communities have traditionally demonstrated competing interests, thereby failing to reach agreement on key environmental interventions such as the establishment of MPAs and halting the flow of plastic waste to the ocean. Science has traditionally accused business of either intemperate self-interest or ignorance whilst business has accused science of ideologically driven conclusions which are most often, particularly in the current era, seen as the politicisation of science. However, without argument from the scientific community, humankind and in particular the business community, has demonstrated a capacity to ignore long-term self-preservation for short-term self-interest, confirming the need for an accelerated delivery of evidence on the state of our oceans.

7.3 NEXT STEPS

The significance of this PhD lies in its linking threats with solutions and linking the role of science with that of the markets. There remains more to do.

7.3.1 FOR MARINE PROTECTED AREAS

The Aichi Protocol and its target to bestow protection in some form, effective or otherwise, on 10% of the oceans (CBD 2010) has been roundly denounced as inadequate and meaningless. The science community (OSCA 2017) argues that 30% of the ocean should be in MPAs (O'Leary et al. 2016), a position supported by international fora such as the World

Congress on Protected Areas (WCC 2016). Civil society is also engaging strongly, with ex-governmental partnerships aiming for 10% of the ocean's volume to be in a network of strongly protected MPAs that allow only artisanal and subsistence fishing, and for these MPAs to be ecologically representative across the EEZs of all coastal nations (JAH Forrest, Fortescue Metal Group, pers. obs.)

7.3.2 TO ACCELERATE OCEAN MEASUREMENT

ROMP, as described, will make nations aware that their EEZs can be measured in an economically sustainable and efficient manner, yielding the knowledge that they need to make better decisions to conserve their ocean assets and meet their responsibilities of sustainability to their citizens and the environment. As this area of technological advancement is accelerating at an exponential rate, the harvesting and combination of this first library of frontier technologies into the one vessel will be very like the first Model T Ford compared to the current Tesla. This is not to underestimate the importance of the first motorised vehicle – or ROMP – but it is to stress that philanthropy often proceeds government action when the risk for government action is too high to not follow a precedent but on demonstration of proof-of-concept, government will follow. I expect that once the scientific knowledge is attained of a nation's EEZ, that this will provide a powerful economic incentive to governments to develop their own or contract with ROMP.

7.3.3 TO DELIVER MARKET INTERVENTIONS

Regardless of the “Sea the Future” initiative being voluntarily adopted by industry or not, the toothpaste will be out of the tube. The common knowledge that polymer-to-polymer permanent recycling is globally scalable at a commercial level, if the price of plastic is raised externally (from the market) in either a voluntary contribution or a tax, will drive governments to act. The dual dividend of popular support for making waste plastic a cashable commodity and driving the wasteful linear plastics industry to a wasteless circular industry, as well as raising substantial taxation revenues, will be too hard to resist.

Industry may initially resist the voluntary contribution and will argue that competition regulations prevent the application of the tax. Government can however clear the ground for the application of the voluntary contribution by introducing legislation to facilitate “price fixing in the common good” that will enable the levels in the supply chain of plastics including energy, manufacturing, distribution, and retailing. By this action, the legislation will formalise and empower the circularity of plastic use.

In the absence of “Sea the Future”, taxation may be applied. However, taxation does not have the benefit of raising capital which can be deployed cross-borders as such foreign investment requires “national interest” test approval. This will hamper and bureaucratise the

decision-making required to remove pollution in the rivers and seas of those countries receiving waste plastic. From a government perspective, a further knock-on effect will be the disaggregated economic empowerment of small investors to enter the polymer production industry, disrupting the current stability between fossil fuel companies and petrochemical industries. While some taxation revenues will be lost from these major companies, from a government revenue perspective, this will be more than compensated by the increase in taxable profits achieved from the emergent circular polymer recycling industry.

7.4 CONCLUSION

The commonality of all of the papers presented in this Dissertation is the curtailment of, and intervention into, the market to encourage anthropogenic self-preservation as opposed to short-term environmental destruction led by purely non-regulated, market-driven decisions that only serve short-term interest. The examples of the GFC, the plastic epidemic, fishing industry subsidies and modern slavery are symptomatic of situations where market levers have been applied incorrectly or not at all. Their power to have large and disastrous results, conversely reflect that properly applied market interventions can also tap the power of the market to create global ecosystem-wide environmental solutions and public good. Whilst the use of market levers has here been applied to waste plastic, it may also have applications to fisheries and climate change.

Since oceans were first exploited with the most basic technologies, the cycle of wildlife extraction leading to local extirpation that incentivises increasingly efficient technology, which, in turn, expands the area of extirpation, has accelerated the unsustainable exploitation of the oceans (Roberts 2007). The primary influence on all oceanic diversity and abundance has been established as anthropogenic, as technology and market motive have concurrently risen. This cycle of influence now threatens the very persistence of ocean wildlife. To avoid this historically-evidenced and scientifically-predictable conclusion of the Anthropocene, measurement and protection of wildlife must be our foremost priority. Turning to the very drivers of wildlife extirpation, technology and markets hold significant promise to both halt and reverse environmental decline, with a return to flourishing oceans.

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APPENDIX 1: MODERN SLAVERY AND THE RACE TO FISH

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ABSTRACT

Marine fisheries are in crisis, requiring twice the fishing effort of the 1950s to catch the same quantity of fish, and with many fleets operating beyond economic or ecological sustainability. A possible consequence of diminishing returns in this race to fish is serious labour abuses, including modern slavery, which exploit vulnerable workers to reduce costs. Here, we use the Global Slavery Index (GSI), a national-level indicator, as a proxy for modern slavery and labour abuses in fisheries. GSI estimates and fisheries governance are correlated at the national level among the major fishing countries. Furthermore, countries having documented labour abuses at sea share key features, including higher levels of subsidised distant-water fishing and poor catch reporting. Further research into modern slavery in the fisheries sector is needed to better understand how the issue relates to overfishing and fisheries policy, and for measures to reduce risk in these labour markets.

INTRODUCTION

Since the mid-1990s, global marine fisheries catches have steadily decreased¹ while fishing effort has continued to increase, leading to intense competition, declining catch-per-unit-of-effort (CPUE) and fisheries profitability, and the over-exploitation of many stocks¹⁻⁴. The consequent race to fish has been exacerbated by harmful government subsidies that enable fishing effort to persist beyond bio-economic limits⁵. The underlying pattern of decline has been masked in the officially reported data by highly unreliable data from some areas of the world^{1,6,7}, and by a presentist bias⁸ that assumes improved catch reporting equals increased catches⁹. The resultant overly-optimistic trend in official data has fostered poor policy decisions, in particular the allocation of government resources to harmful capacity-enhancing subsidies rather than enforcement or stock rebuilding^{1,6,7}. Failure to manage fisheries sustainably has serious implications for human welfare, as fish (here meaning finfish and invertebrates) provide billions of people with protein and vital nutrients¹⁰, as well as employment and livelihoods for hundreds of millions of people¹¹.

Falling productivity and financial returns in commercial fisheries can pressure vessels to cut operating costs, at the extreme by fishing illegally, circumventing licensing costs and catch limits¹², and by reducing expenditure on crew pay, safety and living conditions. Estimates of fishing labour costs suggest they comprise 30-50% of total fishing costs^{4,13}. The large contribution of labour to fishing costs suggests that, in addition to government subsidies received for fuel, vessel operators can capture a significant additional subsidy by aggressively reducing expenditure on crew, for example by non-compliance with labour and safety standards or by withholding pay.

The push to reduce operating expenses to maintain profitability has occurred in the context of rising living standards and employment expectations in industrialized fishing countries, leading to domestic crew shortages and higher wage demands^{14,15}. Concurrently, the political marginalization of coastal, small-scale fisheries throughout the developing world¹⁶, exacerbated by population growth, has contributed to a surplus of domestic and migrant labour in developing countries¹⁷⁻¹⁹. This has polarised labour supply and demand between developed/emerging and developing economies, forcing people in the latter group to engage in any work available, including as fishing crew in an industry highly motivated to cut costs, and which often operates out of reach of enforcement agencies^{14,20}.

Given the nature of working at sea, labour conditions of fishing crews are difficult to monitor. Supported by reefers and supply ships, fishing vessels can remain at sea for months during which time the crew may be unable to disembark²¹ with living and working conditions on such vessels generally beyond the oversight of regulators¹⁵. Given jurisdictional

complexities, it is also often unclear in which country a crew member can seek redress in cases of abuse²². While flag-state responsibility matters, the growing use of flags of convenience further weakens the capacity to enforce regulations^{23,24}. These factors facilitate the use of exploitative employment practices to reduce labour costs at the expense of worker pay, safety and freedom²⁵.

The isolation of workers at sea makes the extent of labour issues in fisheries difficult to quantify. In recent years, however, high profile media investigations have identified a number of cases of extreme labour abuses in fisheries, some involving hundreds of fishing crew. Investigations of the Thai, Taiwanese and South Korean fishing industries identified cases of human trafficking, forced confinement, physical abuse and even murder^{26–30}. These incidents have not been confined just to the high seas or the waters of weaker jurisdictions. Some of the cases involving South Korean vessels took place while under charter in New Zealand waters^{31–33}. There have also been allegations of human trafficking and debt-bondage of African and Asian crew on domestic vessels in British and Irish fisheries^{34–36}, and trafficking and confinement among South East Asian fishers employed in US fisheries in Hawaii³⁷. The US State Department lists 40 countries as source, destination or transit countries for human trafficking in fisheries³⁸, and vessels exploiting fishing crew have been encountered in the waters of Indonesia, Papua New Guinea, Russia and South Africa, as well as New Zealand^{25,39–41}. Labour rights abuses in fisheries appear widespread and serious, in many cases meeting the definition of modern slavery.

Modern slavery is defined by the International Labour Organization (ILO) and the Walk Free Foundation (WFF) as “any situation of exploitation that a person cannot refuse or leave because of threats, violence, coercion, deception, and/or abuse of power”. This includes “forced labour, debt bondage, forced marriage, slavery and slavery-like practices and human trafficking”⁴². As the United Nations Office on Drugs and Crime (UNODC) notes, “the common denominator of these crimes is that they are all forms of exploitation in which one person is under the control of another”⁴³. At present at least 40 million people are estimated to be trapped in modern slavery in textile, agriculture, construction and fisheries sectors, as well as in the sex industry and in forced marriage⁴². Modern slavery exists at the extreme end of a spectrum of exploitative and abusive labour practises, many of which remain legal in the jurisdiction in which they occur and/or are entered into voluntarily by workers^{14,20,25,44}. Commentators rightly argue that a narrow focus on slavery, without broader attention to the needs, ambitions and vulnerability of workers, risks inadequate or even counter-productive responses^{25,44}. However, unlike other labour issues, slavery is universally illegal, with prohibitions enshrined in global agreements including the 1926 Slavery Convention and the

United Nations Convention on the Law of the Sea. It can therefore, in principal, be addressed using existing legal frameworks and instruments, and measures that identify and tackle slavery may have a positive impact on other less explicit forms of labour abuse. Importantly for the purpose of identifying global patterns, the above definition of modern slavery has allowed country-level estimates of the prevalence of modern slavery to be made by the Global Slavery Index (GSI)⁴⁵. While not directly quantifying slavery at sea, the GSI data provide a proxy for analysing the relationship between the prevalence of slavery-like practices in a country and fisheries' characteristics at the global level, which may help identify drivers and policy priorities.

In addition to the structural elements in industrial fisheries that may incentivise and enable modern slavery and labour rights abuses, the global seafood trade is another critical dimension of the issue. Seafood is the world's most widely traded food commodity⁴⁶, involving complex supply chains, with the chain of custody often passing through several intermediaries and countries before reaching the consumer. Traceability issues often arise before the fish even enter the supply chain, with the widely used practice of transshipment at sea allowing catches of multiple fishing vessels to be combined before landing, making the tracing of fish back to individual vessels currently impossible⁴⁷. A lack of consistent, accurate and transparent data from the point of capture to its final destination means that seafood caught illegally or unethically can effectively be laundered by combining it with legally caught fish in subsequent processing steps. The large consumer markets of the global north, including the USA and Europe, import large volumes of seafood to supplement domestic supply. Given that, for example, up to 32% of wild caught fish imported into the US is estimated to have been caught illegally⁴⁸, it seems likely that fish caught under conditions of modern slavery can also enter the domestic supply chains of countries otherwise considered low risk for labour issues in fishing.

Kittinger et al.⁴⁹ called for the research community to more explicitly recognise and address the social dimensions of the ecological crises in the oceans. Modern slavery at sea is such an issue, but there is currently a paucity of quantitative research. The global data on country-level slavery from the GSI⁴⁵ and comprehensive data on fisheries and seafood trade from the *Sea Around Us*^{1,50} and the United Nations' COMTRADE database provide a base for a preliminary investigation. Here, we (1) examine the empirical relationship between the GSI's country-wide prevalence of modern slavery (in all aspects of a country's economy) and fisheries' governance and financial performance; (2) separately identify factors common to those countries with reported labour issues specific to fisheries; and (3) model potential consumer exposure to modern slavery derived seafood products by quantifying the flows of

fish from high (GSI-based) slavery risk environments to relatively lower slavery risk markets.

RESULTS

ANALYSES

Our analyses were performed in three separate stages. The first used linear models to test the overall relationship between the national prevalence of modern slavery, across all aspects of a country's economy, and industrial fisheries attributes among the major fishing countries of the world. Country-level estimates of the overall prevalence of modern slavery (of all types and across all economic aspects of a country) were taken from the GSI⁴⁵, and fisheries catch and economic data were obtained from the 'Sea Around Us'^{1,50}. Here, national level GSI data covering all socio-economic aspects of a country were used as a proxy for likely fisheries-specific estimates of slavery prevalence, which are currently lacking for fisheries at the global level. The second stage used a multivariate clustering approach to identify additional fisheries and economic factors shared by countries with specifically identified slavery issues in fisheries, as reported in the literature and media; this second analysis did not use GSI data. The goal was to develop a qualitative risk model based on the fisheries and socio-economic factors associated with reported incidents of slavery which can frame further research efforts. The third analysis used United Nations' COMTRADE data and the GSI slavery prevalence measure to model the impact of the global trade in seafood on the presence of potentially slave-caught or processed seafood in consumer markets in the United States and Europe, regions where the risk of slave-produced seafood in domestic fisheries is otherwise considered low.

COUNTRY-LEVEL SLAVERY AND FISHERIES METRICS

Linear regression modelling focused on the 20 highest-volume fishing countries, collectively landing over 80% of global industrial fisheries catch. Exploratory analysis found the best explanatory variables to be per cent unreported catch and landed value of catch (Supplementary Table 1). The mapping of unreported catch (Figure 1a) means landed value of the catch (Figure 1b) and the overall prevalence of modern slavery at the country level (Figure 1c) for the world's major fishing countries suggest regional hot-spots of forced labour or modern slavery in Asia, Sub-Saharan Africa, and parts of South America. Generally, these are areas with relatively high levels of unreported catch, predominantly low value fisheries and a relatively high overall prevalence of modern slavery at a national level. The country-wide prevalence of modern slavery in a given country is positively correlated with higher levels of unreported catch ($R^2 = 0.24$, $p = 0.017$, Figure 1d) and negatively correlated with the landed value per tonne of fish being caught ($R^2 = 0.26$, $p = 0.013$, Figure 1e). The multiple linear regression model using both variables explained 46% of the variance in the

overall prevalence of country-wide modern slavery among countries ($p < 0.01$, Figure 1f). Thus, a high level of unreported catch, representing poor management or enforcement oversight of fisheries, and a low unit-value catch, indicating poorer profitability, all other things being equal, correlate with a higher prevalence of modern slavery in the general economy of that country (Figure 1f). While correlation is not causation, these results suggest a link between the presence of slavery and the overall performance of a country's fisheries. The analysis suggests broad underlying trends, yet also identifies outliers whose fisheries performance and country-level modern slavery prevalence do not fit the overall trend. Whilst caution is needed when making inferences about specific economic sector-level labour abuses from the country-level GSI, the present analysis provides a basis for further, detailed sector-specific investigation.

RISK FACTORS ASSOCIATED WITH KNOWN LABOUR ABUSES AT SEA

Having identified in the first analysis a broad correlation between the prevalence of modern slavery at the country level and two key fisheries attributes (unreported catch and mean landed value) for the top 20 fishing countries, we performed a separate principal component analysis (PCA) for the same 20 countries. The PCA grouped countries across six variables describing their economic status and fisheries performance/policy: unreported catch (% Unreported), percentage of catch caught outside their own EEZ (% Catch outside EEZ), per person Gross Domestic Product (GDP per capita; www.imf.org), level of harmful subsidies as a percentage of landed value (% Subsidy), mean landed value per fisher (Value per fisher), and mean distance of catch (Distance). No GSI data were used for this analysis. PCA summarises information contained in a group of n predictor variables as n principal components which capture the main dimensions of variation among the groups being measured, in this case the top 20 fishing countries. The first two components of the PCA explained 74% of the variation between countries. The first principal component axis (PC1) explained 44% of variance between countries and was correlated most strongly with '% Subsidy', '% Catch outside EEZ' and 'Distance'. The second principal component axis (PC2) explained a further 30% of variance, and was correlated positively with '% Unreported', and negatively with 'GDP per capita' and 'Value per fisher (Figure 2). Overall, the individual explanatory variables made similar contributions to the model (Supplementary Figure 1). Clustering countries based on their score (i.e. location) on the first two PCA dimensions divided them into three distinct groups (Figure 2). The first cluster comprised seven countries (red in Figure 2), most of which have been reported for or suspected in serious labour abuses on fishing vessels^{15,32,39,40,51,52}. Countries with documented incidents of serious labour abuses in fisheries are therefore characterised by high levels of unreported catch ('% Unreported'), a high proportion of catch taken outside their own EEZs ('% Catch

outside EEZ') at a greater distance from home waters ('Distance'), and higher than average levels of harmful subsidies ('% Subsidy'). It appears that distance from home waters, non-EEZ fishing and poor fisheries oversight ('% Unreported') may substitute as potential risk factors for modern slavery in fisheries. However, due to a lack of fisheries specific data on modern slavery by country, such conclusions must be drawn with caution and require further investigation.

The second group of countries (orange in Figure 2) included mainly South American and Asian fishing countries with largely domestic fisheries or fisheries that use the waters of immediate neighbours. These countries were characterised by low levels of fishing outside their own or immediate neighbours' EEZs ('% Catch outside EEZ'), low levels of harmful subsidies ('% Subsidy'), but also relatively low GDP per capita ('GDP per capita') and low value fisheries ('Value per fisher'). Future research may show how these countries and these fisheries parameters relate to potential labour abuses or modern slavery in fisheries. The third group (green in Figure 2) consisted of countries generally deemed low slavery risk (the USA and three European fishing countries) that were associated with low levels of unreported catch ('% Unreported'), high GDP per person ('GDP per capita') and high landed value per fisher ('Value per fisher').

GLOBAL TRADE AND SLAVE-PRODUCED SEAFOOD

Finally, we assessed seafood trade data in relation to modern slavery risk to understand the extent to which fish being caught and processed by high slavery-risk countries is potentially consumed in markets which have a low risk of slavery in their own domestic supply chain. Globally, an average of more than 33 million tonnes of seafood were traded annually between 2005 and 2014, based on harmonised UN COMTRADE data (www.cepii.fr). Seafood supply in the top developed countries includes significant proportions of imported wild-caught fish: in the United States, around 45% of domestically consumed seafood is imported wild-caught fish (<http://www.nmfs.noaa.gov>), while in the EU this is 50%⁵³. Total imports are even higher when aquaculture products are considered. Consequently, the seafood available to consumers in these otherwise low slavery-risk countries can end up being a mix of domestic products from local fisheries, predominately in national waters, and products imported from a wide variety of other countries, including from countries with a higher risk of country-wide slavery.

The United States is highly dependent on imported seafood to meet domestic demand, and accounts for roughly 14% of global seafood imports. It has a national slavery prevalence of 1.8 victims per 10,000 persons in the population (0.018%)⁴⁵. Expressed in term of kilograms of potential slavery-risk seafood per tonne, this equates to a slavery risk of 0.2 kg per tonne

of domestically produced seafood, assuming the national prevalence of slavery is applied to all sectors of the seafood industry. Based on the average volumes of seafood imported from other countries, in particular from Asia-Pacific countries, seafood imported into the US has an average potential slavery risk of 3.1 kg per tonne, 17 times higher than the risk of seafood sourced from domestic fisheries (Figure 3a). After accounting for the mix of domestic and imported seafood in US domestic supply, the potential slavery risk of seafood supply within the United States increases 8.5 times due to its dependence on imports (Figure 3a).

Similarly, the low slavery-risk countries of Europe also account for 14% of global seafood imports. Based on the GSI assessment, these countries (i.e., Denmark, France, Germany, Ireland, Netherlands, Norway, Spain, Sweden, and United Kingdom) have an average national slavery prevalence of 2.8 victims per 10,000 persons (0.028%) across their combined populations. Considering the slavery prevalence of the countries from where seafood is imported into this block, the potential slavery risk of imported seafood is 3.8 kg per tonne, 13 times higher than that for their domestically sourced seafood (0.3 kg per tonne). Thus, the mix of imported and domestically sourced seafood increases consumer exposure to potentially slavery-derived products is 8.6 times (Figure 3b), similar to the modelled effect in the United States.

DISCUSSION

Sustainable fisheries underpin both environmental and socio-economic development goals for the oceans⁵⁴, but until recently much of the research has focused on environmental and economic impacts, with less focus on human rights^{4,29}. While links between modern slavery and environmental destruction in illegal mining and deforestation are now well recognised⁵⁵, the connections between environmental challenges and human rights in fisheries have been less systematically documented. However, labour issues in fisheries have received increased attention in recent years^{14,15,25,56}, leading to emerging responses from governments and trading partners (e.g. Thailand-EU), NGOs (e.g. Fair Trade), and major industry-research partnerships such as the Seafood Business for Ocean Stewardship initiative (SeaBOS^{57,58}). An understanding of potential slavery at sea at the global level can place these isolated cases and responses in a broader policy context.

The present analyses have focussed on using comprehensive and publicly-available global datasets to examine empirical links between country-level slavery prevalence and industrial fisheries, and the role of the global trade in seafood in moving seafood products from potentially high slavery-risk producer to low-risk consumer countries. Treating the national,

non-fisheries specific prevalence of modern slavery measured by the GSI⁴⁵ as a proxy for the as-yet unmeasured slavery risk across fishing industry sectors, we found a correlation between the prevalence of modern slavery within a country, and proxies for poor fisheries accountability (i.e., high levels of unreported catch) and low profitability (i.e., low landed value of the catch) in the industrial fisheries of the major fishing countries. It should be emphasised that the GSI is not currently designed to differentiate sector-specific slavery risks, such as for fisheries. Indeed, localised fisheries-specific surveys conducted by NGOs suggest that the national, country-level GSI measure used here may in fact underestimate modern slavery practices in some industrialised fishing fleets. For example, interviews with migrant fishers in Thailand found 17% of respondents had experienced conditions of modern slavery⁵⁹, compared with the GSI's estimate of less than 1% of workers nationally across all sectors. Conversely, for countries where land-based slavery practices dominate (for example mining or agriculture), the GSI's estimate may imply a higher risk for fisheries than may be the case. With this caveat, there remains a broadly linear relationship between national, country-wide levels of slavery prevalence and poor fisheries performance, based on the global data currently available.

To explore risk factors linking the smaller subset of known incidents of slavery at sea, a separate multivariate analysis was then used to identify fisheries and economic attributes shared by those countries with documented fisheries-specific labour abuses. Cluster analysis indicated that countries with documented labour abuses in sections of their fishing industry share several key features: high levels of harmful capacity-enhancing subsidies, likely leading to excess fishing capacity, increased competition, and reduced per-vessel profitability; low catch value per individual fisher, suggesting downward pressure on wages; high levels of undocumented fishing activity, implying poor monitoring and enforcement of vessel operations at sea; and a reliance on fishing far from home in the waters of other countries where regulatory violations may be more likely to go undetected by domestic agencies. Additional evidence of the role of distant-water fisheries in slavery at sea appears in reports detailing specific cases of labour abuse in fisheries, with many victims never even visiting their employer's country (i.e., the vessel's flag- or beneficial ownership state), instead transiting through maritime hubs or countries closer to fishing grounds^{15,51}. The nature of distant-water fishing operations, where transshipment of catch and crew at sea are commonplace, and observer coverage is typically low, appears to facilitate illegal behaviour⁴⁷. The last factor in our multivariate model, GDP per capita, may reflect the importance of economic disparity between labour demand and labour supply countries in driving labour migration, with documented incidents of slavery occurring in countries with relatively high per capita wealth compared to the country of origin of the victims⁵⁹. For

example, Thailand's GDP per capita is over three and four times that of Myanmar and Cambodia respectively, i.e., countries from which it sources the majority of its foreign fishing labour (www.imf.org)⁶⁰. In drawing these conclusions from our analyses, we recognise that fisheries within a single country will differ widely on both social and environmental performance metrics, as the coexistence of Fair Trade certified tuna fisheries (www.FairTradeUSA.org) and fishing slaves trapped on islands in Indonesia²⁶ demonstrates. Nevertheless, while such distinctions must be factored into domestic policy, a model of the common drivers of potential slavery at sea across fishing countries can provide a framework to prioritise research and policy development at the international level. While exploratory in nature, our findings suggest that the well-recognised subsidy-fuelled race to fish, a lack of adequate monitoring, control and surveillance of industrial fishing activities, and the influence of economic disparity on labour markets, has made this sector a fertile ground for modern slavery and other violations^{24,61,62}.

The volume, diversity and global scale of the international trade in seafood⁴⁶ means that seafood produced by countries with poor records in both modern slavery and fisheries governance may find its way into the domestic markets of better regulated countries. Potentially slave-caught or processed seafood can reach consumers directly, as wild-caught product, and indirectly via fishmeal used in livestock and aquaculture feed. Fishmeal supplied by reduction fisheries targeting pelagic fishes, together with millions of tonnes of unmarketable trash fish caught as bycatch, eventually end up on consumer plates as farmed salmon, tuna or prawns, or even pork, chicken, eggs or beef^{63,64}. Many wealthy seafood producing countries, including the United States and European countries, export much of the fish produced by their own fisheries and meet net domestic demand with imports of cheaper seafood products from areas such as Southeast Asia, Africa and Russia^{65,66}. Our analysis of UN trade data suggested that this could result in a greater than eight-fold increase in the exposure of their consumers to potentially slave-caught or produced seafood. To date however, cases linking specific products to labour abuses have been isolated, and further work on traceability as well as fisheries slavery is required to confirm this hypothesis. For comparison, work done to model the flow of illegally caught seafood into the major consumer markets of the US and Japan (together almost 30% of global seafood imports) found that illegally caught products likely constituted 20-32% and 24-36%, respectively, of each country's wild seafood imports^{48,67}. It seems plausible that the current lack of supply chain transparency and product traceability that allows the products of illegal and unreported fishing to enter supply chains also facilitates the international movement of slave-caught and processed seafood.

The issues raised by our modelling of slavery, fisheries, and seafood trade suggest four broad areas of policy engagement: 1) regulation and enforcement, specifically universal minimum standards for crew pay and conditions, such as those specified in the International Labour Organization's Work in Fishing Convention (C-188), and improved monitoring and enforcement of currently weak jurisdictions, including the high seas, to reduce the scope for unsustainable and unethical fishing practices⁶⁸; 2) supply chain transparency, specifically by adopting supply chain legislation, such as the UK's Modern Slavery Act (Modern Slavery Act 2015, s 54), which can bolster industry-led efforts such as SeaBOS to leverage businesses' market position to tackle sustainability and ethical issues⁵⁸. Policing supply chains can be supported by technologies, such as Blockchain ledgers and smart seafood labelling, which improve the security and lower the cost of reliable supply chain data⁶⁹; 3) industry restructuring, specifically by reducing harmful subsidies that currently overcapitalize fishing capacity^{5,70}, and redirecting subsidies towards enforcement and the rebuilding of sustainably managed small-scale fisheries capable of providing more and better livelihoods^{11,71}; and 4) improving equity between stakeholders in fisheries, specifically by restricting high seas fishing, which is currently dominated by higher income countries⁷². Complete closure of the high seas to fishing has been modelled to reduce income inequality among fishing countries by 50%, by ensuring more equitable access to valuable migratory fish stocks⁷³.

These issues have also emerged as key topics in the broader discussions of sustainability in global fisheries as they affect our current ability to effectively manage fisheries for the collective benefit of humanity. This apparent overlap offers an opportunity to leverage regional and international initiatives to benefit both ecological sustainability and social/ethical goals. As research around labour issues in fisheries crystallises, there is great potential for marine scientists and social scientists to collaborate in developing policy frameworks which jointly tackle sustainability and human rights issues. The rapid expansion of industrialised fishing over the last 60+ years has negatively impacted the ability of marine ecosystems to sustainably supply humanity with seafood. The concurrent failure by government decision makers, policy developers and fisheries managers in many regions to adapt to the changes in industrial fisheries has rendered much of the high seas, as well as the waters of developing countries in fisheries-rich areas such as West Africa, open to abuse of both fisheries regulations and international labour standards, allowing illegal fishing and, potentially, labour abuses to flourish^{15,24,73,74}.

Modern slavery and fisheries' performance appear linked at the international level, with a correlation between increased prevalence of country-level modern slavery and higher levels of unreported catches and lower mean value of the catch of industrial fisheries for the 20

countries who supply the bulk of the world's wild-caught seafood. Further research and improved data are urgently needed, as the GSI can presently only report on the risk of slavery at the whole country-level. Given the current lack of reliable data on the prevalence of fishery -specific slavery and labour abuses, the country-level GSI is the most appropriate substitute metric currently available. Based on the limited information available on specific instances of slavery at sea, the over-subsidized and often poorly governed, distant-water fishing fleets of higher income countries may be at particular risk of labour abuses and modern slavery. Our preliminary trade model, using peer-to-peer trade in seafood products, indicates that products of fisheries from slavery-prone regions/countries may be consumed in developed countries in significant quantities, potentially making seafood consumers in developed countries unwitting participants in modern slavery.

Much additional work is required to quantify the prevalence of labour abuses and modern slavery in seafood capture, aquaculture, processing and in the seafood supply chain. Generating comprehensive and accurate estimates of the prevalence of modern slavery in the fishing industry and seafood supply chain will not be easy, as fishing vessels rank among the world's most inaccessible workplaces. However, like the challenge of enforcing environmentally more benign fishing practices, it is an obstacle that must be overcome.

METHODS

DATA SOURCES

Data on global fish catches by fishing country were obtained from the *Sea Around Us* reconstructed global catch database¹. The methods used for catch data reconstructions and the spatial allocation of global catches are well-established⁷⁵ and individual country reconstructions are summarized in Pauly & Zeller⁵⁰, with detailed technical descriptions accessible via www.seaaroundus.org for each country. Using the *Sea Around Us* reconstructed catch data, we calculated the annual mean (+/- SE) reported and unreported industrial landings (in tonnes, excluding discarded catch) for the decade between 2005 and 2014 for the top 20 industrial fishing countries representing 80% of global landings. Thus, here the term catch is used to represent landed catch (i.e., landings) and excludes discarded catch⁷⁶. In line with international data reporting mechanisms, all catches are supposed to be reported by the flag-state of the fishing vessel (i.e., the flag flown by the fishing vessel), and not the country of residence of the beneficial owner. The fishing activity modelled in our analysis is therefore that of the flag-state reporting the catch on behalf of its flagged fleets. Clearly, flag-hopping, i.e., the tendency by some distant-water fleets to regularly and often rapidly re-register to different flags, makes data reporting for distant-water fleets challenging, and better resolution of this issue needs to be a subject of further investigation.

Data on fisheries employment in the industrial sector used here were taken from Teh & Sumaila¹¹, excluding small-scale fisheries. Estimates of fisheries subsidies by category (beneficial, harmful and ambiguous) and type (fuel, vessel buyback, etc.) were obtained from the *Sea Around Us*⁵. Estimates of GDP per capita, in purchasing power parity adjusted US dollars, were obtained from the International Monetary Foundation's IMF DataMapper site (<https://www.imf.org/external/datamapper/PPPPC@WEO/OEMDC/ADVEC/WEOWORLD>). The catch weighted mean distance of fishing activity from home for each fishing country was calculated using the $\frac{1}{2} \times \frac{1}{2}$ degree cell-allocated catch data of the *Sea Around Us*⁷⁵. *Sea Around Us* catch data are spatially allocated by intersecting biological probability distributions for each taxon in the catch data with a global fishing access database detailing in which country's EEZ foreign fleets are permitted or have been observed to fish⁷⁵. Distance from home for each catch cell was calculated as the great-circle distance between the centroid of each catch cell and the closest domestic port of the fishing country, with port locations taken from the World Ports Index. The catch weighted mean distance was the weighted average of all such cell-port distances, weighted by the catch for that country in each spatial cell, using the methodology employed in Tickler et al.⁷⁷.

Data on the scale of modern slavery were taken from the *Global Slavery Index* (GSI) database⁴⁵, which reports estimates of vulnerability to and prevalence of slavery for 167 countries. Modern slavery was defined as "situations of exploitation that a person cannot refuse or leave because of threats, violence, coercion, abuse of power or deception"⁴⁵. Slavery vulnerability scores in the GSI were generated based on a detailed model of country-level measures of governance and civil protections⁴⁵. Prevalence, defined as the percentage of the population trapped in modern slavery, was estimated from data collected on behalf of the Walk Free Foundation as part of the Gallup World Poll (www.gallup.com) through face-to-face interviews with over 42,000 respondents in 25 countries between 2014 and 2016. Estimates for unsurveyed countries were extrapolated from the subset of surveyed countries using a model based on the relationship between prevalence and vulnerability⁴⁵. Slavery prevalence was presented in this study as individuals per 1000 population rather than a percentage for ease of comprehension, and represents country-wide slavery prevalence across all economic sectors and not fishing-sector specific slavery. A detailed description of the methods used for measuring modern slavery is provided in the 2016 *Global Slavery Index*⁴⁵ and the references therein.

Global trade flows for seafood commodities, estimated as imports and exports of individual seafood commodities in tonnes of seafood product (not wet weight) by country, were taken from the BACI harmonised trade database provided by the Centre d'Etudes Prospectives et

d'Informations Internationales (CEPII) in France (www.cepii.fr). The BACI database uses data from the UN's COMTRADE database, processed so as to resolve inconsistencies between commodity-level import and export volumes and values between countries. BACI data categorised by commodity using the 2012 harmonised system (HS) 6-digit codes were used, wherein the group of commodities beginning with 03---- represents seafood products both wild caught and farmed; it was not possible to distinguish between farmed and wild caught products. The BACI estimates of trade flows were averaged for 2011 to 2014.

GLOBAL SLAVERY INDEX AND FISHERIES PERFORMANCE MEASURES

The relationship between country-wide slavery prevalence and candidate fisheries measures (per cent unreported landings, landed value of catch per kg and tonnes landed per fisher) was tested using multiple linear regression, with competing models compared using sample size corrected Aikike's Information Criteria (AIC) scores (AICc). Model data were taken from the top 20 industrial fishing countries, representing 80% of global catch. Given the high prevalence of land-based modern slavery in India⁴⁵, our approach was to treat India as an outlier for the linear regression analysis. This decision was made based on additional information available for India, for which GSI data were collected at the state level, indicating that modern slavery levels in land-locked states heavily influenced the whole-country estimate. The best model, judged by AICc, used per cent unreported landings and landed value of catch per kg as predictor variables (Supplementary Table 1). The relationships between country-wide slavery prevalence and per cent unreported catch, and between country-wide slavery prevalence and the mean landed value of catch were visualised in individual scatterplots. Model fit for the final model was visualised by plotting observed against fitted values.

Sensitivity analysis was performed on the final multiple regression model to test the effect of uncertainty in the slavery estimates on the model outcome. Fisheries parameters were modelled for each country as being normally distributed with the mean and standard deviation calculated from the 2005-2014 *Sea Around Us* data. Country-wide slavery data were modelled as normally distributed with a mean equal to the reported value and standard deviation equal to the 95% confidence interval divided by 1.96. A Monte Carlo simulation of 10,000 model runs of the multiple linear regression model was used to build a distribution of R^2 values based on likely values for model inputs. Histograms of the output for three alternatives were plotted (varying all variables, varying only fisheries variables and varying only slavery variables, Supplementary Figure 2). The median R^2 value for models varying all variables was 0.29, versus 0.46 for the model using mean fisheries values and the GSI reported country-wide slavery values, which is reported in the results.

To visualise global geographic patterns in both country-wide slavery and fisheries performance, fishing countries' mean values for the predictor and response variables used in the final model (per cent unreported catch, landed value of catch per kg, and slavery prevalence at the national level) were mapped. Countries were classified by the three measures, with red representing poor performance (high unreported catch, low mean landed value, high country-wide slavery prevalence) and green the opposite. The classification of prevalence of modern slavery, as reported in the GSI, are country-wide data, and not specific to the fisheries sector.

MODELLING RISK FACTORS ASSOCIATED WITH SLAVERY AT SEA

Principal components analysis (PCA) followed by k-means clustering was performed on the top 20 fishing countries based on six measures hypothesised to predict the occurrence of modern slavery in fisheries: unreported catch ('% Unreported'), mean landed value per fisher ('Value per fisher'), percentage of catch caught outside their own EEZ ('Catch outside EEZ'), GDP per capita (www.imf.org), level of harmful subsidies as a percentage of landed value ('% Subsidy'⁵), and mean distance of catch ('Distance') calculated from cell-level catch data of the 'Sea Around Us'^{1,75}. The objective of the analysis was to identify the shared characteristics of groups of major fishing countries based on their involvement in known cases of modern slavery in fisheries, to explain outliers in the linear model, and to identify other at risk fisheries that were not highlighted by the linear analysis. Scores on the first two principal components of the PCA, capturing the most important components of variation in the predictor dataset, were used to group the countries using a k-means clustering algorithm (i.e. grouping countries into *k* groups based on their similarity across the composite measures). The optimum number of clusters (*k*) for this step was determined analytically using the NbClust() function in R, which finds the number of clusters which minimises the total within-cluster variance (i.e. makes the group members as alike as possible). The first two components of the PCA were visualised as a biplot, with the cluster members colour-coded (red, orange, green) based on their score on the first two PCA components.

SLAVERY AND GLOBAL SEAFOOD TRADE

The impact of imports of seafood into a country or region on the country-wide slavery prevalence (risk) associated with its domestic seafood supply was modelled using commodity-level country-to-country trade flows in the BACI harmonised UN COMTRADE data. The BACI data allow individual commodity flows between countries to be identified, so that flows of seafood carrying different slavery risks, based on country of production, can be precisely estimated. No distinction could be made between seafood caught by a country and

exported, or imported, processed and re-exported, since that level of information is not supplied. However, this was not a significant issue since national cross-sectoral country-wide slavery prevalence was being used to score seafood exported from a country. Therefore it was implicitly assumed that all seafood exported by a given country, whether caught by domestic fleets or processed from imports, carried the same risk of potentially involving slavery. The slavery prevalence of seafood imports into a particular country or group of countries was then calculated as average of the GSI country-wide slavery prevalence scores of the countries supplying that seafood weighted by tonnes of seafood products imported from each country. Although the GSI slavery prevalence is not specific to the capture fisheries sector, traded fisheries products necessarily involve labour across multiple sectors beyond fisheries, and so a cross-sectoral estimate of the prevalence of slavery gives a reasonable estimate of the slavery risk of products originating in or being re-exported from a particular country. Domestic supply in turn was the average of the slavery prevalence of imports and domestic production, weighted by import tonnage and domestic production net of exports. Internal trade within a bloc of importing countries was considered part of domestic supply, rather than exports. Seafood trade and consumption flows were visualised using a Sankey plot (also known as a riverplot) where the width of connections between nodes is proportional to tonnes traded or produced. River plots were produced in this way for the United States (14% of global imports) and the low slavery risk seafood importing countries of Western Europe (Denmark, France, Germany, Ireland, Netherlands, Norway, Spain, Sweden, and United Kingdom; 14% of global imports).

All statistical analyses were performed using the R statistical language and packages in R Studio.

DATA AVAILABILITY

All relevant data are available on request from the authors. All *Sea Around Us* data are freely available via www.seaaroundus.org, and can also be accessed via the R package *seaaroundus* (see <https://github.com/seaaroundus/>). Teh & Sumaila's fisheries employment estimates are available at https://onlinelibrary.wiley.com/action/downloadSupplement?doi=10.1111/j.1467-2979.2011.00450.x&file=faf450_sm_TableS1.doc. Country-level estimates of the prevalence of modern slavery were taken from the *Global Slavery Index* (<https://www.walkfreefoundation.org/>). Global trade flows for seafood commodities are provided by the Centre d'Etudes Prospectives et d'Informations Internationales (CEPII) (<http://www.cepii.fr>). The economic data used can be obtained from the International Monetary Foundation's DataMapper site (<https://www.imf.org/external/datamapper>).

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AUTHOR CONTRIBUTIONS

J.J.M, J.A.F. F.D., D.P., D.T., D.Z. were involved in the conception of the project, and bringing together the Walk Free Foundation and *Sea Around Us* databases. D.T. performed all data analyses and figure preparation, supported by E.G., J.J.L. and B.O. who collated and prepared the data sources and performed supporting analyses. D.T, J.J.M, F.D, and D.Z. drafted the manuscript, with substantial editorial input from K.B., J.A.F., E.G., B.O., D.P. and U.R.S.

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Reprints and permissions information is available at www.nature.com/reprints. Readers are welcome to comment on the online version of the paper. Correspondence and requests for materials should be addressed to D.T. (david.tickler@research.uwa.edu.au) or D.Z. (dirk.zeller@uwa.edu.au).

COMPETING INTERESTS

The Authors declare that they have no competing financial or non-financial interests.

FIGURES

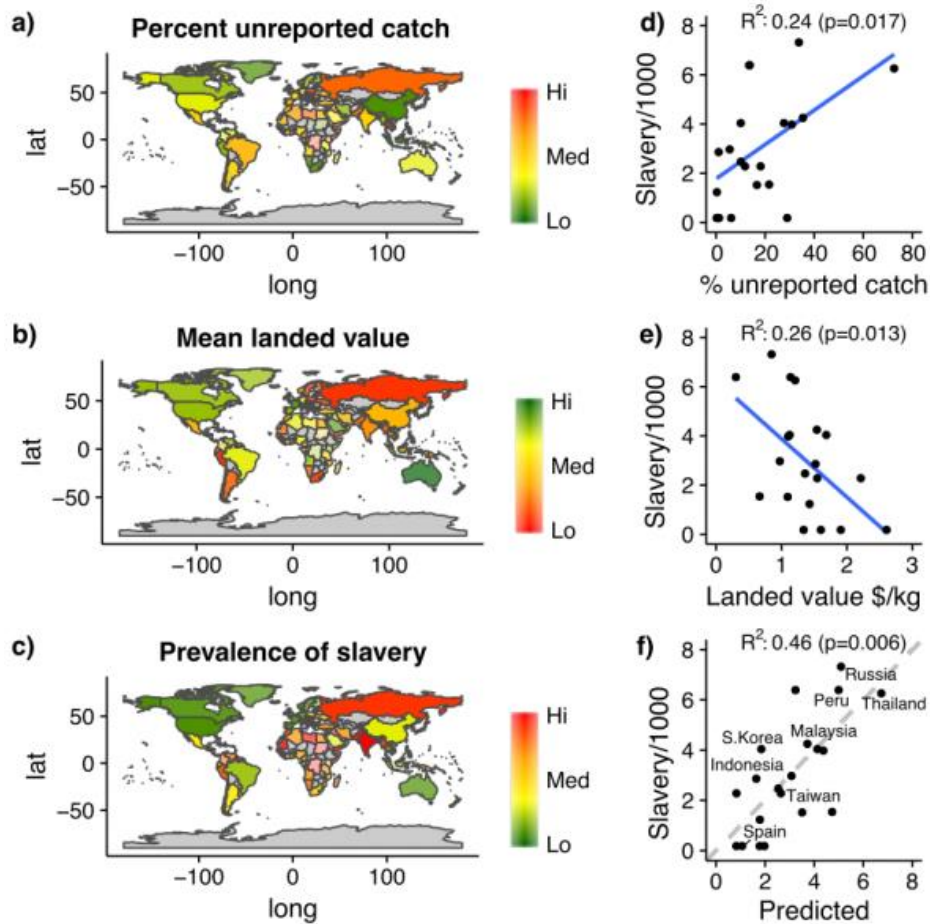


Figure A.1. 1 Global patterns in country-level slavery and fisheries catch and value. Maps show a) per cent unreported catch, b) mean landed catch value per kg, and c) national prevalence of slavery, colour-coded by country. Scatterplots show relationship between and country-level slavery prevalence and individual fisheries variables for the 20 largest fishing countries: d) prevalence of slavery per thousand people (Slavery/1000) vs unreported catch (% Unreported catch), e) prevalence of slavery vs mean landed value (Landed value \$/kg), and f) observed against predicted values for a combined model, with selected European, Asian and South American countries labelled. Regression model R^2 values and F-test p-values are labelled on scatterplots.

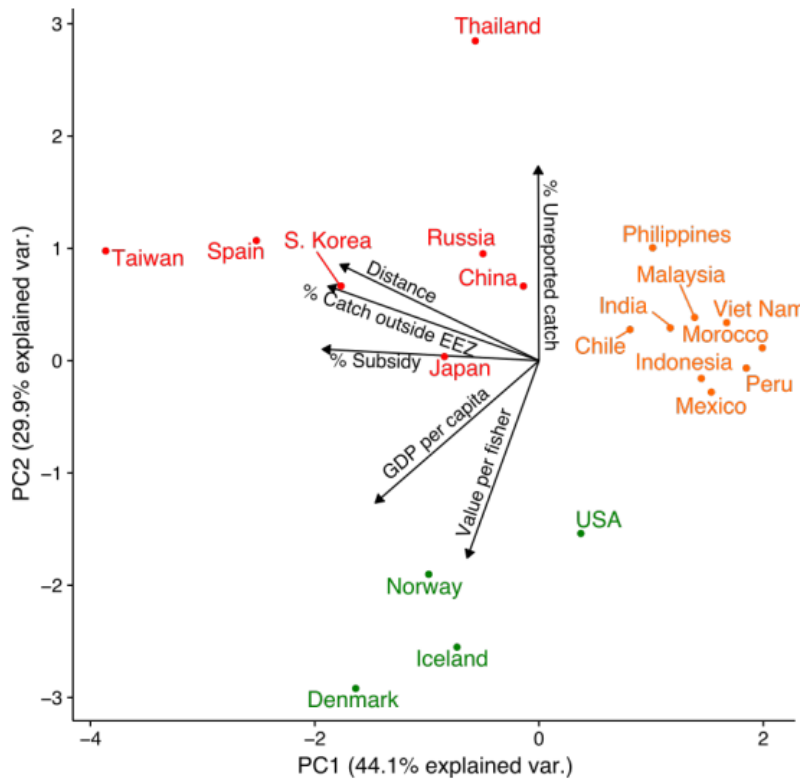


Figure A.1. 2 Biplot of Principal Components Analysis (PCA) for the top 20 industrial fishing countries. Countries are represented based on their aggregate scores across three economic and three fishing activity measures. Arrows indicate direction of increasing value for each variable. Colour-coding indicates cluster membership determined by k-means clustering of countries based on their scores on the main PCA dimensions (PC1 and PC2).

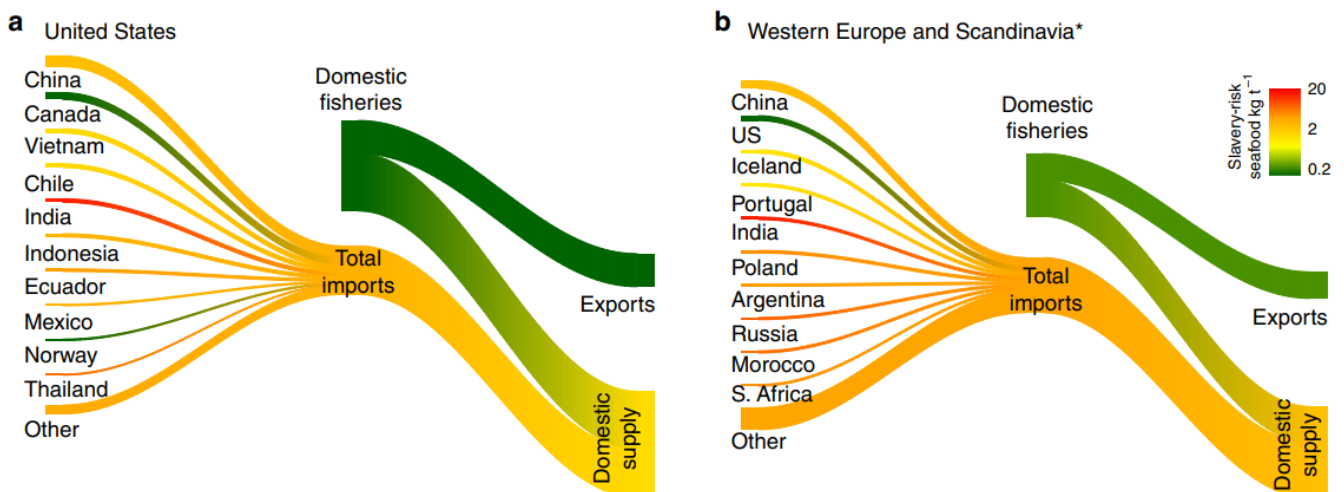


Figure A.1. 3 River plots showing the impact of seafood imports on the modern slavery risk of domestically consumed seafood. Slavery risk is expressed in kilograms of seafood from slavery-risk countries per tonne consumed. Slavery risk scores based on the *Global Slavery Index*; trade flows from CEPII's BACI database of harmonised UN COMTRADE data. Plots show seafood imports for a) the United States and b) Western Europe and Scandinavia (includes Denmark, France, Germany, Ireland, Netherlands, Norway, Spain, Sweden, and United Kingdom). Colour of trade flow components indicates the intensity of slavery risk.

APPENDIX 2: SUMMARY OF EXPEDITIONS

Chapters 2-4 of this Dissertation include data from 29 research expeditions at 15 locations across the Great West Ozzie Transect (GWOT) between 2013 and 2019, amounting to a total of 269 sampling days and 2,756 deployed longlines. The location, year, duration, latitude (LAT) and longitude (LON) in decimal degrees, vessel (POE is the Pangaea Ocean Explorer and represents a transfer of ownership (was Pangaea)), and source of funding for each expedition is summarised in **Table A.2.1**. Chapter 2 also includes data from all mid-water surveys undertaken by the Marine Futures Lab and these surveys are summarised in **Table A.2.2**. A total of 2,699 longlines were deployed at 16 locations across 23 surveys. * indicates the five surveys on which the PhD candidate completed field work.

Table A.2. 1 Summary of research expeditions undertaken on the Great West Ozzie Transect

Location	Year	Start date	End date	LAT	LON	Days	n	Vessel	Funding
Cocos (Keeling) Island	2016	11/10/2016	21/11/2016	-12.12	96.82	12	109	Pangaea	Teach Green
Ashmore Reef	2017	14/07/2017	21/07/2017	-12.21	123.06	8	80	Pangaea	Teach Green
	2018*	2/10/2018	7/10/2018	-12.22	123.05	6	120	POE	Self-funded
Long Reef	2017	30/06/2017	13/07/2017	-13.82	125.74	8	80	Pangaea	Teach Green
	2018*	18/09/2018	23/09/2018	-13.88	125.74	6	120	POE	Self-funded
Argo - Rowley Terrace	2017	16/11/2017	18/11/2017	-15.15	118.5	3	60	Browse Express	Ian Potter Foundation
	2018	4/08/2018	10/08/2018	-15.45	118.53	7	180	Browse Express	Ian Potter Foundation
Rowley Shoals	2017	19/11/2017	22/11/2017	-17.1	119.42	4	55	Browse Express	Ian Potter Foundation
Montebello Islands	2018	15/08/2018	23/08/2018	-20.08	115.36	9	200	Browse Express	Ian Potter Foundation

Location	Year	Start date	End date	LAT	LON	Days	n	Vessel	Funding
Ningaloo	2016	15/09/2016	22/09/2016	-21.84	113.58	8	80	Thalanyji	Self-funded
	2018	24/07/2018	30/07/2018	-21.84	113.88	6	100	NEGU	Self-funded
Shark Bay	2017	15/09/2017	21/09/2017	-26.19	113.05	6	75	NEGU	Self-funded
	2018	6/08/2018	11/08/2018	-26.17	113.07	6	100	NEGU	Self-funded
Perth Canyon	2013	30/03/2013	17/05/2013	-31.95	115.11	5	52	Whale Song	Margaret Middleton Award
	2016*	27/04/2016	4/05/2016	-31.96	115.1	4	50	Thalanyji	Self-funded
	2016	25/11/2016	2/12/2016	-31.95	115.1	4	55	NEGU	Self-funded
	2018	17/04/2018	12/05/2018	-31.98	115.12	6	100	NEGU	Self-funded
	2018	6/12/2018	14/12/2018	-31.99	115.11	3	60	NEGU	Self-funded
	2019	9/04/2019	12/04/2019	-31.96	115.12	4	100	NEGU	Self-funded
Geographe Bay	2017	4/02/2017	8/02/2017	-33.54	115.21	5	50	NEGU	Self-funded
	2018	9/02/2018	13/02/2018	-33.45	115.26	5	100	NEGU	Self-funded
	2019	7/03/2019	10/03/2019	-33.49	115.23	4	100	NEGU	Self-funded
Gracetown	2018	21/05/2018	2/06/2018	-34.02	114.78	4	100	NEGU	Self-funded

Location	Year	Start date	End date	LAT	LON	Days	n	Vessel	Funding
	2019	28/02/2019	6/03/2019	-34.03	114.77	4	100	NEGU	Self-funded
Bremer Canyon	2017*	27/02/2017	8/03/2017	-34.71	119.71	10	100	Big Dreams	NESP
	2019	6/03/2019	10/03/2019	-34.72	119.73	5	100	Big Dreams	Self-funded / Ian Potter Foundation
Recherche Archipelago East	2019*	28/01/2019	2/02/2019	-33.82	124.28	6	110	POE	Self-funded
Recherche Archipelago Central	2019	4/02/2019	9/02/2019	-34.23	123.35	6	110	POE	Self-funded
Recherche Archipelago West	2019	12/02/2019	17/02/2019	-34.29	122.02	6	110	POE	Self-funded

Table A.2. 2 Summary of research expeditions undertaken outside of the Great West Ozzie Transect in collaboration with the Marine Futures Lab

Location	Year	Start date	End date	LAT	LON	Days	n	Vessel	Funding
Ascension Island, UK	2017	16/01/2017	4/06/2017	-8.4971	-13.923	43	410	MV Extractor	Darwin Foundation
	2017	26/05/2017	3/06/2017	-9.6743	-12.718	8	85	RRS James Clark Ross	Pristine Seas, NGS
	2018	19/01/2018	20/02/2018	-7.4622	-14.225	22	160	MV Extractor	Ascension Department of Fisheries
Azores, POR	2018	3/06/2018	22/06/2018	39.0891	-30.182	17	155	Santa Maria Manuela	Pristine Seas, NGS
British Indian Ocean Territory	2012	22/11/2012	7/12/2012	-5.7502	72.1747	11	126	MV Pacific Marlin	Darwin Foundation
	2015	10/01/2015	24/01/2015	-6.3557	72.4434	14	260	MV Pacific Marlin	Bertarelli Foundation
	2016	7/02/2016	21/02/2016	-6.1136	71.9673	10	160	MV Pacific Marlin	Bertarelli Foundation
Clipperton Island, FRA	2016	13/03/2016	22/03/2016	10.2964	-109.21	10	51	MV Argo	Pristine Seas, NGS
Far North Queensland, AUS	2017	7/06/2017	18/06/2017	-10.991	143.458	8	79	Pangaea	Teach Green
	2017	28/11/2017	6/12/2017	-11.498	143.443	9	85	Pangaea	Teach Green
Malpelo, COL	2018	25/04/2018	4/05/2018	4.00653	-81.612	10	45	MV Argo	Pristine Seas, NGS
New Caledonia, FRA	2012	18/10/2012	26/10/2012	-19.113	163.46	9	85	ALIS	IRD / Fondation Total
	2014	1/02/2014	9/10/2014	-20.658	164.854	9	45	ALIS	IRD / Fondation Total
Niue	2016	27/09/2016	8/10/2016	-19.531	-168.83	23	115	Island Passage	Pristine Seas, NGS

Location	Year	Start date	End date	LAT	LON	Days	n	Vessel	Funding
Osa, Costa Rica	2019	11/03/2019	29/03/2019	8.59127	-83.802	12	100	MV Argo	Pristine Seas, NGS
Palau	2014	2/09/2014	22/09/2014	7.42987	134.455	18	169	Ocean Hunter III	Pristine Seas, NGS
Rapa Iti, FRA	2014	18/10/2014	29/10/2014	-27.681	-144.1	19	147	ALIS	Pristine Seas, NGS
Revillagigedo, MEX	2016	30/03/2016	11/04/2016	19.0469	-111.06	11	53	MV Argo	Pristine Seas, NGS
Selvagens, POR	2015	6/09/2015	15/09/2015	30.1149	-15.916	13	75	MY Plan B	Pristine Seas, NGS
Timor Sea, AUS	2012	16/09/2012	2/10/2012	-11.816	127.152	10	57	RV Solander	NESP
Tonga	2013	6/06/2013	15/06/2013	-20.665	-138.02	12	120	ALIS	IRD / Fondation Total
	2013	5/07/2013	14/07/2013	-23.512	-178.51	9	36	ALIS	IRD / Fondation Total
Tristan da Cunha, UK	2017	16/01/2017	2/02/2017	-37.909	-11.843	15	81	SVS Grenville	Pristine Sea, NGS

APPENDIX 3: MAPS SHOWING DISTRIBUTION OF SAMPLING EFFORT BY LOCATION

This appendix presents maps of each of the 15 locations that have been sampled and underpin Chapters 2 to 4. Each map shows the regional position of the survey location. The individual maps show the marine park zoning of the area. Each longline, or string, of 5 rigs is shown with respect to its trajectory and the rectangles around each string represent a buffer of approximately 200 m around the string which was the basis for generating the biophysical data in Chapter 4. Where multiple surveys were undertaken, these are indicated in different colours of the longline trajectories. These maps represent 29 surveys at 15 locations and 2,756 deployed longlines.

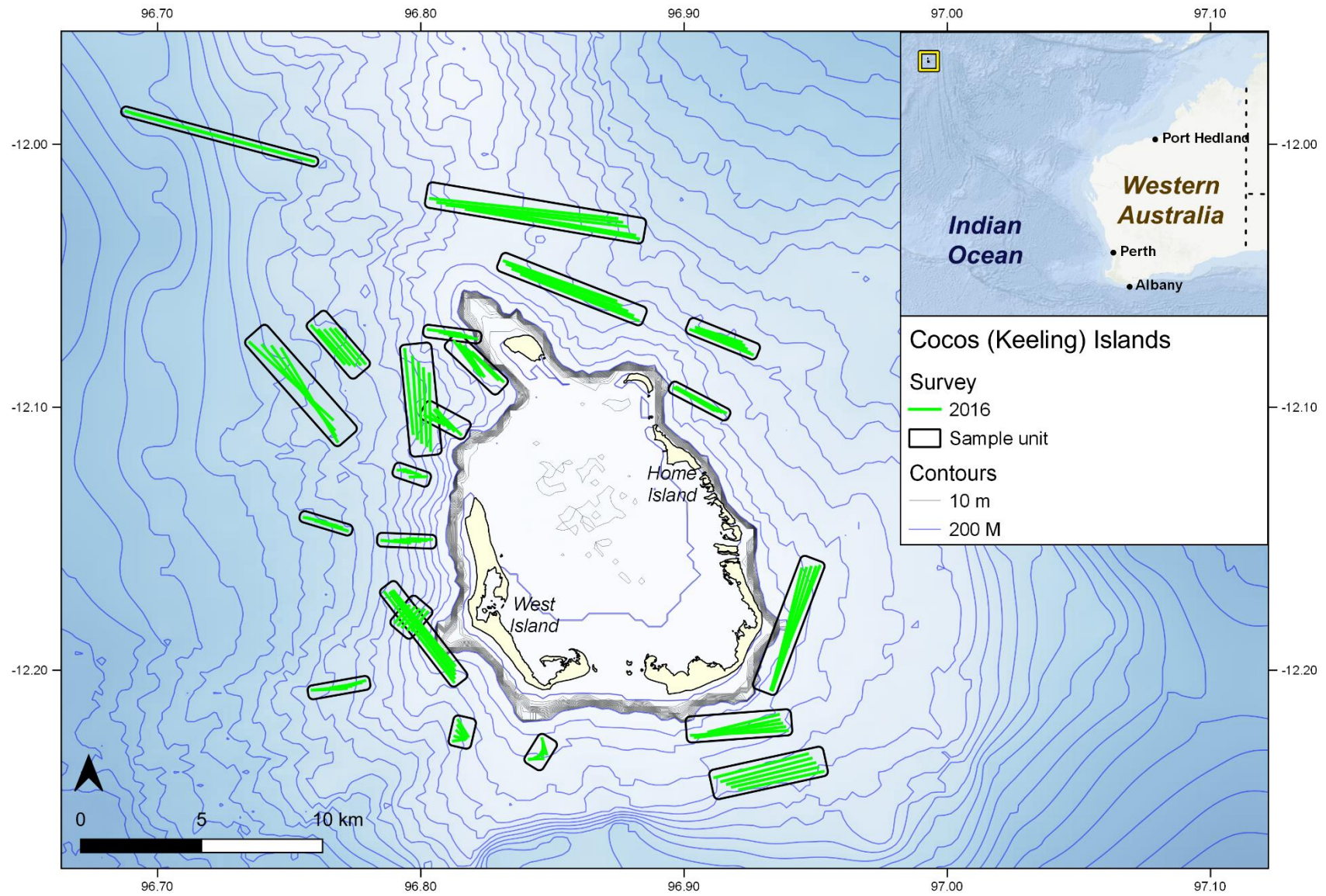


Figure A.3. 1 Cocos (Keeling) Islands expedition map for the survey conducted in 2016.

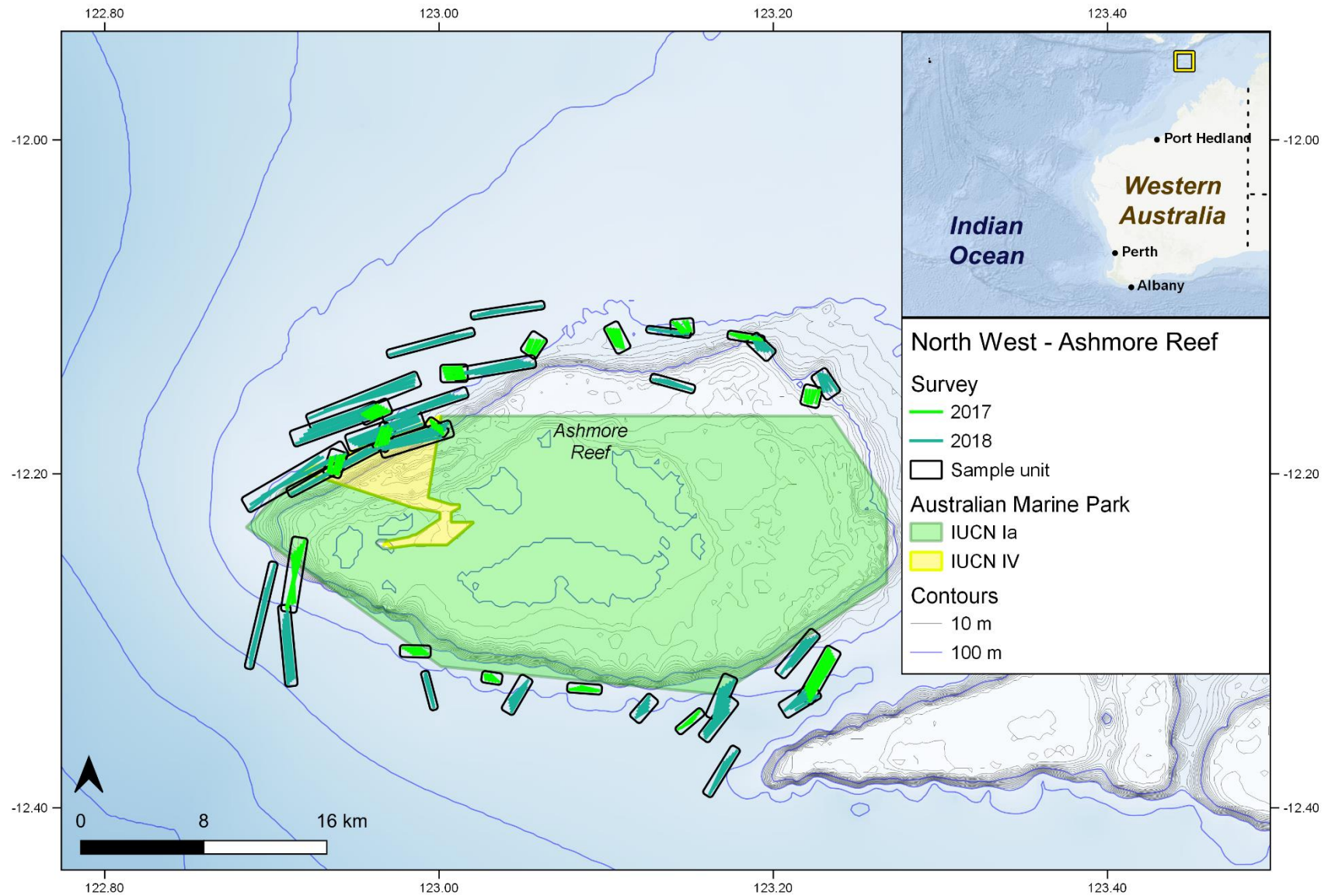


Figure A.3. 2 Ashmore Reef expedition map for the surveys conducted in 2017 and 2018.

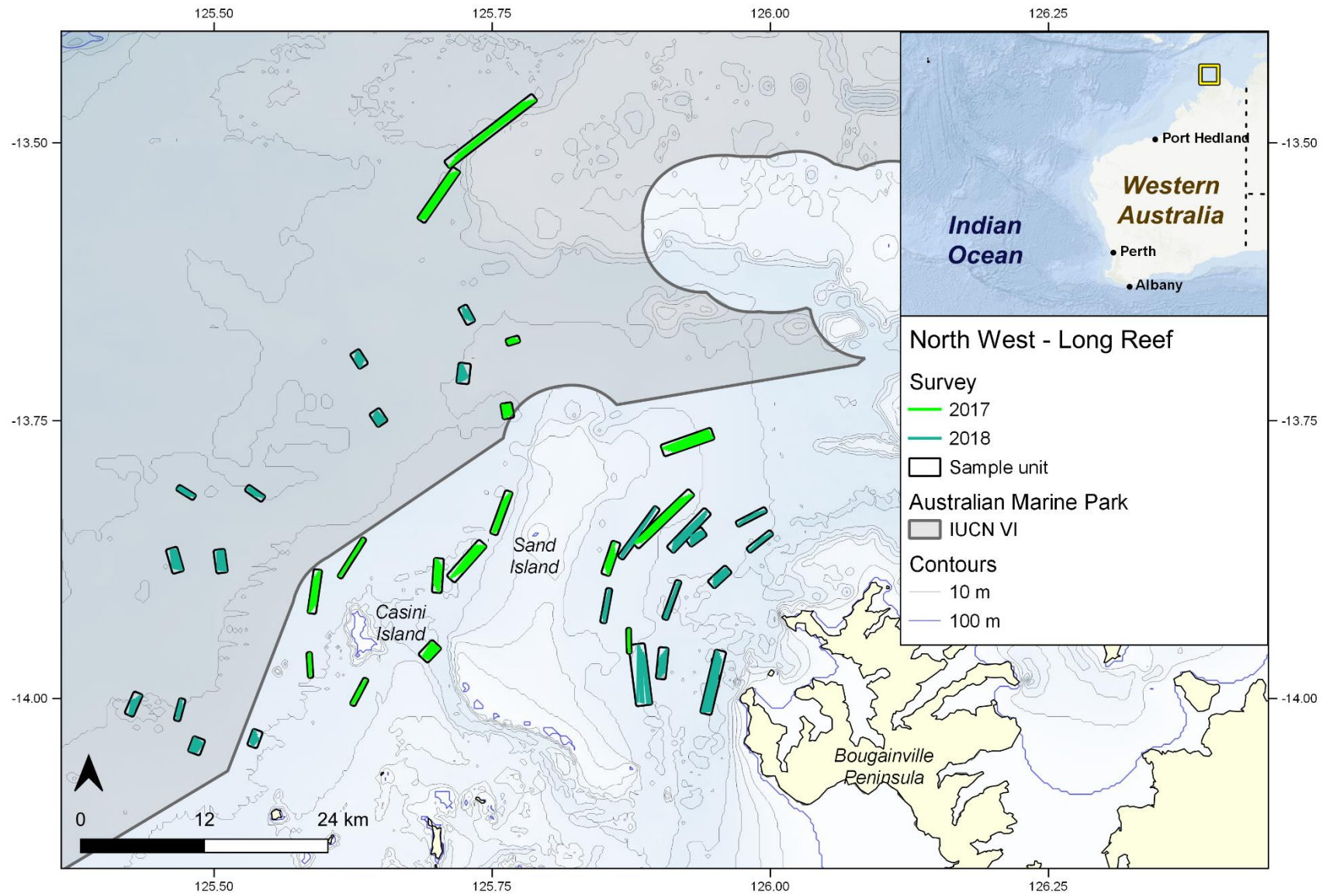


Figure A.3. 3 Long Reef expedition map for the surveys conducted in 2017 and 2018.

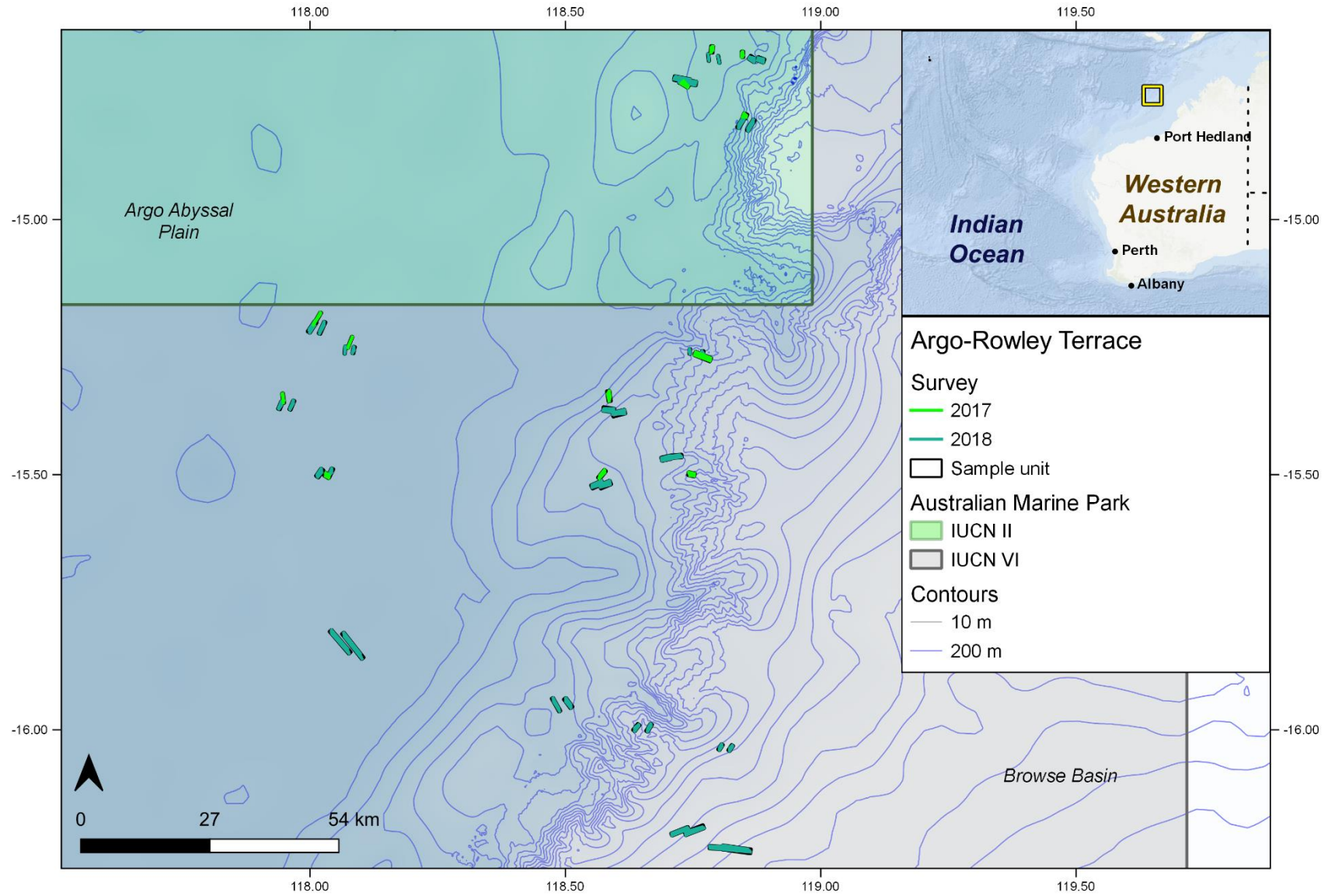


Figure A.3. 4 Argo-Rowley Terrace expedition map for the surveys conducted in 2017 and 2018.

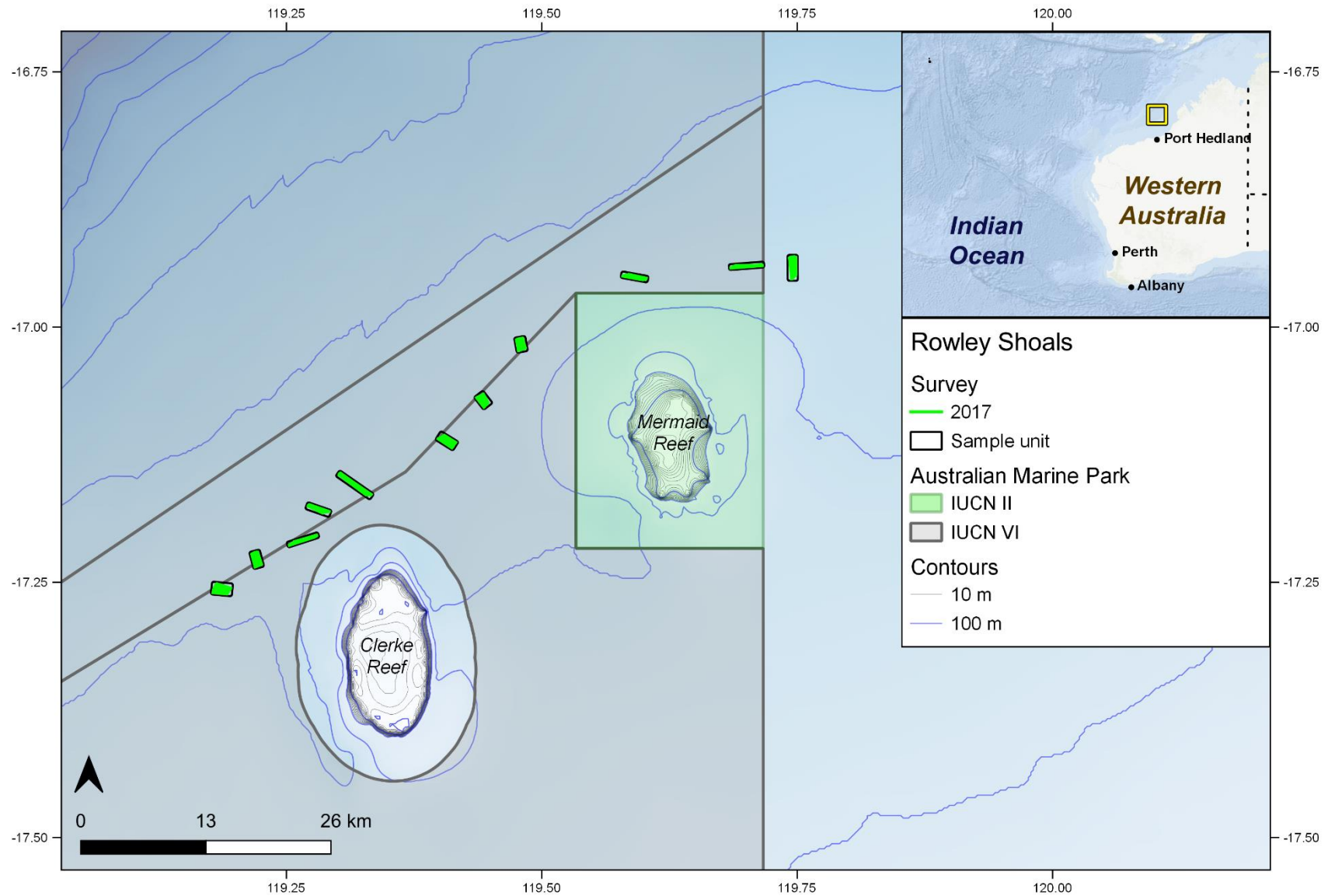


Figure A.3. 5 Rowley Shoals expedition map for the survey conducted in 2017.

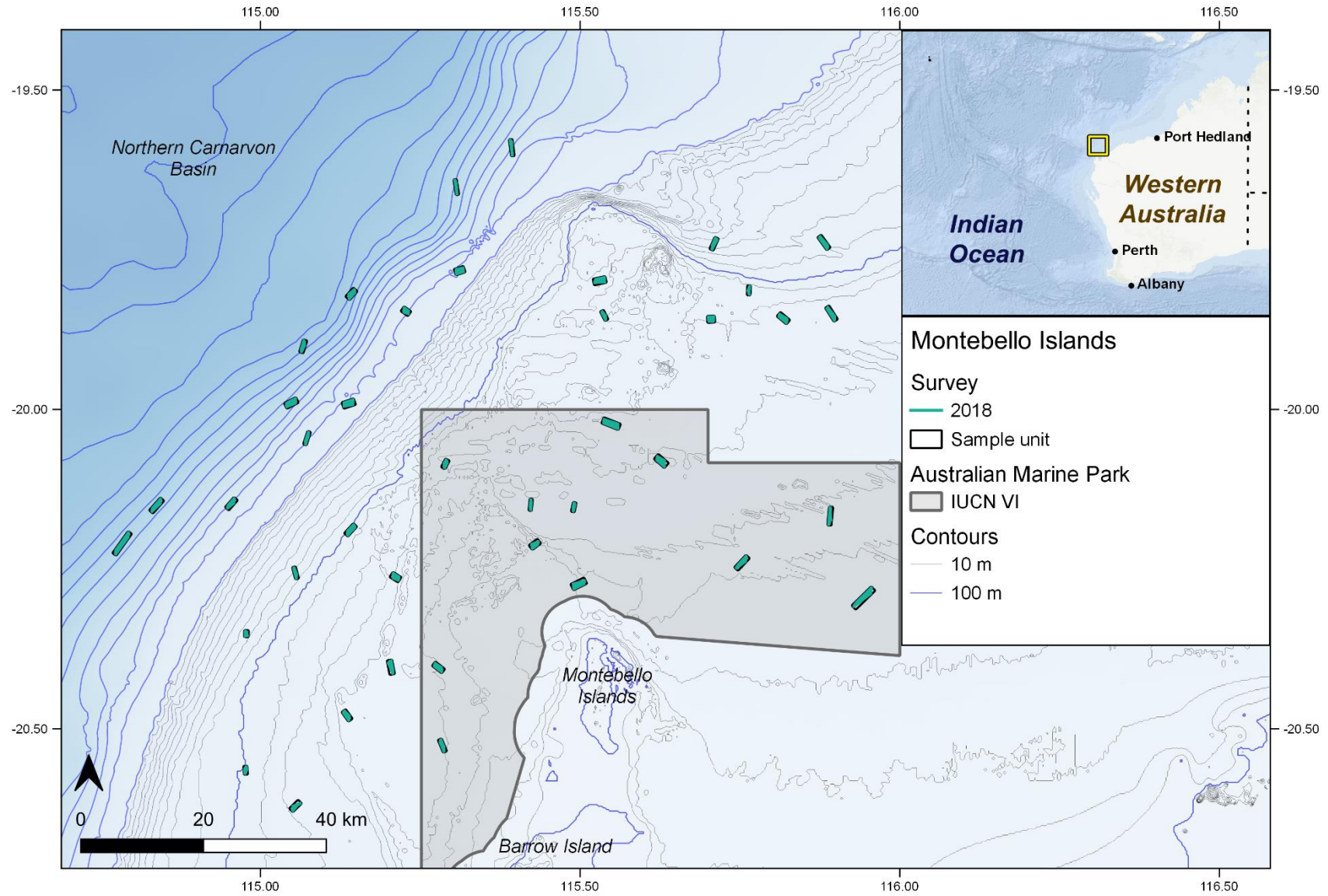


Figure A.3. 6 Montebellos expedition map for the survey conducted in 2018.

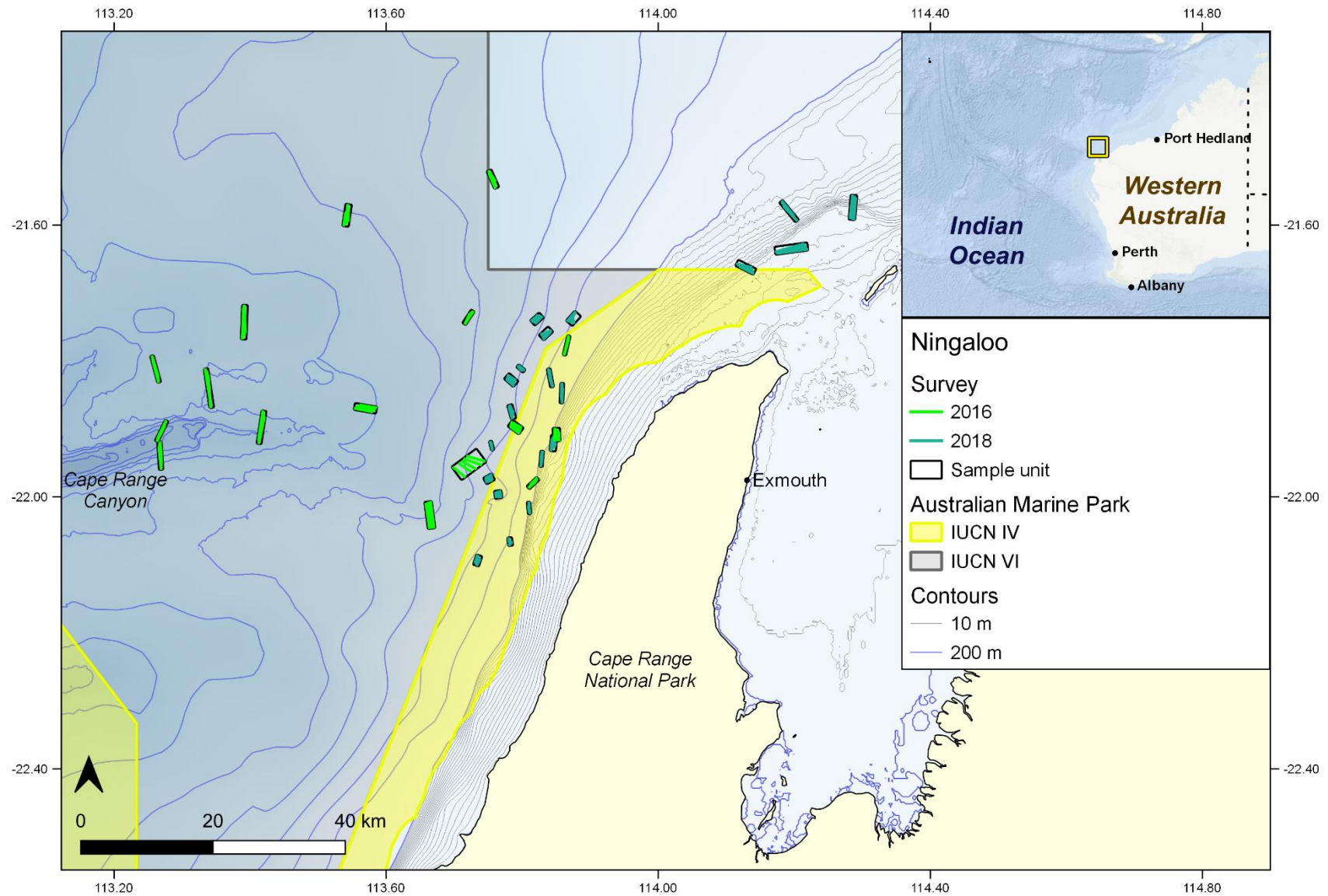


Figure A.3. 7 Ningaloo expedition map for the surveys conducted in 2016 and 2018.

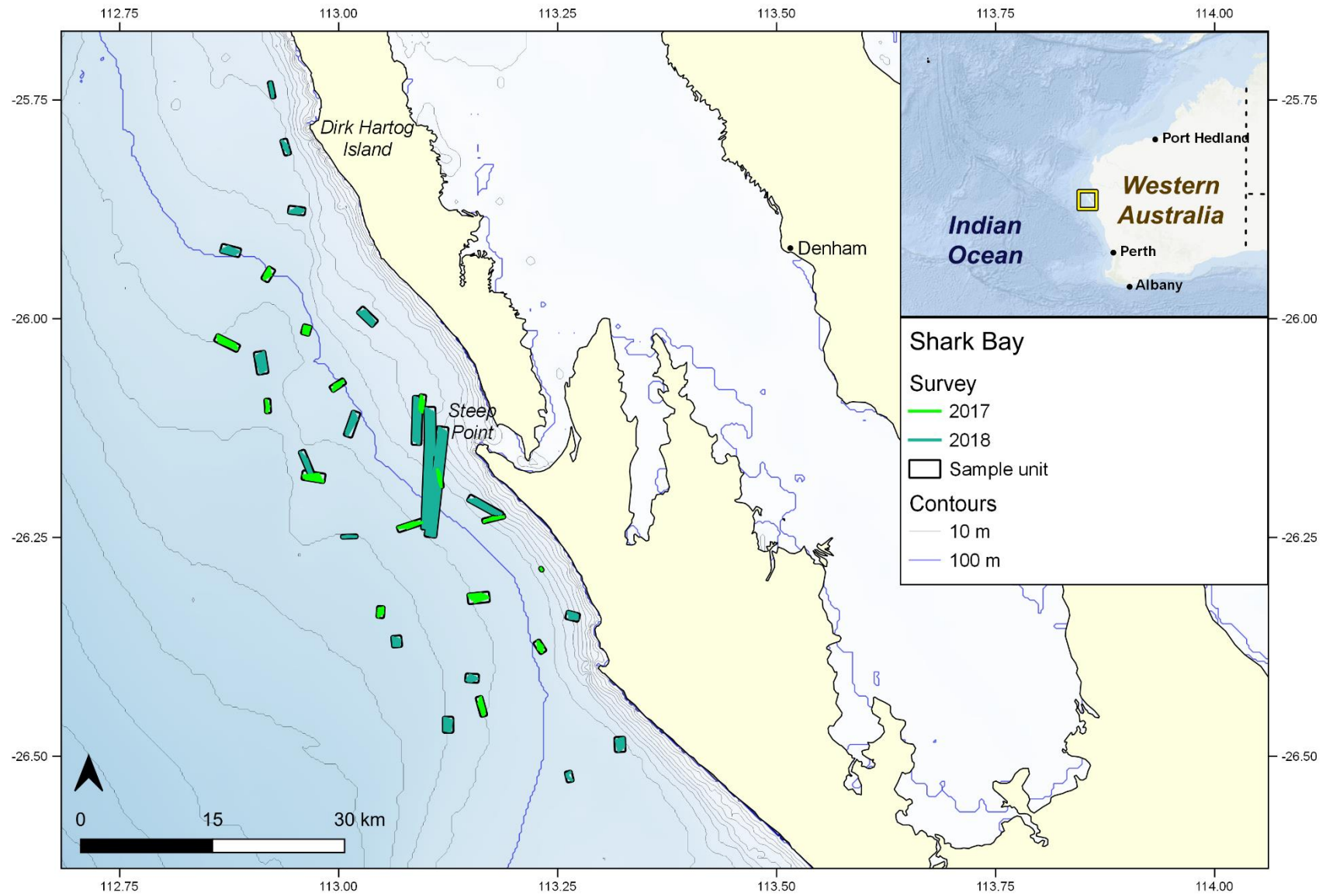


Figure A.3. 8 Shark Bay expedition map for the surveys conducted in 2017 and 2018.

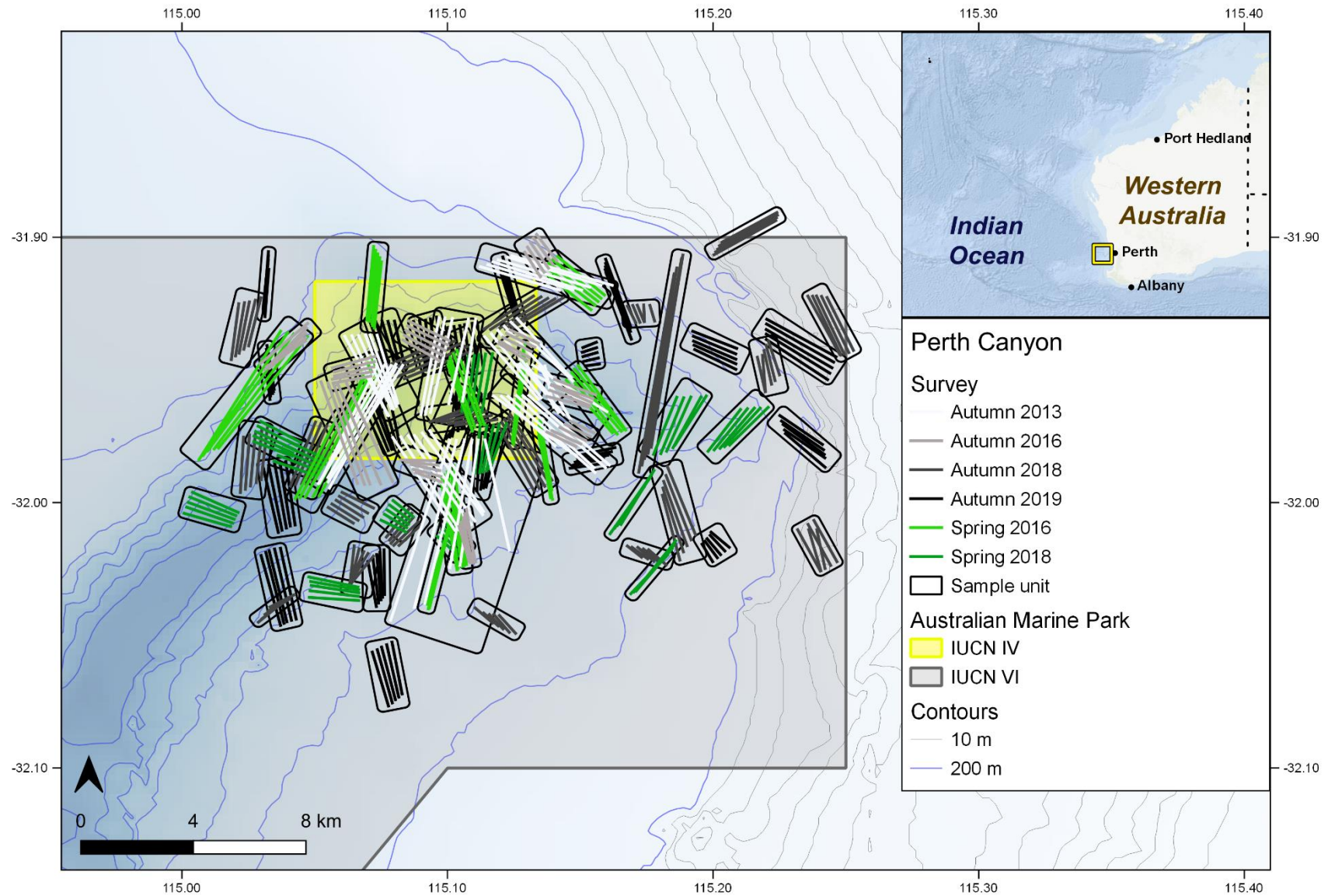


Figure A.3. 9 Perth Canyon expedition map for the surveys conducted in 2013, 2016, 2017, 2018 and 2019.

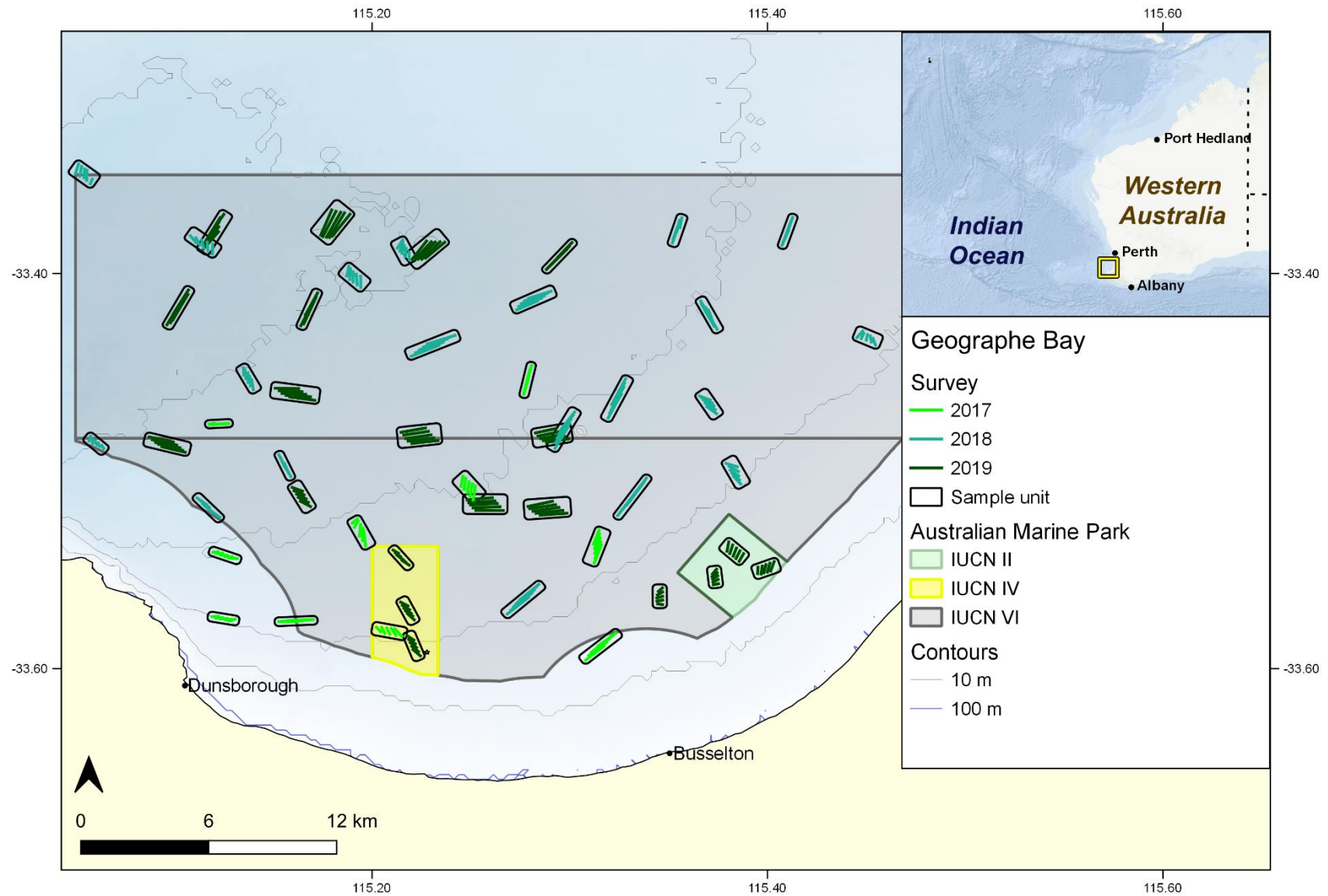


Figure A.3. 10 Geographe Bay Terrace expedition map for the surveys conducted in 2017, 2018 and 2019.

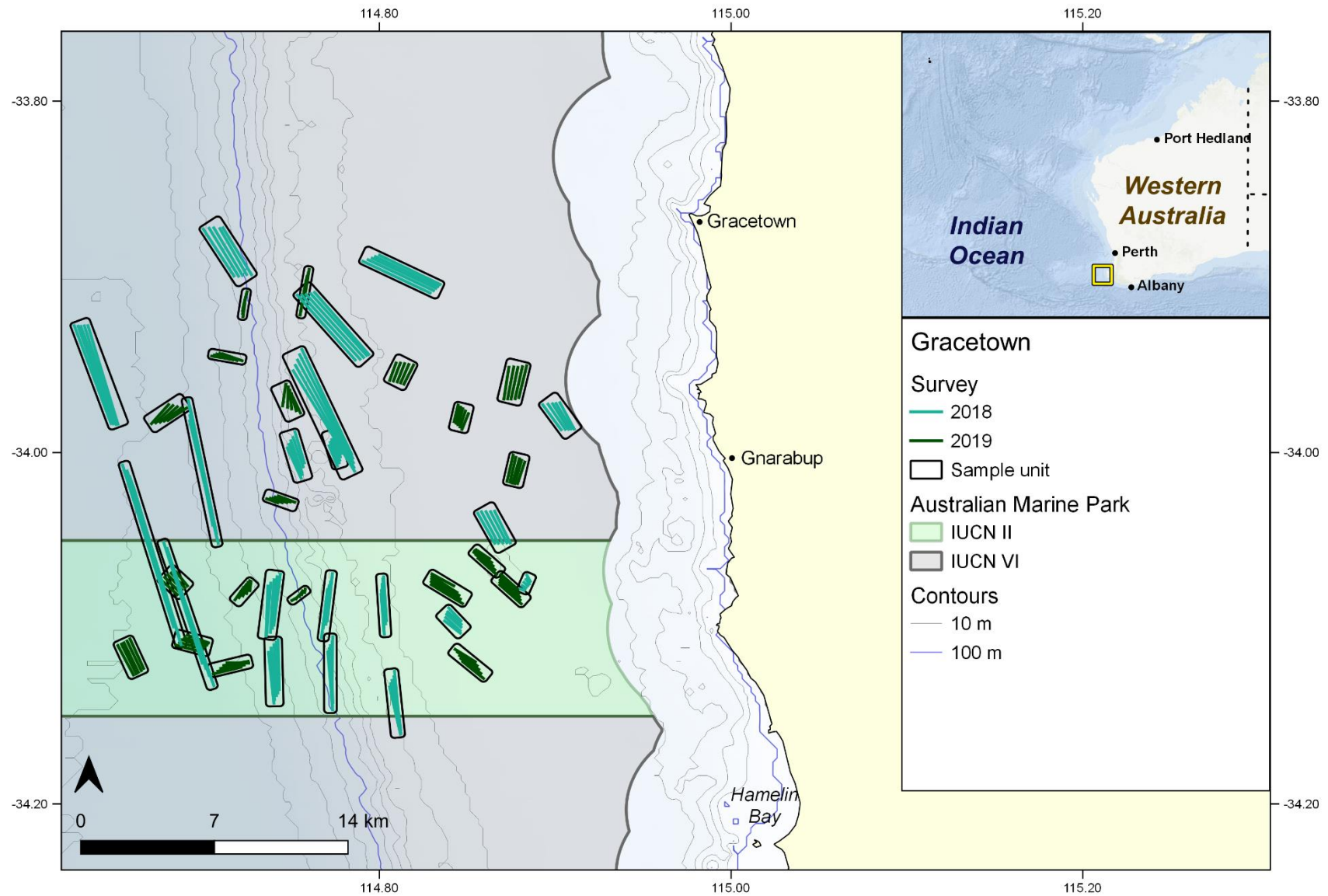


Figure A.3. 11 Gracetown expedition map for the surveys conducted in 2018 and 2019.

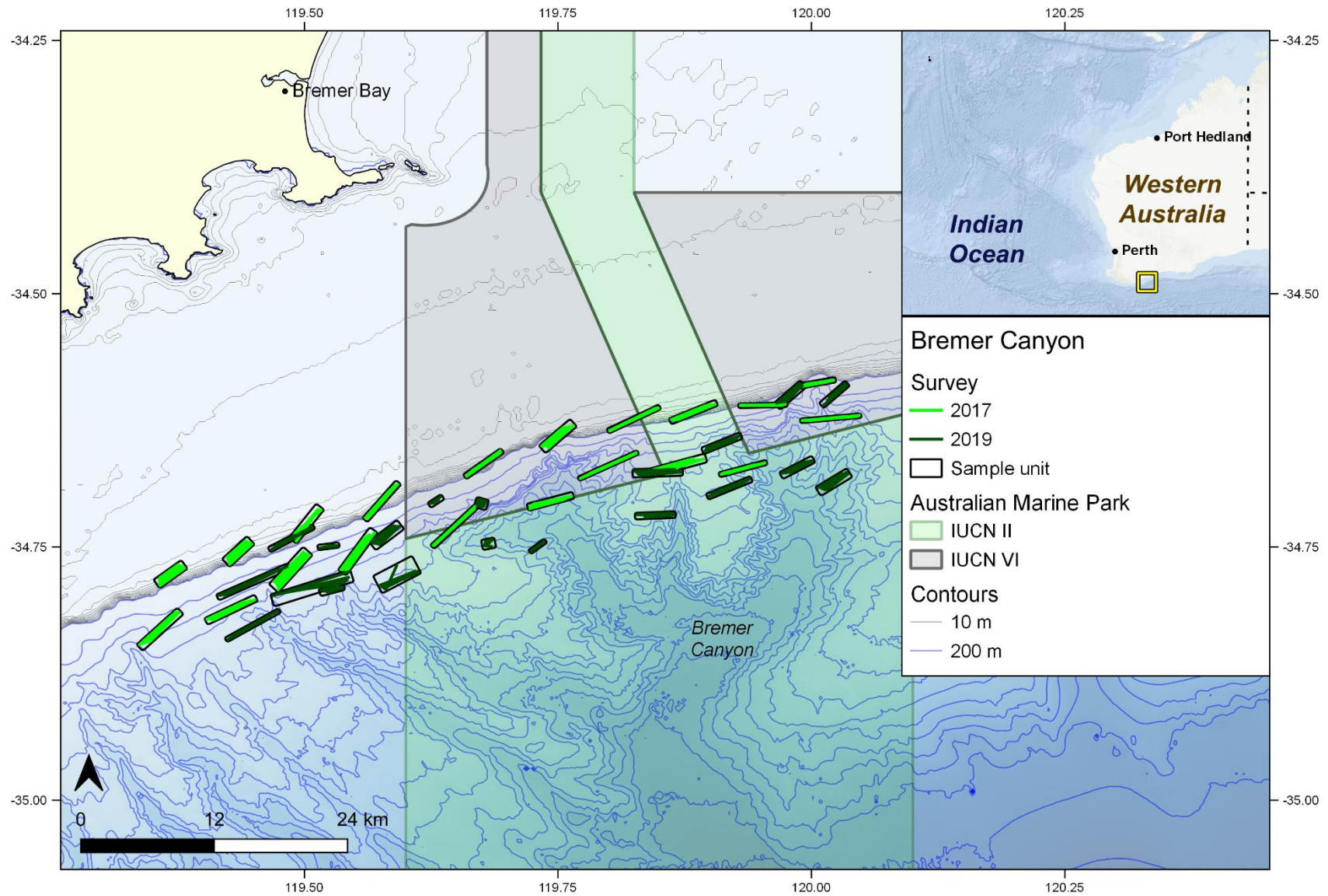


Figure A.3. 12 Bremer expedition map for the surveys conducted in 2017 and 2019.

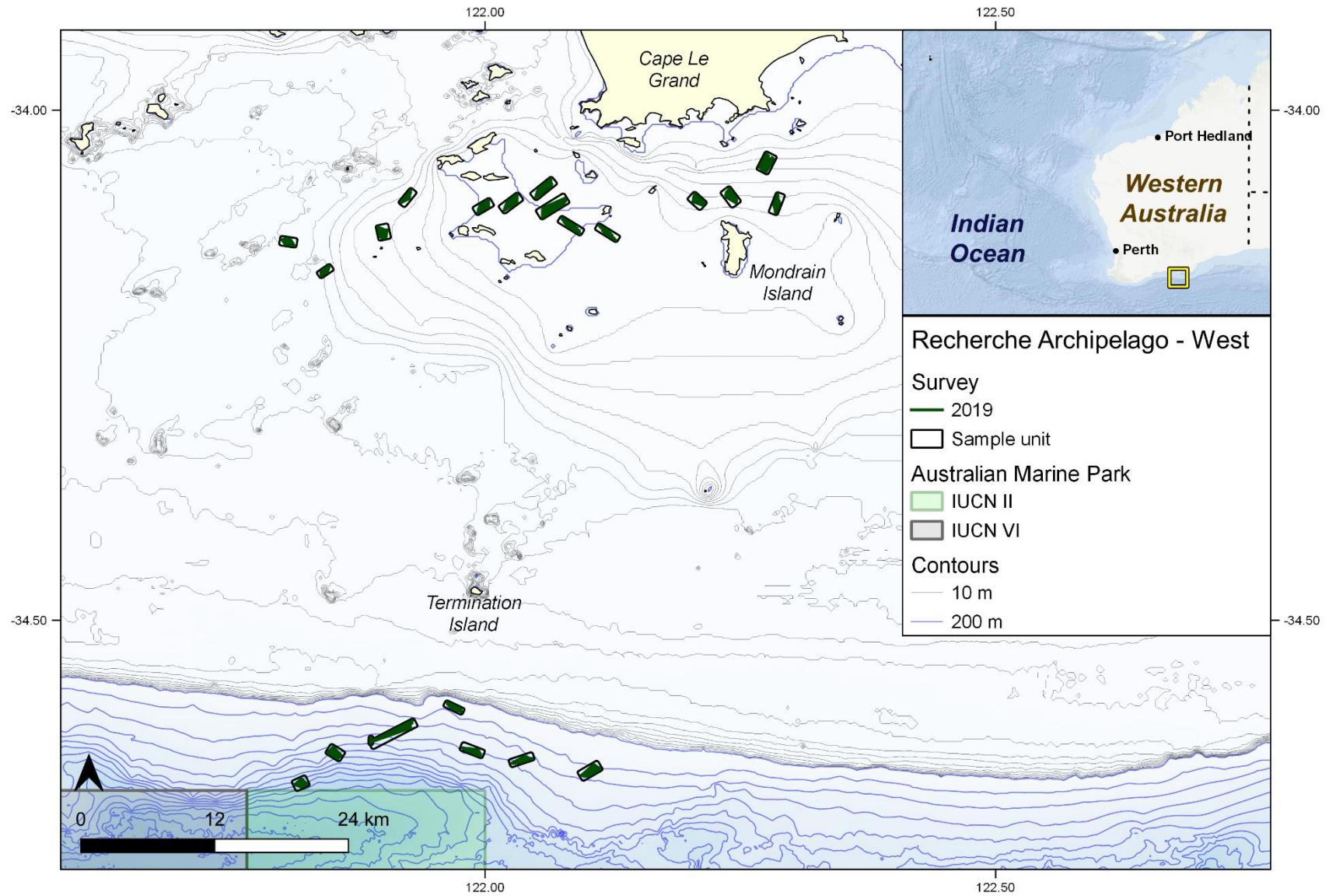


Figure A.3. 13 Western Recherche Archipelago expedition map for the survey conducted in 2019.

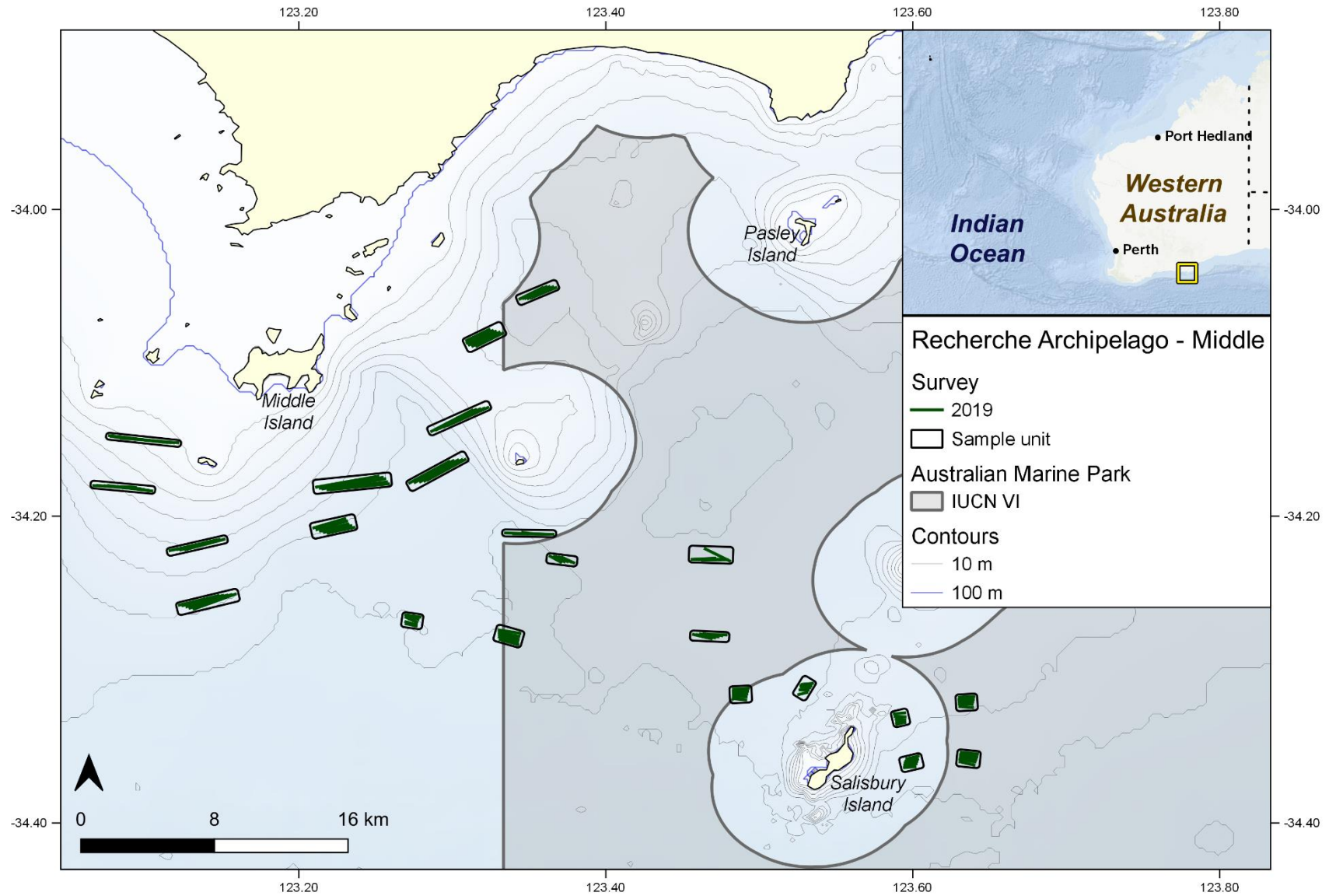


Figure A.3. 14 Middle Recherche Archipelago expedition map for the survey conducted in 2019.

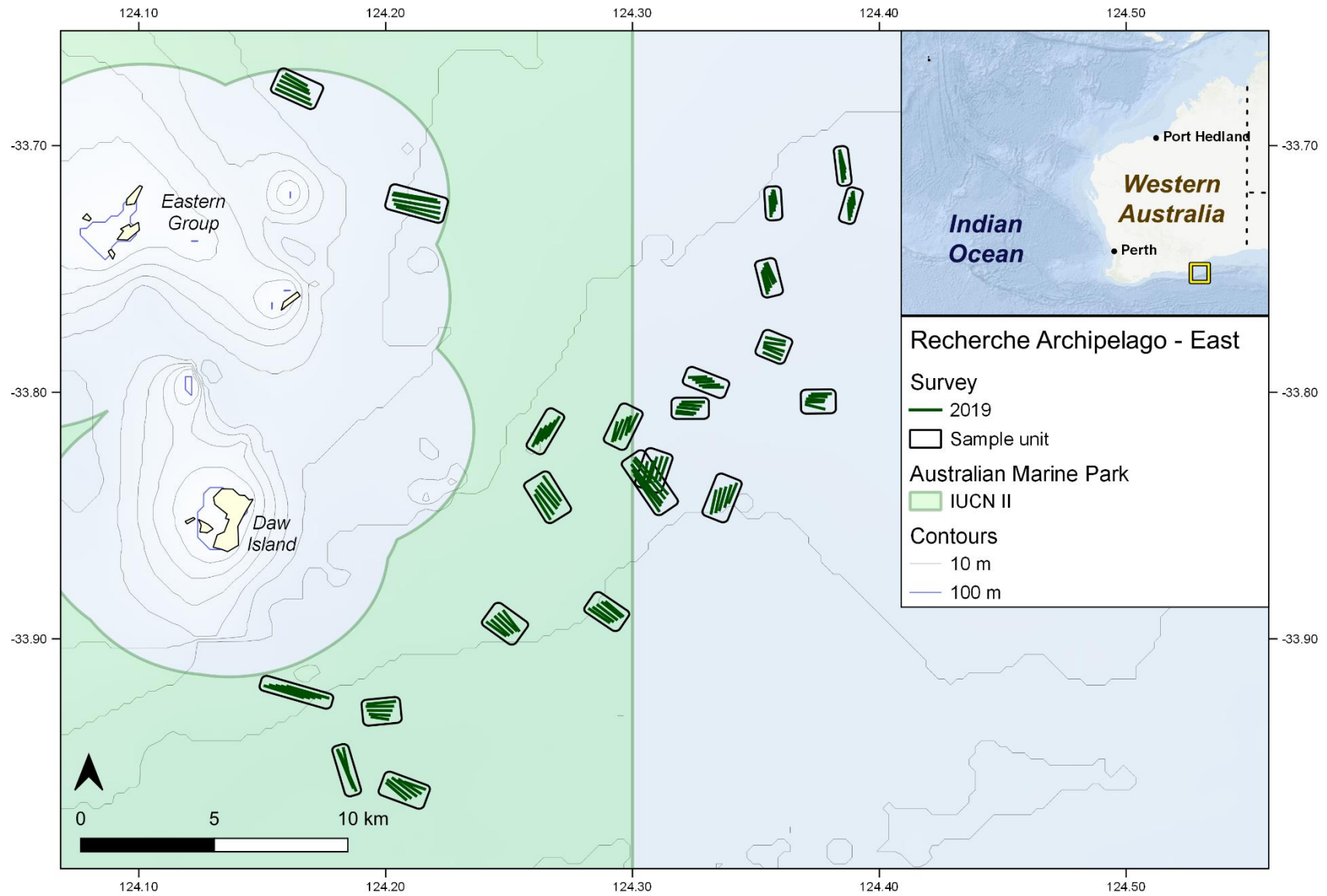


Figure A.3. 15 Eastern Recherche Archipelago expedition map for the survey conducted in 2019.

APPENDIX 4: SPECIES LIFE HISTORIES

To provide more information on the taxa observed in this study, FishBase was used to compile information on trophic level (TR), vulnerability (VUL), and phylogenetic diversity (PD) (Froese and Pauly 2010).

- Trophic level ranges from 2 to 4.5 and represents the position of an individual in the foodweb. The estimates based on food items or diet studies were used where available and in their absence, TR was estimated based on size and trophs of closest relatives as reported on the species page.
- Vulnerability is an index that ranges from 1-100, with larger numbers indicating greater vulnerability. It integrates a number of life history characteristics such that its resilience to declines in abundance can be assessed (Cheung et al. 2005). Our taxa ranged in vulnerability from 10 to 90.
- Phylogenetic diversity or uniqueness is scored from 0.5 to 2 and is an indicator of evolutionary history and thus diversity (Faith et al. 2004). Our taxa ranged in PD from 0.5 to 1.5.
- Additionally, the IUCN Red List of Endangered species was accessed (1 September 2019) to identify the most recent status of species in our study We found that 60.9% were Least Concern (LC), 8.2% were Near Threatened (NT), 13.6% were Vulnerable (VU), 2.7% were Endangered (EN), 0.9% were Critically Endangered (CE), 3.6% were Data Deficient (DD) and 10.0% were Not Evaluated (NE).

Additionally, the IUCN Red List of Endangered species was accessed (1 September 2019) to identify the most recent status of species in this Dissertation. We found that 60.9% were Least Concern (LC), 8.2% were Near Threatened (NT), 13.6% were Vulnerable (VU), 2.7% were Endangered (EN), 0.9% were Critically Endangered (CE), 3.6% were Data Deficient (DD) and 10.0% were Not Evaluated (NE).

Biomass is a key attribute of fish assemblages and fundamental to its calculation are species-specific length-weight relationships (LWR): $W=aL^b$ where W = weight (g) and L = length (cm). The LWR is comprised of two key coefficients: a describes body shape and b quantifies allometric growth in body proportions. These two coefficients vary among taxa and also as a function of the length estimate which can be fork length (FL), total length (TL) or standard length (SL) and in the case of rays, disk width (DW). Once individual weights are determined, they can be multiplied by abundance to determine biomass (B) for the species of interest. The basis on which decisions were made on choice of LWR and the length to be

used are detailed in the methods section of Manuscript 3 and Table A.4.1 here provides the values that were used.

Not all individuals could be identified to species. In these cases, mean values for attributes such as TR, VUL were calculated based on conspecifics recorded from the region (Appendix 5), noting that PD is reported at the genus level. In these cases, either the coefficients for the likely species given the distribution (Appendix 5), a close congener, or failing that, the Bayesian LWR for the genus was used. For those individuals only identified to family (%), mean values for those species within the Family recorded for the region were calculated (Appendix 5) and a Bayesian LWR chosen.

LWR are not included for marine mammals as we excluded them from the biomass calculation due to their rarity and huge size which masked underlying patterns in biomass. Data for other nonfish vertebrates was sourced from SeaLifeBase (Palomares and Pauly 2019), the sister site of FishBase.

Table A.4. 1 Taxonomic information from FishBase (Froese and Pauly 2000) including family, scientific name used in the Dissertation, common name, length-weight coefficients, source of the coefficients with regards to which length measure (fork length (FL), total length (TL) or Bayesian-derived total length (TL_B) methods) the length weight coefficients were based, the taxon's trophic level (TR), vulnerability (VUL), phylogenetic diversity (PD), IUCN Red List status (NE = not evaluated, LC = Least Concern, VUL = Vulnerable, EN = Endangered and FL_B is the mean fork length (cm) for each taxon as observed on the BRUVS footage. '-' indicates that no information was available on FishBase for that parameter or that the FL was not measured on the BRUVS.

Family	Scientific name	Common name	a	b	Source	TL:FL	TR	VUL	PD	IUCN	FL _B
Aracanidae	<i>Anoplocapros amygdaloides</i>	western smooth boxfish	0.01995	3.010	TL _B	1.00	3.4	16	0.63	NE	-
Arripidae	<i>Arripis georgianus</i>	Australian herring	0.00955	3.010	TL _B	0.91	4.3	22	0.63	NE	-
Aulostomidae	<i>Aulostomus chinensis</i>	Chinese trumpetfish	0.00021	3.514	FL	1.00	4.2	34	0.75	LC	15.8
Balaenopteridae	<i>Balaenoptera acutorostrata</i>	northern minke whale	0.0001	2.675	TL	1.00	4.4	63	-	-	513.9
Balaenopteridae	<i>Megaptera novaeangliae</i>	humpback whale	0.0261	3.043	TL	1.00	3.6	56	-	LC	1023
Balaenopteridae	<i>Balaenoptera sp</i>	rorquals	0.0001	2.675	TL	1.00	-	-	-	-	-
Balistidae	<i>Canthidermis maculata</i>	rough triggerfish	0.02570	2.940	TL _B	1.00	3.5	41	0.63	LC	26.9
Belonidae	<i>Ablennes hians</i>	flat needlefish	0.00036	3.322	FL	0.97	4.5	40	1	LC	63.8
Blenniidae	<i>Aspidontus dussumieri</i>	lance blenny	0.00550	3.050	TL _B	1.00	2	23	0.63	LC	3.2
Blenniidae	<i>Aspidontus taeniatus</i>	false cleanerfish	0.00550	3.050	TL _B	1.00	3.8	22	0.63	LC	4.8
Blenniidae	<i>Plagiotremus tapeinosoma</i>	piano fangblenny	0.00566	2.908	FL	0.97	3.8	24	0.5	LC	1.6
Carangidae	<i>Carangoides armatus</i>	longfin trevally	0.01145	3.126	FL	0.82	4.2	35	0.5	LC	2.26
Carangidae	<i>Carangoides sp</i>	fat jacks	0.02940	2.932	FL	-	4.2	34	0.5	-	4.09
Carangidae	<i>Carangidae sp</i>	jacks	0.02570	2.937	FL	-	3.9	38	0.57	-	4.5
Carangidae	<i>Caranx sexfasciatus</i>	bigeye trevally	0.05010	2.710	FL	0.92	4.5	45	0.5	LC	5.43
Carangidae	<i>Seriolina nigrofasciata</i>	blackbanded trevally	0.02580	2.913	FL	0.90	4.2	38	1	LC	8.2
Carangidae	<i>Decapterus sp</i>	scads	0.00890	3.139	FL	-	3.5	27	0.5	-	8.66
Carangidae	<i>Selar boops</i>	oxeye scad	0.01622	3.030	TL _B	0.89	3.5	14	0.75	LC	9.85
Carangidae	<i>Gnathanodon speciosus</i>	golden trevally	0.01992	2.995	FL	0.82	3.8	38	1	LC	10.64
Carangidae	<i>Atule mate</i>	yellowtail scad	0.01660	2.949	FL	0.94	4.2	19	1	LC	11.7
Carangidae	<i>Alepes vari</i>	herring scad	0.01349	2.960	TL _B	0.88	3.7	35	0.53	LC	12.3
Carangidae	<i>Selar crumenophthalmus</i>	bigeye scad	0.00400	3.259	FL	0.89	3.8	39	0.75	LC	14.54
Carangidae	<i>Seriola sp</i>	amberjacks	0.02580	2.913	FL	-	4.3	54	0.5	-	14.8
Carangidae	<i>Trachurus novaezelandiae</i>	yellowtail horse mackerel	0.04780	2.770	FL	0.86	3.2	38	0.5	LC	15.23

Family	Scientific name	Common name	a	b	Source	TL:FL	TR	VUL	PD	IUCN	FL _B
Carangidae	<i>Seriola dumerili</i>	greater amberjack	0.04230	2.747	FL	0.88	4.5	54	0.5	LC	16.71
Carangidae	<i>Seriola rivoliana</i>	longfin yellowtail	0.03590	2.801	FL	0.90	4.5	76	0.5	LC	18.33
Carangidae	<i>Carangoides orthogrammus</i>	island trevally	0.02230	2.980	FL	0.90	4.5	40	0.5	LC	21.45
Carangidae	<i>Decapterus macarellus</i>	mackerel scad	0.00783	3.140	TL	0.91	4	20	0.5	LC	21.72
Carangidae	<i>Scomberoides lysan</i>	doublespotted queenfish	0.04910	2.640	FL	0.83	4	40	0.56	LC	22.13
Carangidae	<i>Naucrates ductor</i>	pilotfish	0.01470	3.040	FL	0.94	3.4	24	1	LC	25.52
Carangidae	<i>Alepes sp</i>	scads	0.01740	3.113	TL	0.89	3.5	25	0.53	-	31.6
Carangidae	<i>Pseudocaranx dentex</i>	white trevally	0.02710	2.886	FL	0.83	3.9	74	0.56	LC	36.54
Carangidae	<i>Seriola lalandi</i>	yellowtail amberjack	0.04320	2.850	FL	0.90	4.2	69	0.5	LC	55.49
Carangidae	<i>Elagatis bipinnulata</i>	rainbow runner	0.01000	2.850	TL _B	0.77	4.3	51	1	LC	64.33
Carangidae	<i>Caranx ignobilis</i>	giant trevally	0.03530	3.050	FL	0.93	4.2	82	0.5	LC	69.61
Carangidae	<i>Seriola hippos</i>	samson fish	0.01370	3.000	TL	0.89	4.6	71	0.5	LC	85.19
Carangidae	<i>Carangoides gymnostethus</i>	bludger	0.04631	2.746	FL	0.91	4.1	47	0.5	LC	94.58
Carangidae	<i>Alepes apercna</i>	smallmouth scad	0.01349	2.960	TL _B	0.88	3.5	24	0.53	LC	-
Carangidae	<i>Carangoides ferdau</i>	blue trevally	0.03160	2.910	FL	0.83	4.3	44	0.5	LC	-
Carangidae	<i>Megalaspis cordyla</i>	torpedo scad	0.03200	2.582	TL	0.90	3.9	29	1	LC	-
Carangidae	<i>Parastromateus niger</i>	black pomfret	0.03220	3.010	FL	0.89	2.9	30	1	LC	-
Carangidae	<i>Pseudocaranx sp</i>	trevallies	0.01413	2.960	TL _B	0.83	-	-	-	-	-
Carcharhinidae	<i>Carcharhinidae sp</i>	requiem sharks	0.00427	3.100	TL _B	-	4.3	62	0.58	-	65.67
Carcharhinidae	<i>Rhizoprionodon acutus</i>	milk shark	0.00790	2.987	TL	0.82	4.3	61	0.51	LC	68.78
Carcharhinidae	<i>Carcharhinus sorrah</i>	spot-tail shark	0.00072	3.656	FL	0.84	4.2	51	0.5	NT	86.81
Carcharhinidae	<i>Carcharhinus limbatus</i>	blacktip shark	0.00614	3.010	TL _B	0.86	4.4	55	0.5	VU	94.96
Carcharhinidae	<i>Carcharhinus amblyrhynchos</i>	blacktail reef shark	0.00227	3.373	FL	0.85	4.1	85	0.5	NT	123.32
Carcharhinidae	<i>Carcharhinus albimarginatus</i>	silvertip shark	0.00427	3.100	TL _B	0.81	4.2	76	0.5	VU	124.14
Carcharhinidae	<i>Carcharhinus plumbeus</i>	sandbar shark	0.01090	3.012	FL	0.82	4.5	88	0.5	VU	140.9
Carcharhinidae	<i>Carcharhinus brevipinna</i>	spinner shark	0.00751	2.970	TL	0.83	4.2	62	0.5	VU	146.95
Carcharhinidae	<i>Prionace glauca</i>	blue shark	0.00318	3.131	FL	0.82	4.4	77	1	NT	154.46
Carcharhinidae	<i>Carcharhinus sp</i>	requiem sharks	0.00427	3.100	TL	-	4.3	63	0.5	-	156.04
Carcharhinidae	<i>Carcharhinus amboinensis</i>	pigeeye shark	0.00479	3.100	TL _B	0.84	4.3	74	0.5	NT	181.22

Family	Scientific name	Common name	a	b	Source	TL:FL	TR	VUL	PD	IUCN	FL _B
Carcharhinidae	<i>Carcharhinus falciformis</i>	silky shark	0.01540	2.922	FL	0.82	4.5	79	0.5	VU	182.55
Carcharhinidae	<i>Carcharhinus altimus</i>	bignose shark	0.01020	3.461	FL	0.83	4.5	76	0.5	DD	185.15
Carcharhinidae	<i>Carcharhinus brachyurus</i>	copper shark	0.00468	3.900	TL _B	0.82	4.5	87	0.5	VU	191.59
Carcharhinidae	<i>Carcharhinus longimanus</i>	oceanic whitetip shark	0.01700	2.980	FL	0.82	4.2	79	0.5	VU	194.55
Carcharhinidae	<i>Negaprion acutidens</i>	sicklefin lemonshark	0.00841	3.000	TL	0.89	4.1	81	0.75	EN	209.63
Carcharhinidae	<i>Carcharhinus obscurus</i>	dusky shark	0.03240	2.786	FL	0.84	4.3	88	0.5	VU	212.58
Carcharhinidae	<i>Galeocerdo cuvier</i>	tiger shark	0.00253	3.260	FL	0.88	4.5	64	1	NT	251.65
Centrolophidae	<i>Centrolophus niger</i>	black ruff	0.00240	3.346	TL	0.93	3.9	85	1	LC	19.53
Cheilodactylidae	<i>Dactylophora nigricans</i>	dusky morwong	0.00389	3.120	TL _B	0.92	2.9	73	1	NE	-
Cheloniidae	<i>Cheloniidae sp</i>	turtles	0.01700	3.000	CL	1.00	3	0	-	-	96.58
Cheloniidae	<i>Chelonia mydas</i>	green sea turtle	0.16740	2.908	CL	1.00	3	79	-	LC	-
Clupeidae	<i>Sardinella sp</i>	sardines	0.01290	2.870	TL	0.87	2.8	16	0.5	-	5.14
Clupeidae	<i>Clupeidae sp</i>	herrings	0.01820	3.131	FL	1.00	3.2	20	0.6	-	15.21
Coryphaenidae	<i>Coryphaena equiselis</i>	pompano dolphinfish	0.01000	2.840	TL	0.85	4.5	39	1	LC	39.44
Coryphaenidae	<i>Coryphaena hippurus</i>	common dolphinfish	0.02020	2.799	FL	0.82	4.4	40	1	LC	64.06
Dasyatidae	<i>Bathytoshia brevicaudata</i>	short-tail stingray	0.03320	2.940	DW	1.00	3.9	87	0.63	LC	-
Dasyatidae	<i>Dasyatidae sp</i>	whiptail stingrays	0.01290	3.031	DW	1.00	-	-	-	-	-
Delphinidae	<i>Delphinus delphis</i>	common dolphin	0.01580	2.910	BL	1.00	4.5	61	-	LC	204.91
Delphinidae	<i>Globicephala macrorhynchus</i>	short-finned pilot whale	0.08320	2.662	BL	1.00	4.4	86	-	LC	403.61
Delphinidae	<i>Orcinus orca</i>	killer whale	0.20800	2.577	BL	1.00	4.5	66	-	DD	-
Echeneidae	<i>Remora sp</i>	shark suckers	0.00080	3.358	FL	1.00	3.4	37	0.54	-	8.99
Echeneidae	<i>Remora remora</i>	shark sucker	0.00080	3.358	TL	0.95	3.5	48	0.54	LC	25.39
Echeneidae	<i>Echeneis naucrates</i>	live sharksucker	0.00075	3.358	FL	1.00	3.7	54	0.75	LC	40.6
Echeneidae	<i>Remora australis</i>	whalesucker	0.00275	3.150	TL _B	0.97	3.5	45	0.54	LC	-
Elapidae	<i>Elapidae sp</i>	sea snakes	0.00120	3.000	FL	1.00	4.3	0	-	-	51.88
Elapidae	<i>Hydrophis sp</i>	sea snakes	0.00450	3.000	FL	1.00	-	-	-	-	-
Exocoetidae	<i>Exocoetidae sp</i>	flyingfishes	0.00430	2.878	TL _B	0.86	3.5	16	0.52	-	30.51
Fistulariidae	<i>Fistularia commersonii</i>	bluespotted cornetfish	0.01180	2.727	FL	1.00	4.3	68	0.63	LC	13.05
Fistulariidae	<i>Fistularia sp</i>	cornetfishes	0.00030	3.182	FL	1.00	4.3	64	0.63	-	14.11

Family	Scientific name	Common name	a	b	Source	TL:FL	TR	VUL	PD	IUCN	FL _B
Fistulariidae	<i>Fistularia petimba</i>	red cornetfish	0.00030	3.158	FL	1.00	4.4	71	0.63	LC	-
Istiophoridae	<i>Istiompax indica</i>	black marlin	0.00447	3.130	TL _B	0.89	4.5	78	1	DD	190.82
Istiophoridae	<i>Istiophorus platypterus</i>	Indo-Pacific sailfish	0.00589	3.140	TL _B	0.86	4.5	68	0.75	LC	209.84
Istiophoridae	<i>Makaira mazara</i>	Indo-Pacific blue marlin	0.00200	3.276	FL	-	4.5	70	0.75	VU	209.9
Istiophoridae	<i>Istiophoridae sp</i>	billfishes	0.00710	2.989	FL	-	4.5	50	0.71	-	213.58
Istiophoridae	<i>Kajikia audax</i>	striped marlin	0.00562	3.150	TL _B	0.92	4.5	43	0.75	NT	-
Kyphosidae	<i>Neatypus obliquus</i>	footballer sweep	0.02291	2.990	TL _B	0.93	3.5	27	1	NE	14.81
Kyphosidae	<i>Scorpis aequipinnis</i>	sea sweep	0.01445	3.000	TL _B	0.89	3.3	48	0.53	NE	36.06
Kyphosidae	<i>Kyphosus sydneyanus</i>	silver drummer	0.02260	3.055	FL	0.95	2	59	0.5	NE	50.76
Labridae	<i>Labroides dimidiatus</i>	bluestreak cleaner wrasse	0.00585	3.231	FL	1.00	3.5	24	0.53	LC	5.12
Labridae	<i>Ophthalmolepis lineolata</i>	southern maori wrasse	0.00447	3.140	TL _B	1.00	3.5	36	1	LC	26.12
Labridae	<i>Choerodon rubescens</i>	baldchin groper	0.01698	3.030	TL _B	1.00	3.6	65	0.5	LC	29.81
Labridae	<i>Coris auricularis</i>	western king wrasse	0.01000	3.060	TL _B	1.00	3.5	50	0.5	LC	32.87
Labridae	<i>Notolabrus parilus</i>	brownspotted wrasse	0.01000	3.050	TL _B	1.00	3.6	52	0.51	LC	34.51
Labridae	<i>Suezichthys sp</i>	rainbow wrasse	0.02320	2.927	TL	1.00	-	-	-	-	-
Lamnidae	<i>Isurus oxyrinchus</i>	shortfin mako	0.01670	2.847	FL	0.93	4.5	83	0.78	NT	139.95
Lamnidae	<i>Carcharodon carcharias</i>	white pointer	0.00758	3.085	FL	0.94	4.5	86	1.03	VU	318.64
Lobotidae	<i>Lobotes surinamensis</i>	tripletail	0.02399	3.000	TL _B	1.00	4	35	1	LC	-
Mobulidae	<i>Mobula sp</i>	mobula rays	0.00630	3.000	WD	1.00	3.6	56	0.5	-	99.85
Mobulidae	<i>Mobula birostris</i>	giant mobula	0.16400	3.000	WD	1.00	3.5	78	0.5	VU	-
Molidae	<i>Mola alexandrini</i>	southern ocean sunfish	0.02455	3.010	TL _B	1.00	3.8	81	0.66	NE	260.51
Molidae	<i>Mola mola</i>	ocean sunfish	0.04540	3.050	FL	1.00	3.3	67	0.66	VU	-
Monacanthidae	<i>Rudarius excelsus</i>	diamond leatherjacket	0.02089	2.930	TL _B	1.00	3	10	0.63	LC	1.88
Monacanthidae	<i>Aluterus sp</i>	leatherjacket filefish	0.02140	2.910	FL	1.00	3.3	58	0.56	-	3.67
Monacanthidae	<i>Monacanthidae sp</i>	leatherjackets	0.01440	3.076	FL	1.00	3	32	0.58	-	4.4
Monacanthidae	<i>Eubalichthys caeruleoguttatus</i>	blue-spotted leatherjacket	0.02089	2.930	TL _B	0.95	2.8	36	0.52	DD	4.88
Monacanthidae	<i>Pervagor aspricaudus</i>	orangetail filefish	0.02089	2.930	TL _B	1.00	2.9	20	0.5	LC	8.53
Monacanthidae	<i>Aluterus scriptus</i>	scribbled leatherjacket filefish	0.82300	1.814	TL	1.00	2.8	68	0.56	LC	9.82
Monacanthidae	<i>Nelusetta ayraud</i>	ocean leatherjacket	0.01930	2.808	TL	1.00	3.7	56	1	LC	12.75

Family	Scientific name	Common name	a	b	Source	TL:FL	TR	VUL	PD	IUCN	FL _B
Monacanthidae	<i>Aluterus monoceros</i>	unicorn leatherjacket filefish	0.02220	2.860	TL	0.97	3.8	48	0.56	LC	20.75
Monacanthidae	<i>Cantherhines dumerilii</i>	whitespotted filefish	0.02455	2.920	TL _B	1.00	3.1	36	0.5	LC	24.09
Monacanthidae	<i>Acanthaluteres sp</i>	spiny leatherjacket	0.02140	2.910	FL	-	-	-	-	-	-
Mullidae	<i>Upeneichthys vlamingii</i>	southern goatfish	0.01023	3.090	TL _B	0.88	3.5	31	0.63	NE	-
Myliobatidae	<i>Myliobatis australis</i>	Australian bull ray	0.04410	3.000	TL	1.00	3.6	60	0.5	NE	-
Nomeidae	<i>Psenes sp</i>	driftfishes	0.02000	3.010	FL	-	3.7	27	0.53	-	2.51
Pempheridae	<i>Pempheridae sp</i>	sweeper	0.00370	3.000	TL	-	-	-	-	-	-
Pomacentridae	<i>Chromis klunzingeri</i>	black-headed puller	0.01820	3.000	TL _B	0.88	2.7	15	0.5	NE	-
Pomacentridae	<i>Parma sp</i>	scalyfin	0.02500	2.946	TL	0.91	-	-	-	-	-
Rachycentridae	<i>Rachycentron canadum</i>	cobia	0.00153	3.428	TL	0.89	4	44	1.5	LC	81.69
Rhincodontidae	<i>Rhincodon typus</i>	whaleshark	0.00425	3.000	TL	0.99	3.6	87	1.5	EN	-
Scombridae	<i>Scomber australasicus</i>	blue mackerel	0.00500	3.247	TL	0.92	4.2	43	0.56	LC	31.69
Scombridae	<i>Scombridae sp</i>	mackerels	0.01710	3.006	FL	-	4.2	46	0.6	-	37.51
Scombridae	<i>Thunnus albacares</i>	yellowfin tuna	0.05200	2.798	FL	0.90	4.4	51	0.5	NT	43.4
Scombridae	<i>Thunnus sp</i>	tunas	0.01020	3.080	TL	0.92	4.3	61	0.5	-	46.59
Scombridae	<i>Thunnus obesus</i>	bigeye tuna	0.01190	3.090	FL	0.91	4.5	56	0.5	VU	51.11
Scombridae	<i>Katsuwonus pelamis</i>	skipjack tuna	0.00654	3.293	FL	1.00	4.4	38	1	LC	53.74
Scombridae	<i>Thunnus maccoyii</i>	southern bluefin tuna	0.01670	3.060	FL	0.93	3.9	67	0.5	CR	59.77
Scombridae	<i>Euthynnus affinis</i>	kawakawa	0.02860	2.858	FL	0.94	4.5	37	0.63	LC	66.35
Scombridae	<i>Grammatorcynus sp</i>	spanish mackerels	0.00450	3.000	TL	0.95	4.4	53	0.75	LC	79.15
Scombridae	<i>Scomberomorus commerson</i>	Spanish mackerel	0.01600	2.802	TL	0.94	4.5	52	0.5	NT	118.09
Scombridae	<i>Acanthocybium solandri</i>	wahoo	0.00160	3.275	FL	0.95	4.3	46	1	LC	129.18
Serranidae	<i>Epinephelides armatus</i>	breaksea cod	0.00933	2.970	TL _B	1.00	3.7	54	1	NT	-
Sparidae	<i>Pagrus auratus</i>	pink snapper	0.04470	2.793	FL	0.87	3.6	69	0.52	LC	37.7
Sphyraenidae	<i>Sphyraena barracuda</i>	great barracuda	0.00380	3.086	TL	0.89	4.5	79	0.5	LC	77.3
Sphyraenidae	<i>Sphyraena sp</i>	barracudas	0.00190	3.160	TL	0.89	-	-	-	-	-
Sphyrnidae	<i>Sphyrna mokarran</i>	great hammerhead	0.00191	3.160	TL	0.75	4.3	86	0.5	EN	194.3
Sphyrnidae	<i>Sphyrna lewini</i>	scalloped hammerhead	0.00777	3.067	FL	0.78	4.1	81	0.5	VU	242
Syngnathidae	<i>Syngnathidae sp</i>	Pipefishes	0.02320	2.927	TL	1.00	3.5	17	0.56	-	6.6

Family	Scientific name	Common name	a	b	Source	TL:FL	TR	VUL	PD	IUCN	FL _B
Tetraodontidae	<i>Arothron firmamentum</i>	starry toado	0.03388	2.870	TL _B	1.00	3.4	36	0.5	LC	38.5
Tetraodontidae	<i>Lagocephalus lagocephalus</i>	oceanic puffer	0.00660	3.302	TL	0.96	3.7	44	0.5	LC	50
Tetraodontidae	<i>Lagocephalus sceleratus</i>	silver-cheeked toadfish	0.01940	2.904	FL	0.98	3.7	71	0.5	LC	66.4
Zipphidae	<i>Ziphiidae</i> sp	beaked whales	-	-	BL	1.00	4.5	90	-	DD	261.8

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APPENDIX 5: IDENTIFICATION OF POTENTIAL SPECIES POOL

In processing the video for the locations included in this Dissertation, there were records for 57,634 individual animals. Of these, 49% were recorded to species with 34% recorded to genus (i.e. *Psenes* sp.; **Table A.5. 1**) and 15% to family (i.e. Nomeidae sp.; **Table A.5. 2**); 2.5% of individuals were recorded as “juveniles”. This outcome was largely a function of high visual similarity within genera and families for some taxa.

Resolution of such uncertainties has conventionally required physical examination of specimens (i.e. counting the number of fin rays). However, advances in environmental DNA (eDNA) will help resolve location-specific identification of species (Jarman et al. 2018) and are thus a powerful tool to combine with stereo-BRUVS.

The nomenclature used, that of “sp.” rather than “spp.” follows the guidelines of Winston (1999) where “sp.” refers to any unknown species within a genus (or family) and “spp” denotes that there are multiple conspecifics or confamilials within a sample, noting that for schooling taxa, there may have been multiple species within a record.

To provide transparency with respect to taxonomic identification and to understand the degree of uncertainty associated with identifications to genus and family, this appendix includes two sections:

Section 1 summarises the decisions that were made with regards to overlapping taxa where a species-designation was taken or where identification might span multiple families.

Section 2 identifies the possible species to which an individual can be allocated where an individual can only be classified to family (“family sp.”), and then for those genera within the family, where an individual is allocated to genus (“genus sp.”). Taxa are ordered alphabetically by family and genus and include marine mammals and reptiles, as well as fishes. Where an individual is identified to genus level but there are no individual allocations to family for the associated family, no list of species within that family are included.

The potential pool of species in Section 2 was determined based on distribution maps from FishBase (www.fishbase.org), Fishes of Australia (www.fishesofaustralia.net.au) and the Atlas of Living Australia (www.bie.ala.org.au). Where maps did not agree on distributions, alternative sources were considered including Reef Life Survey (www.reeflifesurvey.com), ID guides and other books (Hutchins and Swainston 1986, Kuitert and Debelius 2007, Last and Stevens 1994, Stuart-Smith et al. 2015), written accounts, and expert opinion. As the mid-water stereo-BRUVS sample at 10 m, species with reported depth distributions deeper than 60 m were excluded even if their horizontal distributions overlapped the region of study.

As distributions vary within genera and families, three regions in the eastern Indian Ocean were used to allocate species to surveys:

- Northern region: Latitudes higher than -17.5° comprising the surveys at Cocos (Keeling) Islands, Ashmore Reef, Long Reef, Argo-Rowley Terrace and the Rowley Shoals;
- Central region: Latitudes between -17.5° and -30° comprising the surveys at the Montebello Islands, Ningaloo Reef / Gascoyne, and Shark Bay; and
- South region: Latitudes lower than -30° comprising the surveys at Perth Canyon, Geographe Bay, Gracetown, Bremer Canyon and the Recherche Archipelago (western, central and eastern).

These largely map to the bioregions used by the DPIRD (Gaughan et al. 2019). In Chapter 4, the southern region was separated into the Southwest (Perth, Geographe and Gracetown) and South (Bremer and Recherche) due to the fact that they are located on separate oceans, the distributions of most temperate species do not allow this granularity of separation and as such, they are here presented as a combined “south” distribution.

Summary of results

There were 28,152 individuals (49%) allocated to 111 species representing 81 genera and 35 families. This represents 48.8% of all individuals recorded. Juveniles, individuals typically about 2 cm in length and often relatively translucent, numbered 1,420 individuals, or 2.5% of the total records.

There were 19,484 individuals (34%) allocated to “genus sp.” These individuals represented 21 genera from 14 families (**Table A.5.1**). The genera are globally speciose (230 species; median of 8 species per genus). However, when regional distribution and depth are taken into account, only 108 species are potential candidates (47% of global species pool) and, generally, the “genus sp.” classification was likely to be one of a median of 2-3 species within a genus by region (Section 2). Moreover, the number of genera to which “sp.” designations were made was limited: small individuals of the genera *Decapterus* and *Psenes* accounted for 68% and 18% of the individuals allocated to a genus, with six and two potential species as potential candidates respectively (**Table A.5.1**). The remaining 19 genera were largely under 1% of the records identified to “genus sp.”

There were 8,578 individuals (15%) allocated to 13 families (**Table A.5.2**). Of these, three families, the flyingfishes, seahorses and pipefishes, and beaked whales, had no records allocated at the genus or species level. The Clupeidae and Carangidae accounted for 59%

and 39% of these records. Records of Clupeidae are from a pool of 17 species of the 197 species identified globally, representing 8 genera. For the Carangidae, the pool of potential species is 64 from a global pool of 146 species, and 22 genera.

Table A.5. 1 Number of species within each taxa globally (n-Global) and the study region (n-Region) and the number of records (n-obs) and percentage (%) of individuals identified to “genus sp.” by family.

<i>Family</i>	<i>Genus</i>	<i>Common name</i>	<i>n-Global</i>	<i>n-Region</i>	<i>n-obs</i>	<i>%</i>
Balaenopteridae	<i>Balaenoptera</i>	rorquals	9	8	1	<0.1
Carangidae	<i>Alepes</i>	scads	5	3	3	<0.1
Carangidae	<i>Carangoides</i>	jacks	20	14	254	1.2
Carangidae	<i>Decapterus</i>	mackerel scads	10	6	14140	67.6
Carangidae	<i>Pseudocaranx</i>	trevallies	4	3	1	<0.1
Carangidae	<i>Seriola</i>	amberjacks	11	4	5	<0.1
Carcharhinidae	<i>Carcharhinus</i>	whaler sharks	35	18	66	<1
Clupeidae	<i>Sardinella</i>	sardines	21	4	798	3.8
Echeneididae	<i>Remora</i>	suckerfishes	5	4	2	<0.1
Elapidae	<i>Hydrophis</i>	seasnakes	30	7	1	<0.1
Fistularidae	<i>Fistularia</i>	flutemouths	4	2	65	<1
Labridae	<i>Suezichthys</i>	rainbow wrasses	12	2	1	<0.1
Monacanthidae	<i>Acanthaluteres</i>	leatherjackets	3	3	203	<1
Monacanthidae	<i>Aluterus</i>	filefishes	4	2	2	<0.1
Myliobatidae	<i>Mobula</i>	mantas	6	5	5	<0.1
Nomeidae [†]	<i>Psenes</i>	drift fishes	5	2	3801	18.2
Pomacentridae	<i>Parma</i>	scalyfins	8	4	1	<0.1
Scombridae	<i>Grammatorcynus</i>	seerfishes	2	2	1	<0.1
Scombridae	<i>Thunnus</i>	tunas	8	6	132	<1
Sphyraenidae	<i>Sphyraena</i>	barracudas	28	9	2	<0.1
Total			230	108	19,484	

[†] The Nomeidae were particularly challenging. Whilst for the majority of individuals, there was confidence that individuals were in the *Psenes* genus, it is possible that these are confamilials or representatives of other families. See Section 1 for details.

Table A.5. 2 Number of species within each taxa globally (n-Global) and the study region (n-Region) and the number of records (n-obs) and percentage (%) of individuals identified to “family sp.”.

Family	Genus sp	n- Global	n- Region	n-obs	%tot	%fam
Carangidae	jacks	146	64	3,345	39.1	17.4
Carcharhinidae	requiem sharks	59	28	8	0.1	0.8
Cheloniidae	sea turtles	6	5	2	<0.1	66.7
Clupeidae	herrings	197	17	5,049	59.0	86.4
Dasyatidae	whiptail stringrays	102	24	2	<0.1	25.0
Elapidae	sea snakes	68	25	11	0.1	91.7
Exocoetidae	flyingfishes	71	23	3	<0.1	100
Istiophoridae	billfishes	11	5	6	0.1	9.7
Monacanthidae	leatherjackets	107	41	132	1.5	0.6
Pempheridae	sweepers	78	9	3	<0.1	100
Scombridae	mackerels and tunas	54	24	4	<0.1	0.2
Syngnathidae	seahorses and pipefishes	304	73	5	0.1	100
Ziphiidae	beaked whales	22	10	8	0.1	100
Total		1,225	348	8,578		

SECTION 1: ALLOCATIONS AT THE SPECIES LEVEL

There were a limited number of species that are difficult to distinguish from other taxa and decisions were made to allocate them to a single species, dependent on the region in which they were found. These decisions were made as follows:

- 1) The blacktip shark *Carcharhinus limbatus* and the Australian blacktip shark *Carcharhinus tilstoni* are visually nearly identical and have similar distributions (Rory McAuley, Minderoo Foundation, 2019, pers. comm). Thus, on the balance of their commonness, all individuals were recorded as *C. limbatus*. This was done in order to avoid identifying them as *Carcharhinus* sp. as it was clear these individuals were either one of the two species rather than any of the 18 whaler sharks found in the region.
- 2) The copper shark *Carcharhinus brachyurus* and the dusky shark *Carcharhinus obscurus* can only be distinguished if the interdorsal ridge or upper teeth are visible on *C. brachyurus*. In general, individuals in the northern and central regions (n=127 and n=1 respectively) were identified as *C. obscurus* and individuals in the south (n=88) were identified as *C. brachyurus* given their known relative distributions in Western Australia (Rory McAuley, Minderoo Foundation, 2019, pers. comm.).
- 3) The blue cleaner wrasse *Labroides dimidiatus*, a member of the Labridae, looks very similar to three species of *Aspidontus*, a genus within the Blenniidae: the lance blenny *Aspidontus dussumieri* and the false cleanerfish *Aspidontus taeniatus* and the mimic blenny *Aspidontus tractus*. The two individuals, mean FL of 4.7 cm, were assigned to *L. dimidiatus*, noting that cleaning behaviour of oceanic sharks by *L. dimidiatus* (Oliver et al. 2011) might result in its transfer as “associated fauna” from reefs to open ocean environments and that *L. dimidiatus* has a more pelagic larval stage than the blennies whose larval stages are described as “coastal and shallow”. There are also reports of metamorphosed *L. dimidiatus* observed in pelagic environments although “pelagic” behaviour has also been identified in *A. taeniatus* (Losey 1974). Individuals were also assigned to two genera where, in fact, these records might represent individuals from other families (i.e. the Labridae and the Blenniidae).

- 1) We identified 3,801 individuals as *Psenes* sp, with 60% recorded in the northern region and 39% recorded in the southern region. On average these individuals were 2.5 cm \pm 0.04 SE in length. There are seven species within the family Nomeidae that individuals could represent (**Table A.5.34**) from three genera (*Cubiceps*, *Nomeus* and *Psenes*). However, it is also possible that these records are of young trevallas (Centrolophidae) with potential taxa in the north including blackspot butterflyfish *Psenopsis humerosa*, obscure driftfish, *Psenopsis obscura*, and in the central and south regions, the rudderfish *Centrolophus niger* and the blue-eye trevalla *Hyperoglyphe antarctica*. Although the adults of these species live at depths, typically greater than 200 m, their young are known to frequent shallow waters. Additionally, there is a single species of the Ariommatidae, the Indian driftfish *Ariomma indicum* that may be identified as *Psenes* sp. in the northern region.
- 2) We identified 14,140 individuals as *Decapterus* sp, with a mean FL of 8.6 cm \pm 0.12 SE. In addition to the six potential species identified in **Table A.5.8**, the common jack mackerel, *Trachurus declivis* and yellowtail scad *Trachurus novaezelandiae* are found in the Central and Southern regions of the study area and are difficult to distinguish from decapterids.

SECTION 2: POTENTIAL CONFAMILIALS AND CONGENERS

The following section identifies all potential species within families for individuals identified to “family” and within genera, for individuals identified to “genus”. First, for each family, the number of species for each genus are presented, globally, within Australia, along GWOT and by region (North, Central and South). This provides a sense of how many species are likely to be represented in a “family” in a given region and places this number in the context of the number of species within the family. Second, the species that are likely to be present in GWOT for each family are identified by region. Where taxa are identified to genus level, the individual species for each genus by region are listed. For instance, eight individuals are identified as belonging to the family Carcharhinidae and 66 individuals are identified as *Carcharhinus* sp. There are thus family tables that identify all species for the family, including the genera of *Galeocerdo*, *Negaprion*, *Prionace* and *Rhizoprionodon*, as well as a table for *Carcharhinus* specifically. Where no individuals are identified to genus within a family, as is the case with the flyingfishes Exocoetidae, no genus tables are included. Where a taxa is identified to genus level but there are no individuals identified for that family, no family tables are included. For instance, one individual is identified as *Balaenoptera*, but as no individuals are identified as Balaenopteridae, no family tables are included.

BALAENOPTERIDAE (RORQUALS)

BALAENOPTERA SP. (FINBACK WHALES)

There are 9 species found globally, within the genus *Balaenoptera*, of which 8 species are found in GWOT. Of these, 6 are found in the North, 6 in the Central and 7 in the South.

Species within the *Balaenoptera* genera likely to be found by region and by species are as follows:

Table A.5. 3 Possible GWOT species in the *Balaenoptera* genera.

Taxa	Common name	N	C	S
<i>Balaenoptera acutorostrata</i>	Northern minke whale	+	+	+
<i>Balaenoptera borealis</i>	Sei whale	+	+	+
<i>Balaenoptera brydei</i>	Bryde's whale	+	+	+
<i>Balaenoptera edeni</i>	Eden's whale	-	+	+
<i>Balaenoptera musculus</i>	Blue whale	+	+	+
<i>Balaenoptera omurai</i>	Omura's whale	+	-	-
<i>Balaenoptera physalus</i>	Fin whale	+	+	+
<i>Balaenoptera bonaerensis</i>	Antarctic minke whale	-	-	+

CARANGIDAE SP. (JACKS AND POMPANOS)

There are 146 species found globally, of which 64 species are found in GWOT. Of these 55 are found in North, 61 in the Central and 13 in the South.

Table A.5. 4 Number of species by genus and the number of genera and species by family for the Carangidae family within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Alectis</i> sp.	3	2	2	2	2	1
<i>Alepes</i> sp.	5	4	3	3	2	0
<i>Atule</i> sp.	1	1	1	1	1	0
<i>Carangoides</i> sp.	20	14	13	13	12	0
<i>Caranx</i> sp.	18	8	8	7	8	0
<i>Decapterus</i> sp.	10	7	6	5	6	1
<i>Elegatis</i> sp.	1	1	1	1	1	0
<i>Gnathanodon</i> sp.	1	1	1	1	1	0
<i>Megalaspis</i> sp.	1	1	1	1	1	0
<i>Naucrates</i> sp.	1	1	1	1	1	1
<i>Pantolabus</i> sp.	1	1	1	1	1	0
<i>Parastromateus</i> sp.	1	1	1	1	1	0
<i>Pseudocaranx</i> sp.	4	3	3	0	3	2
<i>Scomberoides</i> sp.	4	4	4	4	4	0
<i>Selar</i> sp.	2	2	2	2	2	0
<i>Selaroides</i> sp.	1	1	1	1	1	0
<i>Seriola</i> sp.	11	4	4	2	4	3
<i>Seriolina</i> sp.	1	1	1	1	1	0
<i>Trachinotus</i> sp.	20	5	4	4	3	2
<i>Trachurus</i> sp.	14	3	2	0	2	2
<i>Ulua</i> sp.	2	2	2	2	2	0
<i>Uraspis</i> sp.	3	3	2	2	2	1
Total						
Genera	30	22	22	20	22	8
Species	146	70	64	55	61	13

Table A.5. 5 Possible GWOT species in the Carangidae family.

Taxa	Common name	N	C	S
<i>Alectis ciliaris</i>	Pennantfish	+	+	+
<i>Alectis indica</i>	Diamond trevally	+	+	-
<i>Alepes apercna</i>	Smallmouth scad	+	+	-
<i>Alepes kleinii</i>	Razorbelly trevally	+	-	-
<i>Alepes vari</i>	Herring scad	+	+	-
<i>Atule mate</i>	Barred yellowtail scad	+	+	-
<i>Carangoides chrysophrys</i>	Longnose trevally	+	+	-
<i>Carangoides coeruleopinnatus</i>	Onion trevally	+	+	-
<i>Carangoides dinema</i>	Shadow trevally	+	+	-
<i>Carangoides equula</i>	Whitefin trevally	+	+	-
<i>Carangoides ferdau</i>	Blue trevally	+	+	-
<i>Carangoides fulvoguttatus</i>	Turum	+	+	-
<i>Carangoides gymnostethus</i>	Bludger trevally	+	+	-
<i>Carangoides headlandensis</i>	Bumpnose trevally	+	+	-
<i>Carangoides humerosus</i>	Epaulette trevally	+	+	-
<i>Carangoides malabaricus</i>	Malabar trevally	+	+	-
<i>Carangoides oblongus</i>	Coachwhip trevally	+	+	-

Taxa	Common name	N	C	S
<i>Carangoides orthogrammus</i>	Thicklip trevally	+	+	-
<i>Carangoides plagiotaenia</i>	Barcheek trevally	+	-	-
<i>Carangoides talamparoides</i>	Whitetongue trevally	+	+	-
<i>Caranx bucculentus</i>	Bluespotted trevally	+	+	-
<i>Caranx heberi</i>	Papuan trevally	-	+	-
<i>Caranx ignobilis</i>	Giant trevally	+	+	-
<i>Caranx lugubris</i>	Black trevally	+	+	-
<i>Caranx melampygus</i>	Bluefin trevally	+	+	-
<i>Caranx papuensis</i>	Brassy trevally	+	+	-
<i>Caranx sexfasciatus</i>	Bigeye trevally	+	+	-
<i>Caranx tille</i>	Tille trevally	+	+	-
<i>Decapterus kurroides</i>	Redtail scab	+	+	-
<i>Decapterus macarellus</i>	Mackerel scad	+	+	-
<i>Decapterus macrosoma</i>	Slender scad	+	+	-
<i>Decapterus muroadsi</i>	Southern mackerel scad	-	+	+
<i>Decapterus russelli</i>	Indian scad	+	+	-
<i>Decapterus tabl</i>	Rough-ear scad	+	+	-
<i>Elegatis bipinnulata</i>	Rainbow runner	+	+	-
<i>Gnathanodon speciosus</i>	Golden trevally	+	+	-
<i>Megalaspis cordyla</i>	Finny scad	+	+	-
<i>Naucrates ductor</i>	Pilotfish	+	+	+
<i>Pantolabus radiates</i>	Fringefin trevally	+	+	-
<i>Parastromateus niger</i>	Black pomfret	+	+	-
<i>Pseudocaranx dinjerra</i>	Dinjerra trevally	-	+	-
<i>Pseudocaranx georgianus</i>	Silver trevally	-	+	+
<i>Pseudocaranx wrighti</i>	Skipjack trevally	-	+	+
<i>Scomberoides commersonnianus</i>	Giant queenfish	+	+	-
<i>Scomberoides lysan</i>	Lesser queenfish	+	+	-
<i>Scomberoides tala</i>	Barred queenfish	+	+	-
<i>Scomberoides tol</i>	Needleskin queenfish	+	+	-
<i>Selar boops</i>	Oxeye scad	+	+	-
<i>Selar crumenophthalmus</i>	Bigeye scad	+	+	-
<i>Selaroides leptolepis</i>	Yellowstripe scad	+	+	-
<i>Seriola dumerili</i>	Amberjack	+	+	+
<i>Seriola hippos</i>	Samsonfish	-	+	+
<i>Seriola lalandi</i>	Yellowtail kingfish	-	+	+
<i>Seriola rivoliana</i>	Highfin amberjack	+	+	-
<i>Seriolina nigrofasciata</i>	Blackbanded amberjack	+	+	-
<i>Trachinotus anak</i>	Giant oystercracker	+	-	-
<i>Trachinotus baillonii</i>	Smallspotted dart	+	+	+
<i>Trachinotus blochii</i>	Snubnose dart	+	+	-
<i>Trachinotus botla</i>	Common dart	+	+	+
<i>Trachurus declivis</i>	Common jack mackerel	-	+	+
<i>Trachurus novaezelandiae</i>	Yellowtail scad	-	+	+
<i>Ulua aurochs</i>	Silvermouth trevally	+	+	-
<i>Ulua mentalis</i>	Longraker trevally	+	+	-
<i>Uraspis secunda</i>	Cottonmouth trevally	+	+	-
<i>Uraspis uraspis</i>	Whitemouth trevally	+	+	+

Species within the family Carangidae likely to be found by region and by genus are as follows:

ALEPES SP. (JACKS)

There are 5 species within the genus *Alepes* found globally and of these, 3 species are found within the GWOT. Of these, 3 are found in the North, 2 in the Central and 0 in the South.

Table A.5. 6 Possible GWOT species in the *Alepes* genus.

Taxa	Common name	N	C	S
<i>Alepes apercna</i>	Smallmouth scad	+	+	-
<i>Alepes kleinii</i>	Razorbelly trevally	+	-	-
<i>Alepes vari</i>	Herring scad	+	+	-

CARANGOIDES SP. (JACKS)

There are 20 species within the genus *Carangoides* found globally and of these, 14 species are found within the GWOT. Of these, 14 are found in the North, 13 in the Central and 0 in the South.

Table A.5. 7 Possible GWOT species in the *Carangoides* genus.

Taxa	Common name	N	C	S
<i>Carangoides chrysophrys</i>	Longnose trevally	+	+	-
<i>Carangoides coeruleopinnatus</i>	Onion trevally	+	+	-
<i>Carangoides dinema</i>	Shadow trevally	+	+	-
<i>Carangoides equula</i>	Whitefin trevally	+	+	-
<i>Carangoides ferdau</i>	Blue trevally	+	+	-
<i>Carangoides fulvoguttatus</i>	Turrum	+	+	-
<i>Carangoides gymnostethus</i>	Bludger trevally	+	+	-
<i>Carangoides headlandensis</i>	Bumpnose trevally	+	+	-
<i>Carangoides humerosus</i>	Epaulette trevally	+	+	-
<i>Carangoides malabaricus</i>	Malabar trevally	+	+	-
<i>Carangoides oblongus</i>	Coachwhip trevally	+	+	-
<i>Carangoides orthogrammus</i>	Thicklip trevally	+	+	-
<i>Carangoides plagiotaenia</i>	Barcheek trevally	+	-	-
<i>Carangoides talamparoides</i>	Whitetongue trevally	+	+	-

DECAPTERUS SP. (SCADS)

There are ten species within the genus *Decapterus* found globally and of these, 6 species are found within the GWOT. Of these, 5 are found in the North, 6 in the Central and 1 in the South.

Table A.5. 8 Possible GWOT species in the *Decapterus* genus.

Taxa	Common name	N	C	S
<i>Decapterus kurroides</i>	Redtail scad	+	+	-
<i>Decapterus macarellus</i>	Mackerel scad	+	+	-
<i>Decapterus macrosoma</i>	Slender scad	+	+	-
<i>Decapterus muroadsi</i> *	Southern mackerel scad	-	+	+
<i>Decapterus russelli</i>	Indian scad	+	+	-
<i>Decapterus tabl</i>	Rough-ear scad	+	+	-

*Distribution table shows that only *D. muroadsi* is located in the southern region, however all *Decapterus* sp. records from this area could also be species within the *Trachurus* genus.

PSEUDOCARANX SP. (TREVALLIES)

There are 4 species within the genus *Pseudocaranx* found globally and of these, 3 species are found within the GWOT. Of these, 0 are found in the North, 3 in the Central and 2 in the South.

Table A.5. 9 Possible GWOT species in the *Pseudocaranx* genus.

Taxa	Common name	N	C	S
<i>Pseudocaranx dinjerra</i>	Dinjerra trevally	-	+	-
<i>Pseudocaranx georgianus</i>	Silver trevally	-	+	+
<i>Pseudocaranx wright</i>	Skipjack trevally	-	+	+

SERIOLA SP. (AMBERJACKS)

There are 11 species within the genus *Seriola* found globally and of these, 4 species are found within the GWOT. Of these, 2 are found in the North, 4 in the Central and 3 in the South.

Table A.5. 10 Possible GWOT species in the *Seriola* genus.

Taxa	Common name	N	C	S
<i>Seriola dumerilli</i>	Amberjack	+	+	+
<i>Seriola hippos</i>	Samsonfish	-	+	+
<i>Seriola lalandi</i>	Yellowtail kingfish	-	+	+
<i>Seriola rivoliana</i>	Highfin amberjack	+	+	-

CARCHARHINIDAE SP. (REQUIEM SHARKS)

There are 59 species found globally, of which 28 species are found in GWOT. Of these 26 are found in North, 27 in the Central and 9 in the South.

Table A.5. 11 Number of species by genus and the number of genera and species by family for the Carcharinidae family within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Carcharhinus</i> sp.	35	24	20	18	20	7
<i>Galeocerdo</i> sp.	1	1	1	1	1	1
<i>Glyphis</i> sp.	5	2	1	1	0	0
<i>Loxodon</i> sp.	1	1	1	1	1	0
<i>Negaprion</i> sp.	2	1	1	1	1	0
<i>Prionace</i> sp.	1	1	1	1	1	1
<i>Rhizoprionodon</i> sp.	7	3	2	2	2	0
<i>Triaenodon</i> sp.	1	1	1	1	1	0
Total						
Genera	12	9	8	6	5	2
Species	59	35	28	26	27	9

Table A.5. 12 Possible GWOT species in the Carcharhinidae family.

Taxa	Common name	N	C	S
<i>Carcharhinus albimarginatus</i>	Silvertip shark	+	+	-
<i>Carcharhinus altimus</i>	Bignose shark	+	+	-
<i>Carcharhinus amblyrhynchooides</i>	Graceful shark	+	+	-
<i>Carcharhinus amblyrhynchos</i>	Blacktail reef shark	+	+	-
<i>Carcharhinus amboinensis</i>	Pigeye shark	+	+	-
<i>Carcharhinus brachyurus</i>	Copper shark	-	+	+
<i>Carcharhinus brevipinna</i>	Spinner shark	+	+	+
<i>Carcharhinus cautus</i>	Nervous shark	+	+	-
<i>Carcharhinus falciformis</i>	Silky shark	+	+	-
<i>Carcharhinus fitzroyensis</i>	Creek whaler	+	+	-
<i>Carcharhinus galapagensis</i>	Galapagos shark	-	*	-
<i>Carcharhinus leucas</i>	Bull shark	+	+	+
<i>Carcharhinus limbatus</i>	Blacktip shark	+	+	+
<i>Carcharhinus longimanus</i>	Oceanic whitetip shark	+	+	+
<i>Carcharhinus macloti</i>	Hardnose shark	+	+	-
<i>Carcharhinus melanopterus</i>	Blacktip reef shark	+	+	-
<i>Carcharhinus obscurus</i>	Dusky shark	+	+	+
<i>Carcharhinus plumbeus</i>	Sandbar shark	+	+	+
<i>Carcharhinus sorrah</i>	Spot-tail shark	+	+	-
<i>Carcharhinus tilstoni</i>	Australian blacktip shark	+	+	-
<i>Galeocerdo cuvier</i>	Tiger shark	+	+	+
<i>Glyphis garricki</i>	Northern river shark	+	-	-
<i>Loxodon macrorhinus</i>	Sliteye shark	+	+	-
<i>Negaprion acutidens</i>	Lemon shark	+	+	-
<i>Prionace glauca</i>	Blue shark	+	+	+
<i>Rhizoprionodon acutus</i>	Milk shark	+	+	-
<i>Rhizoprionodon taylori</i>	Australian sharpnose shark	+	+	-

Taxa	Common name	N	C	S
<i>Triaenodon obesus</i>	White tip reef shark	+	+	-

CARCHARHINUS SP. (WHALER SHARKS)

There are 35 species within the genus *Carcharhinus* found globally and of these, 18 species are found within the GWOT. Of these, 18 are found in the North, 20 in the Central and 7 in the South.

Table A.5. 13 Possible GWOT species in the *Carcharhinus* genus.

Taxa	Common name	N	C	S
<i>Carcharhinus albimarginatus</i>	Silvertip shark	+	+	-
<i>Carcharhinus altimus</i>	Bignose shark	+	+	-
<i>Carcharhinus amblyrhynchooides</i>	Graceful shark	+	+	-
<i>Carcharhinus amblyrhynchos</i>	Blacktail reef shark	+	+	-
<i>Carcharhinus amboinensis</i>	Pigeye shark	+	+	-
<i>Carcharhinus brachyurus</i>	Copper shark	-	+	+
<i>Carcharhinus brevipinna</i>	Spinner shark	+	+	+
<i>Carcharhinus cautus</i>	Nervous shark	+	+	-
<i>Carcharhinus falciformis</i>	Silky shark	+	+	-
<i>Carcharhinus fitzroyensis</i>	Creek whaler	+	+	-
<i>Carcharhinus galapagensis</i>	Galapagos shark	-	+	-
<i>Carcharhinus leucas</i>	Bull shark	+	+	+
<i>Carcharhinus limbatus</i>	Blacktip shark	+	+	+
<i>Carcharhinus longimanus</i>	Oceanic whitetip shark	+	+	+
<i>Carcharhinus macloti</i>	Hardnose shark	+	+	-
<i>Carcharhinus melanopterus</i>	Blacktip reef shark	+	+	-
<i>Carcharhinus obscurus</i>	Dusky shark	+	+	+
<i>Carcharhinus plumbeus</i>	Sandbar shark	+	+	+
<i>Carcharhinus sorrah</i>	Spot-tail shark	+	+	-
<i>Carcharhinus tilstoni</i>	Australian blacktip shark	+	+	-

CHELONIIDAE SP. (SEA TURTLES)

There are 6 species found globally, of which 5 species are found in GWOT. Of these, 5 are found in the North, 3 in the Central and 3 in the South.

Table A.5. 14 Number of species by genus and the number of genera and species by family for the Cheloniidae family within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Caretta</i> sp.	1	1	1	1	1	1
<i>Chelonia</i> sp.	1	1	1	1	1	1
<i>Eretmochelys</i> sp.	1	1	1	1	1	1
<i>Lepidochelys</i> sp.	2	1	1	1	0	0
<i>Natator</i> sp.	1	1	1	1	0	0
Total						
Genera	5	5	5	5	3	3
Species	6	5	5	5	3	3

Species within the family Cheloniidae likely to be found by region and by species are as follows:

Table A.5. 15 Possible GWOT species in the Cheloniidae family.

Taxa	Common name	N	C	S
<i>Caretta caretta</i>	Loggerhead turtle	+	+	+
<i>Chelonia mydas</i>	Green sea turtle	+	+	+
<i>Eretmochelys imbricata</i>	Hawksbill turtle	+	+	+
<i>Lepidochelys olivacea</i>	Olive ridley turtle	+	-	-
<i>Natator depressa</i>	Flatback turtle	+	-	-

CLUPEIDAE SP. (HERRINGS)

There are 197 species found globally, of which 17 species are found in GWOT. Of these 14 are found in North, 16 in the Central and 7 in the South.

Table A.5. 16 Number of species by genus and the number of genera and species by family for Clupeidae within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Amblygaster</i> sp.	4	2	2	2	2	0
<i>Anodontostoma</i> sp.	3	1	1	1	1	0
<i>Escualosa</i> sp.	2	1	1	1	1	0
<i>Herklotsichthys</i> sp.	12	7	4	4	4	0
<i>Hyperlophus</i> sp.	3	2	1	0	1	1
<i>Sardinella</i> sp.	21	6	4	3	3	2
<i>Sardinops</i> sp.	14	1	1	0	1	1
<i>Spratelloides</i> sp.	12	3	3	3	3	3
Total						
Genera	55	11	8	6	8	4
Species	197	28	17	14	16	7

Species within the family Clupeidae likely to be found by region and by genus are as follows:

SARDINELLA SP. (SARDINES)

There are 21 species within the *Sardinella* genus found globally and of these, 4 species are found within the GWOT. Of these, 3 are found in the North, 3 in the Central and 2 in the South.

Table A.5. 17 Possible GWOT species in the *Sardinella* genus.

Taxa	Common name	N	C	S
<i>Sardinella albella</i>	White sardinella	+	+	-
<i>Sardinella brachysoma</i>	Deepbody sardinella	+	-	-
<i>Sardinella gibbosa</i>	Goldstripe sardinella	+	+	+
<i>Sardinella lemuru</i>	Scaly mackerel	-	+	+

DASYATIDAE SP. (WHIPTAIL STINGRAYS)

There are 102 species found globally, of which 24 species are found in GWOT. Of these 19 are found in North, 19 in the Central and 3 in the South.

Table A.5. 18 Number of species by genus and the number of genera and species by family for the Dasyatidae within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Bathytoshia</i> sp.	3	1	1	0	1	1
<i>Dasyatis</i> sp.	12	1	1	0	1	1
<i>Hemistrygon</i> sp.	10	2	1	0	1	0
<i>Himantura</i> sp.	15	3	3	3	3	0
<i>Maculabatis</i> sp.	7	2	2	2	1	0
<i>Megatrygon</i> sp.	1	1	1	1	0	0
<i>Neotrygon</i> sp.	8	5	4	2	3	0
<i>Pastinachus</i> sp.	5	2	2	2	2	0
<i>Pateobatis</i> sp.	5	3	3	3	2	0
<i>Pteroplatytrygon</i> sp.	1	1	1	1	1	1
<i>Taeniura</i> sp.	3	1	1	1	1	0
<i>Taeniurops</i> sp.	1	1	1	1	1	0
<i>Urogymnus</i> sp.	6	4	3	3	2	0
Total						
Genera	19	13	13	10	12	3
Species	102	29	24	19	19	3

Species within the family Dasyatidae likely to be found by region and by species are as follows:

Table A.5. 19 Possible GWOT species in the Dasyatidae family.

Taxa	Common name	N	C	S
<i>Bathytoshia brevicaudata</i>	Smooth stingray	-	+	+
<i>Dasyatis thetidis</i>	Black stingray	-	+	+
<i>Hemistrygon parvonigra</i>	Dwarf black stingray	-	+	-
<i>Himantura australis</i>	Australian whipray	+	+	-
<i>Himantura leoparda</i>	Leopard whipray	+	+	-
<i>Himantura uarnak</i>	Honeycomb stingray	+	+	-
<i>Maculabatis astra</i>	Black-spotted whipray	+	+	-
<i>Maculabatis toshi</i>	Brown whipray	+	-	-
<i>Megatrygon microps</i>	Smalleye stingray	+	-	-
<i>Neotrygon annotata</i>	Plain maskray	+	-	-
<i>Neotrygon australiae</i>	Bluespotted maskray	+	+	-
<i>Neotrygon leylandi</i>	Painted maskray	-	+	-
<i>Neotrygon ningalooensis</i>	Ningaloo maskray	-	+	-
<i>Pastinachus ater</i>	Cowtail stingray	+	+	-
<i>Pastinachus sephen</i>	Gundurru	+	+	-
<i>Pateobatis fai</i>	Pink whipray	+	+	-
<i>Pateobatis hortlei</i>	Hortle's whipray	+	-	-
<i>Pateobatis jenkinsii</i>	Jenkin's whipray	+	+	-
<i>Pteroplatytrygon violacea</i>	Pelagic stingray	+	+	+
<i>Taeniura lymma</i>	Yilinggan	+	+	-
<i>Taeniurops meyeri</i>	Blotched fantail ray	+	+	-

Taxa	Common name	N	C	S
<i>Urogymnus acanthobothrium</i>	Mumburarr whipray	+	-	-
<i>Urogymnus asperrimus</i>	Nhiirriil	+	+	-
<i>Urogymnus granulatus</i>	Mangrove whipray	+	+	-

ECHENEIDAE (REMORAS)

REMORA SP. (SUCKERFISHES)

There are 5 species within the genus *Remora* found globally and of these, 4 species are found within the GWOT. Of these, 2 are found in the North, 1 in the Central and 3 in the South.

Table A.5. 20 Possible GWOT species in the *Remora* genus.

Taxa	Common name	N	C	S
<i>Remora albescens</i>	White suckerfish	-	-	+
<i>Remora australis</i>	Whalesucker	-	-	+
<i>Remora osterochir</i>	Marlinsucker	+	-	-
<i>Remora remora</i>	Remora	+	+	+

ELAPIDAE SP. AND HYDROPHIIDAE SP. (SEA SNAKES)

There are of the 68 species found globally, of which 25 species are found in GWOT. Of these, 25 are found in the North, 14 in the Central and 3 in the South.

Table A.5. 21 Number of species by genus and the number of genera and species by families for the Elapidae and Hydrophiidae families within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Acalyptophis</i> sp.	1	1	1	1	1	0
<i>Aipysurus</i> sp.	8	8	7	7	3	0
<i>Astrotia</i> sp.	1	1	1	1	0	0
<i>Disteira</i> sp.	4	2	2	2	2	1
<i>Emydocephalus</i> sp.	2	1	1	1	1	0
<i>Ephalophis</i> sp.	2	1	1	1	1	0
<i>Hydrelaps</i> sp.	1	1	1	1	1	0
<i>Hydrophis</i> sp.	39	10	6	6	4	1
<i>Lapemis</i> sp.	1	1	1	1	0	0
<i>Laticauda</i> sp.	7	2	2	2	0	0
<i>Parahydrophis</i> sp.	1	1	1	1	0	0
<i>Pelamis</i> sp.	1	1	1	1	1	1
Total						
Genera	20	13	12	12	9	3
Species	68	39	25	25	14	3

Species within the families Elapidae and Hydrophiidae likely to be found by region and by species are as follows:

Table A.5. 22 Possible GWOT species in the Elapidae and Hydrophiidae families.

Taxa	Common name	N	C	S
<i>Acalyptophis peronii</i>	Spiny-headed sea snake	+	+	-
<i>Aipysurus apraefrontalis</i>	Short-nosed sea snake	+	+	-
<i>Aipysurus duboisii</i>	Reef shallows sea snake	+	+	-
<i>Aipysurus eydouxii</i>	Spine-tailed sea snake	+	+	-
<i>Aipysurus foliosquama</i>	Leaf-scaled sea snake	+	-	-
<i>Aipysurus fuscus</i>	Dusky sea snake	+	-	-
<i>Aipysurus laevis</i>	Olive-brown sea snake	+	-	-
<i>Aipysurus tenuis</i>	Arafura sea snake	+	-	-
<i>Astrotia stokesii</i>	Stoke's sea snake	+	+	-
<i>Disteira kingii</i>	Spectacled sea snake	+	+	-
<i>Disteira major</i>	Olive headed sea snake	+	+	+
<i>Emydocephalus annulatus</i>	Turtle headed sea snake	+	+	-
<i>Ephalophis greyae</i>	North-western mangrove sea snake	+	+	-
<i>Hydrelaps darwiniensis</i>	Port darwin sea snake	+	+	-
<i>Hydrophis coggeri</i>	Coggers sea snake	+	-	-
<i>Hydrophis czeblukovi</i>	Fine-spined sea snake	+	+	-
<i>Hydrophis elegans</i>	Elegant sea snake	+	+	+
<i>Hydrophis macdowellii</i>	Mcdowells sea snake	+	-	-
<i>Hydrophis ornatus</i>	Ornate reef sea snake	+	+	-
<i>Hydrophis pacificus</i>	Pacific sea snake	+	-	-
<i>Lapemis curtus</i>	Spine-bellied sea snake	+	-	-
<i>Laticauda colubrina</i>	Yellow-lipped sea krait	+	-	-
<i>Laticauda laticaudata</i>	Brown-lipped sea krait	+	-	-
<i>Parahydrophis mertoni</i>	Northern mangrove sea snake	+	-	-
<i>Pelamis platura</i>	Yellowbelly sea snake	+	+	+

EXOCOETIDAE SP. (FLYING FISHES)

There are 71 species found globally, of which 23 species are found in GWOT. Of these 22 are found in North, 17 in the Central and 4 in the South.

Table A.5. 23 Number of species by genus and the number of genera and species by family for the Exocoetidae within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Cheilopogon</i> sp.	31	15	12	11	8	3
<i>Cypselurus</i> sp.	21	5	5	5	4	0
<i>Exocoetus</i> sp.	5	4	2	2	2	1
<i>Fodiator</i> sp.	2	0	0	0	0	0
<i>Hirundichthys</i> sp.	12	3	2	2	2	0
<i>Parexocetus</i> sp.	3	2	2	2	1	0
<i>Prognichthys</i> sp.	6	0	0	0	0	0
Total						
Genera	7	5	5	5	5	2
Species	71	29	23	22	17	4

Species within the family Exocoetidae likely to be found by region and by species are as follows:

Table A.5. 24 Possible GWOT species in the Exocoetidae family.

Taxa	Common name	N	C	S
<i>Cheilopogon abei</i>	Abe's flyingfish	+	-	-
<i>Cheilopogon arcticeps</i>	Bearhead flyingfish	+	-	-
<i>Cheilopogon cyanopterus</i>	Margined flyingfish	+	+	-
<i>Cheilopogon furcatus</i>	Spotfin flyingfish	+	+	+
<i>Cheilopogon heterurus</i>	Piebald flyingfish	+	+	-
<i>Cheilopogon intermedius</i>	Intermediate flyingfish	+	+	-
<i>Cheilopogon katoptron</i>	Indonesian flyingfish	+	-	-
<i>Cheilopogon nigricans</i>	Blacksail flyingfish	+	-	-
<i>Cheilopogon olgae</i>	Olga's flyingfish	-	+	+
<i>Cheilopogon pinnatibarbatus</i>	Tallfin Flyingfish	+	+	+
<i>Cheilopogon spilopterus</i>	Manyspot flyingfish	+	+	-
<i>Cheilopogon suttoni</i>	Sutton's flyingfish	+	+	-
<i>Cypselurus angusticeps</i>	Narrowhead flyingfish	+	+	-
<i>Cypselurus hexazona</i>	Darkbar flyingfish	+	+	-
<i>Cypselurus naresii</i>	Pharao flyingfish	+	-	-
<i>Cypselurus oligolepis</i>	Largescale flyingfish	+	+	-
<i>Cypselurus poecilopterus</i>	Yellow-wing flyingfish	+	+	-
<i>Exocoetus monocirrhus</i>	Barbel flyingfish	+	+	-
<i>Exocoetus volitans</i>	Cosmopolitan flyingfish	+	+	+
<i>Hirundichthys oxycephalus</i>	Bony flyingfish	+	+	-
<i>Hirundichthys speculiger</i>	Mirrorwing flyingfish	+	+	-
<i>Parexocoetus brachypterus</i>	Sailfin flyingfish	+	+	-
<i>Parexocoetus mento</i>	African flyingfish	+	-	-

FISTULARIDAE (CORNETFISHES)

FISTULARIA SP. (FLUTEMOUTHS)

There are 4 species within the genus *Fistularia* found globally and of these, 2 species are found within the GWOT. Of these, 2 are found in the North, 2 in the Central and 1 in the South.

Table A.5. 25 Possible GWOT species in the *Fistularia* genus.

Taxa	Common name	N	C	S
<i>Fistularia commersonii</i>	Smooth flutemouth	+	+	-
<i>Fistularia petimba</i>	Rough flutemouth	+	+	+

ISTIOPHORIDAE SP. (BILLFISHES)

There are 11 species found globally, of which 5 species are found in GWOT. Of these 5 are found in North, 5 in the Central and 4 in the South.

Table A.5. 26 Number of species by genus and the number of genera and species by family for the Istiophoridae within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Istiompax</i> sp.	1	1	1	1	1	1
<i>Istiophorus</i> sp.	2	1	1	1	1	1
<i>Kajikia</i> sp.	1	1	1	1	1	1
<i>Makaira</i> sp.	2	1	1	1	1	0
<i>Tetrapturus</i> sp.	4	1	1	1	1	1
Total						
Genera	5	5	5	5	5	4
Species	11	5	5	5	5	4

Species within the family Istiophoridae likely to be found by region and by species are as follows:

Table A.5. 27 Possible GWOT species in the Istiophoridae family.

Taxa	Common name	N	C	S
<i>Istiompax indica</i>	Black marlin	+	+	+
<i>Istiophorus platypterus</i>	Indo-Pacific sailfish	+	+	+
<i>Kajikia audax</i>	Striped marlin	+	+	+
<i>Makaira mazara</i>	Indo-Pacific blue marlin	+	+	-
<i>Tetrapturus angustirostris</i>	Shortbill spearfish	+	+	+

LABRIDAE (WRASSES)

SUEZICHTHYS SP. (RAINBOW WRASSES)

There are 12 species within the genus *Suezichthys* found globally and of these, 2 species are found within the GWOT. Of these, 0 are found in the North, 2 in the Central and 1 in the South.

Table A.5. 28 Possible GWOT species in the *Suezichthys* genus.

Taxa	Common name	N	C	S
<i>Suezichthys cyanolaemus</i>	Bluethroat rainbow wrasse	-	+	+
<i>Suezichthys soelae</i>	Soela wrasse	-	+	-

MONACANTHIDAE SP. (FILEFISHES)

There are 107 species found globally, of which 41 species are found in GWOT. Of these 28 are found in North, 30 in the Central and 17 in the South.

Table A.5. 29 Number of species by genus and the number of genera and species by family for Monacanthidae within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Acanthaluteres</i> sp.	3	3	3	0	2	3
<i>Aluterus</i> sp.	4	2	2	2	2	1
<i>Amanses</i> sp.	1	1	1	1	0	0
<i>Anacanthus</i> sp.	1	1	1	1	1	0
<i>Brachaluteres</i> sp.	4	2	1	0	0	1
<i>Cantherhines</i> sp.	12	3	3	3	2	0
<i>Chaetodermis</i> sp.	1	1	1	1	1	1
<i>Eubalichthys</i> sp.	6	6	4	1	4	4
<i>Meuschenia</i> sp.	8	8	5	1	4	4
<i>Nelusetta</i> sp.	1	1	1	1	1	1
<i>Oxymonacanthus</i> sp.	2	1	1	1	1	0
<i>Paraluteres</i> sp.	2	1	1	1	0	0
<i>Paramonacanthus</i> sp.	15	6	4	4	4	0
<i>Pervagor</i> sp.	8	5	4	4	1	0
<i>Pseudalutarius</i> sp.	1	1	1	1	2	0
<i>Scobinichthys</i> sp.	1	1	1	0	1	1
<i>Thamnaconus</i> sp.	14	6	5	4	4	1
Total						
Genera	28	24	17	14	14	9
Species	107	60	39	26	30	17

Species within the family Monacanthidae likely to be found by region and by genus are as follows:

ACANTHALUTERES SP. (LEATHERJACKETS)

There are 3 species within the genus *Acanthaluteres* found globally and of these, 3 species are found within the GWOT. Of these, 0 are found in the North, 2 in the Central and 2 in the South.

Table A.5. 30 Possible GWOT species in the *Acanthaluteres* genus.

Taxa	Common name	N	C	S
<i>Acanthaluteres brownii</i>	Spiny-tailed leatherjacket	-	+	+
<i>Acanthaluteres spilomelanurus</i>	Bridled leatherjacket	-	+	+
<i>Acanthaluteres vittiger</i>	Brown leatherjacket	-	-	+

ALUTERUS SP. (FILEFISHES)

There are 4 species within the genus *Aluterus* found globally and of these, 2 species are found within the GWOT. Of these, 2 are found in the North, 2 in the Central and 1 in the South.

Table A.5. 31 Possible GWOT species in the *Aluterus* genus.

Taxa	Common name	N	C	S
<i>Aluterus monoceros</i>	Unicorn leatherjacket	+	+	+
<i>Aluterus scriptus</i>	Scrawled leatherjacket	+	+	-

MOBULIDAE (MANTA RAYS AND DEVILFISHES)

MOBULA SP. (MANTA RAYS)

There are 6 species within the genus *Mobula* found globally and of these, 5 species are found within the GWOT. Of these, 3 are found in the North, 4 in the Central and 0 in the South.

Table A.5. 32 Possible GWOT species in the *Mobula* genus.

Taxa	Common name	N	C	S
<i>Mobula alfredi</i>	Alfred manta	+	+	-
<i>Mobula birostris</i>	Giant manta	+	+	-
<i>Mobula eregoodootenkee</i>	Longhorned manta	+	+	-
<i>Mobula thurstoni</i>	Smoothtail mobula	-	+	-

NOMEIDAE SP. (DRIFTFISHES)

There are 16 species found globally, of which 7 species are found in GWOT. Of these 6 are found in North, 66 in the Central and 2 in the South. Although all nomeids were assigned to *Psenes* sp, the information on the other genera within the family is retained.

Table A.5. 33 Number of species by genus and the number of genera and species by family for the Nomeidae within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Cubiceps</i> sp.	10	6	4	3	3	1
<i>Nomeus</i> sp.	1	1	1	1	1	1
<i>Psenes</i> sp.	5	3	2	2	2	0
Total						
Genera	3	3	3	3	3	2
Species	16	10	7	6	6	2

Species within the family Nomeidae likely to be found by region and by species are as follows:

Table A.5. 34 Possible GWOT species in the Nomeidae family.

Taxa	Common name	N	C	S
<i>Cubiceps baxteri</i>	Black cubehead	+	+	-
<i>Cubiceps caeruleus</i>	Blue cubehead	-	-	+
<i>Cubiceps capensis</i>	Cape cubehead	+	+	-
<i>Cubiceps pauciradiatus</i>	Longfin cubehead	+	+	-
<i>Nomeus gronovii</i>	Bluebottle-fish	+	+	+
<i>Psenes arafurensis</i>	Dusky driftfish	+	+	-
<i>Psenes cyanophrys</i>	Freckled driftfish	+	+	-

CUBICEPS SP. (DRIFTFISH)

There are ten species within the genus *Cubiceps* found globally and of these, 4 species are found within the GWOT. Of these, 3 are found in the North, 3 in the Central and 1 in the South.

Table A.5. 35 Possible GWOT species in the *Cubiceps* genus.

Taxa	Common name	N	C	S
<i>Cubiceps baxteri</i>	Black cubehead	+	+	-
<i>Cubiceps caeruleus</i>	Blue cubehead	-	-	+
<i>Cubiceps capensis</i>	Cape cubehead	+	+	-
<i>Cubiceps pauciradiatus</i>	Longfin cubehead	+	+	-

PSENES SP. (DRIFTFISH)

There are 5 species within the genus *Psenes* found globally and of these, 2 species are found within the GWOT. Of these, 2 are found in the North, 2 in the Central and 0 in the South.

Table A.5. 36 Possible GWOT species in the *Psenes* genus.

Taxa	Common name	N	C	S
<i>Psenes arafurensis</i>	Dusky driftfish	+	+	-
<i>Psenes cyanophrys</i>	Freckled driftfish	+	+	-

PEMPHERIDAE SP. (SWEEPERS)

There are 78 species found globally, of which 9 species are found in GWOT. Of these 4 are found in North, 9 in the Central and 5 in the South.

Table A.5. 37 Number of species by genus and the number of genera and species by family for the Pempheridae within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Parapriacanthus</i> sp.	11	3	2	1	2	1
<i>Pempheris</i> sp.	67	13	7	3	7	4
Total						
Genera	2	2	2	2	2	2
Species	78	16	9	4	9	5

Species within the family Pempheridae likely to be found by region and by genus are as follows:

Table A.5. 38 Possible GWOT species in the Pempheridae family.

Taxa	Common name	N	C	S
<i>Parapriacanthus elongates</i>	Elongate bullseye	-	+	+
<i>Parapriacanthus ransonneti</i>	Golden bullseye	+	+	-
<i>Pempheris analis</i>	Bronze bullseye	-	+	-
<i>Pempheris klunzingeri</i>	Rough bullseye	-	+	+
<i>Pempheris multiradiata</i>	Bigscale bullseye	-	+	+
<i>Pempheris ornata</i>	Orangelined bullseye	-	+	+
<i>Pempheris oualensis</i>	Cave sweeper	+	+	+
<i>Pempheris tominagai</i>	Tominaga's sweeper	+	+	-
<i>Pempheris ypsilychnus</i>	Ypsilon bullseye	+	+	-

POMACENTRIDAE (DAMSELFISHES)

PARMA SP. (SCALYFINS)

There are 8 species within the genus *Parma* found globally and of these, 4 species are found within the GWOT. Of these, 0 are found in the North, 1 in the Central and 4 in the South.

Table A.5. 39 Possible GWOT species in the *Parma* genus.

Taxa	Common name	N	C	S
<i>Parma bicolor</i>	Bicolor scalyfin	-	-	+
<i>Parma mccullochi</i>	McCulloch's scalyfin	-	-	+
<i>Parma occidentalis</i>	Western scalyfin	-	+	+
<i>Parma victoriae</i>	Victorian scalyfin	-	-	+

SCOMBRIDAE SP. (MACKERELS AND TUNAS)

There are 54 species found globally, of which 24 species are found in GWOT. Of these 22 are found in North, 23 in the Central and 16 in the South.

Table A.5. 40 Number of species by genus and the number of genera and species by family for the Scombridae within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Acanthocybium</i> sp.	1	1	1	1	1	1
<i>Allothunnus</i> sp.	1	1	1	0	1	1
<i>Auxis</i> sp.	4	2	2	2	2	2
<i>Cybiosarda</i> sp.	1	1	1	1	1	1
<i>Euthynnus</i> sp.	3	1	1	1	1	0
<i>Gasterochisma</i> sp.	1	1	1	0	1	1
<i>Grammatorcynus</i> sp.	2	2	2	2	2	0
<i>Gymnosarda</i> sp.	1	1	1	1	0	0
<i>Katsuwonus</i> sp.	1	1	1	1	1	1
<i>Rastrelliger</i> sp.	3	1	1	1	1	0
<i>Sarda</i> sp.	5	2	1	1	1	1
<i>Scomber</i> sp.	4	1	1	1	1	1
<i>Scomberomorus</i> sp.	18	4	4	4	4	1
<i>Thunnus</i> sp.	8	6	6	6	6	6
Total						
Genera	15	14	14	12	13	10
Species	54	25	24	22	23	16

Species within the family Scombridae likely to be found by region and by genus are as follows:

Table A.5. 41 Possible GWOT species in the Scombridae family.

Taxa	Common name	N	C	S
<i>Acanthocybium solandri</i>	Wahoo	+	+	+
<i>Allothunnus fallai</i>	Slender tuna	-	+	+
<i>Auxis rochei</i>	Bullet tuna	+	+	+
<i>Auxis thazard</i>	Frigate mackerel	+	+	+
<i>Cybiosarda elegans</i>	Leaping bonito	+	+	+
<i>Euthynnus affinis</i>	Mackerel tuna	+	+	-
<i>Gasterochisma melampus</i>	Butterfly mackerel	-	+	+
<i>Grammatorcynus bicarinatus</i>	Shark mackerel	+	+	-
<i>Grammatorcynus bilineatus</i>	Scad mackerel	+	+	-
<i>Gymnosarda unicolor</i>	Dogtooth tuna	+	-	-
<i>Katsuwonus pelamis</i>	Skipjack tuna	+	+	+
<i>Rastrelliger kanagurta</i>	Mouth mackerel	+	+	-
<i>Sarda orientalis</i>	Oriental bonito	+	+	+
<i>Scomber australasicus</i>	Blue mackerel	+	+	+
<i>Scomberomorus commerson</i>	Narrow-barred Spanish mackerel	+	+	+
<i>Scomberomorus munroi</i>	Spotted mackerel	+	+	-
<i>Scomberomorus queenslandicus</i>	Blotched mackerel	+	+	-
<i>Scomberomorus semifasciatus</i>	Broad-barred mackerel	+	+	-
<i>Thunnus alalunga</i>	Albacore	+	+	+
<i>Thunnus albacares</i>	Yellowfin tuna	+	+	+
<i>Thunnus maccoyii</i>	Southern bluefin tuna	+	+	+
<i>Thunnus obesus</i>	Bigeye tuna	+	+	+
<i>Thunnus orientalis</i>	Northern bluefin tuna	+	+	+

Taxa	Common name	N	C	S
<i>Thunnus tonggol</i>	Longtail tuna	+	+	+

GRAMMATORCYNUS SP. (MACKERELS)

There are 2 species within the genus *Grammatorcynus* found globally and of these, 2 species are found within the GWOT. Of these, 2 are found in the North, 2 in the Central and 0 in the South.

Table A.5. 42 Possible GWOT species in the *Grammatorcynus* genus.

Taxa	Common name	N	C	S
<i>Grammatorcynus bicarinatus</i>	Shark mackerel	+	+	-
<i>Grammatorcynus bilineatus</i>	Double-lined mackerel	+	+	-

THUNNUS SP. (TUNAS)

There are 8 species within the genus *Thunnus* found globally and of these, 6 species are found within the GWOT. Of these, 6 are found in the North, 6 in the Central and 6 in the South.

Table A.5. 43 Possible GWOT species in the *Thunnus* genus.

Taxa	Common name	N	C	S
<i>Thunnus alalunga</i>	Albacore tuna	+	+	+
<i>Thunnus albacares</i>	Yellowfin tuna	+	+	+
<i>Thunnus maccoyii</i>	Southern bluefin tuna	+	+	+
<i>Thunnus obesus</i>	Bigeye tuna	+	+	+
<i>Thunnus orientalis</i>	Northern bluefin tuna	+	+	+
<i>Thunnus tonggol</i>	Longtail tuna	+	+	+

SPHYRAENIDAE (BARRACUDAS)

SPHYRAENA SP. (BARRACUDAS)

There are 28 species within the genus *Sphyraena* found globally and of these, 9 species are found within the GWOT. Of these, 7 are found in the North, 7 in the Central and 3 in the South. *Sphyraena* is the sole genus in the Sphyraenidae.

Table A.5. 44 Possible GWOT species in the *Sphyraena* genus.

Taxa	Common name	N	C	S
<i>Sphyraena acutipinnis</i>	Sharpfin barracuda	-	+	-
<i>Sphyraena barracuda</i>	Great barracuda	+	+	-
<i>Sphyraena flavicauda</i>	Yellowtail barracuda	+	+	+
<i>Sphyraena forsteri</i>	Striped seapike	+	+	-
<i>Sphyraena jello</i>	Pickhandle barracuda	+	+	-
<i>Sphyraena novaehollandiae</i>	Short-finned seapike	-	-	+
<i>Sphyraena obtusata</i>	Striped barracuda	+	+	+
<i>Sphyraena putnamae</i>	Chevron barracuda	+	+	-
<i>Sphyraena qenie</i>	Blackfin barracuda	+	-	-

SYNGNATHIDAE SP. (SEAHORSES AND PIPEFISHES)

There are 304 species found globally, of which 73 species are found in GWOT. Of these 39 are found in North, 44 in the Central and 28 in the South.

Table A.5. 45 Number of species by genus and the number of genera and species by family for the Syngnathidae within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Bhanotia</i> sp.	3	1	1	1	0	0
<i>Bulbonaricus</i> sp.	3	2	1	0	1	0
<i>Campichthys</i> sp.	4	3	2	1	2	1
<i>Choeroichthys</i> sp.	6	5	3	2	3	0
<i>Corythoichthys</i> sp.	12	8	4	4	1	0
<i>Cosmocampus</i> sp.	15	4	1	1	0	0
<i>Doryrhamphus</i> sp.	9	4	3	3	1	0
<i>Dunckerocampus</i> sp.	7	2	2	1	1	0
<i>Festucalex</i> sp.	8	3	1	1	1	0
<i>Filicampus</i> sp.	1	1	1	0	1	1
<i>Halicampus</i> sp.	12	8	6	5	5	0
<i>Haliichthys</i> sp.	1	1	1	1	1	0
<i>Heraldia</i> sp.	1	1	1	0	0	1
<i>Hippichthys</i> sp.	6	5	1	1	1	0
<i>Hippocampus</i> sp.	57	27	12	7	9	2
<i>Histiogamphelus</i> sp.	2	2	1	0	0	1
<i>Hypselognathus</i> sp.	2	2	1	0	0	1
<i>Idiotropiscis</i> sp.	3	3	2	0	1	1
<i>Leptoichthys</i> sp.	1	1	1	0	0	1
<i>Lissocampus</i> sp.	5	3	3	1	2	3
<i>Maroubra</i> sp.	2	1	1	0	0	1
<i>Micrognathus</i> sp.	8	4	3	3	2	0
<i>Mitotichthys</i> sp.	4	4	1	0	0	1
<i>Nannocampus</i> sp.	5	3	1	0	1	1
<i>Notiocampus</i> sp.	1	1	1	0	0	1
<i>Penetopteryx</i> sp.	2	1	1	1	0	0
<i>Phoxocampus</i> sp.	3	2	1	1	1	0
<i>Phycodurus</i> sp.	1	1	1	0	0	1
<i>Phyllopteryx</i> sp.	2	2	2	0	1	2
<i>Pugnaso</i> sp.	1	1	1	0	1	1
<i>Solegnathus</i> sp.	5	5	2	2	2	1
<i>Stigmatopora</i> sp.	4	3	2	0	2	2
<i>Stipecampus</i> sp.	1	1	1	0	0	1
<i>Syngnathoides</i> sp.	1	1	1	1	1	0
<i>Trachyrhamphus</i> sp.	3	2	2	2	2	0
<i>Urocampus</i> sp.	2	1	1	0	0	1
<i>Vanacampus</i> sp.	4	4	3	0	1	3
Total						
Genera	57	45	37	18	25	22
Species	304	104	73	39	44	28

Species within the family Syngnathidae likely to be found by region and by genus are as follows:

Table A.5. 46 Possible GWOT species in the Syngnathidae family.

Taxa	Common name	N	C	S
<i>Bhanotia fasciolata</i>	Barbed pipefish	+	-	-
<i>Bulbonaricus brauni</i>	Braun's pughead pipefish	-	+	-
<i>Campichthys galei</i>	Gale's pipefish	-	+	+
<i>Campichthys tricarinatus</i>	Three-keel pipefish	+	+	-
<i>Choeroichthys brachysoma</i>	Pacific shortbody pipefish	+	+	-
<i>Choeroichthys latispinosus</i>	Murion pipefish	-	+	-
<i>Choeroichthys suillus</i>	Pignout pipefish	+	+	-
<i>Corythoichthys amplexus</i>	Redbanded pipefish	+	+	-
<i>Corythoichthys flavofasciatus</i>	Reticulate pipefish	+	-	-
<i>Corythoichthys intestinalis</i>	Messmate pipefish	+	-	-
<i>Corythoichthys schultzi</i>	Schultz's pipefish	+	-	-
<i>Cosmocampus banneri</i>	Rough-ridge pipefish	+	-	-
<i>Doryrhamphus excisus</i>	Bluestripe pipefish	+	-	-
<i>Doryrhamphus janssi</i>	Cleaner pipefish	+	+	-
<i>Doryrhamphus negrosensis</i>	Flagtail pipefish	+	-	-
<i>Dunckerocampus dactyliophorus</i>	Banded pipefish	+	-	-
<i>Dunckerocampus pessuliferus</i>	Yellowbanded pipefish	-	+	-
<i>Festucalex scalaris</i>	Ladder pipefish	+	+	-
<i>Filicampus tigris</i>	Tiger pipefish	-	+	+
<i>Halicampus brocki</i>	Tasselled pipefish	+	+	-
<i>Halicampus dunckeri</i>	Ridgenose pipefish	+	+	-
<i>Halicampus grayi</i>	Mud pipefish	+	+	-
<i>Halicampus macrorhynchus</i>	Whiskered pipefish	+	-	-
<i>Halicampus nitidus</i>	Glittering pipefish	+	+	-
<i>Halicampus spinirostris</i>	Spinynout pipefish	-	+	-
<i>Haliichthys taeniophorus</i>	Ribboned pipehorse	+	+	-
<i>Heraldia nocturna</i>	Upside-down pipefish	-	-	+
<i>Hippichthys penicillus</i>	Beady pipefish	+	+	-
<i>Hippocampus alatus</i>	Winged seahorse	+	+	-
<i>Hippocampus angustus</i>	Western spiny seahorse	+	+	-
<i>Hippocampus biocellatus</i>	False-eye seahorse	-	+	-
<i>Hippocampus histrix</i>	Thorny seahorse	+	+	-
<i>Hippocampus kuda</i>	Spotted seahorse	+	-	-
<i>Hippocampus montebelloensis</i>	Montebello seahorse	-	+	-
<i>Hippocampus multispinus</i>	Northern spiny seahorse	+	+	-
<i>Hippocampus paradoxus</i>	Paradoxical seahorse	-	-	+
<i>Hippocampus spinosissimus</i>	Hedgehog seahorse	+	+	-
<i>Hippocampus subelongatus</i>	West Australian seahorse	-	+	+
<i>Hippocampus trimaculatus</i>	Lowcrown seahorse	+	-	-
<i>Hippocampus zebra</i>	Zebra seahorse	-	+	-
<i>Histiogamphelus cristatus</i>	Rhino pipefish	-	-	+
<i>Hypselognathus horridus</i>	Shaggy pipefish	-	-	+
<i>Idiotropiscis australe</i>	Southern pygmy pipehorse	-	-	+
<i>Idiotropiscis larsonae</i>	Helen's pygmy pipehorse	-	+	-

Taxa	Common name	N	C	S
<i>Leptoichthys fistularius</i>	Brushtail pipefish	-	-	+
<i>Lissocampus caudalis</i>	Smooth pipefish	-	-	-
<i>Lissocampus fatiloquus</i>	Prophet's pipefish	+	+	+
<i>Lissocampus runa</i>	Javelin pipefish	-	+	+
<i>Maroubra perserrata</i>	Sawtooth pipefish	-	-	+
<i>Micrognathus andersonii</i>	Anderson's pipefish	+	-	-
<i>Micrognathus micronotopterus</i>	Tidepool pipefish	+	+	-
<i>Micrognathus pygmaeus</i>	Thorn-tailed pipefish	+	+	-
<i>Mitotichthys meraculus</i>	Western crested pipefish	-	-	+
<i>Nannocampus subosseus</i>	Bonyhead pipefish	-	+	+
<i>Notiocampus ruber</i>	Red pipefish	-	-	+
<i>Penetopteryx taeniocephalus</i>	Oceanic pipefish	+	-	-
<i>Phoxocampus belcheri</i>	Black rock pipefish	+	+	-
<i>Phycodurus eques</i>	Leafy seadragon	-	-	+
<i>Phyllopteryx dewysea</i>	Ruby seadragon	-	-	+
<i>Phyllopteryx taeniolatus</i>	Common seadragon	-	+	+
<i>Pugnaso curtirostris</i>	Pugnose pipefish	-	+	+
<i>Solegnathus hardwickii</i>	Pallid pipehorse	+	+	-
<i>Solegnathus lettiensis</i>	Gunther's pipehorse	+	+	+
<i>Stigmatopora argus</i>	Spotted pipefish	-	+	+
<i>Stigmatopora nigra</i>	Widebody pipefish	-	+	+
<i>Stipeocampus cristatus</i>	Ringback pipefish	-	-	+
<i>Syngnathoides biaculeatus</i>	Double-end pipehorse	+	+	-
<i>Trachyrhamphus bicoarctatus</i>	Bentstick pipefish	+	+	-
<i>Trachyrhamphus longirostris</i>	Straightstick pipefish	+	+	-
<i>Urocampus carinirostris</i>	Hairy pipefish	-	-	+
<i>Vanacampus margaritifer</i>	Mother-of-pearl pipefish	-	+	+
<i>Vanacampus phillipi</i>	Port Phillip pipefish	-	-	+
<i>Vanacampus poecilolaemus</i>	Longsnout pipefish	-	-	+

ZIPHIIDAE SP. (BEAKED WHALES)

There are 22 species found globally, of which ten species are found in GWOT. Of these, 3 are found in the North, four in the Central and eight in the South.

Table A.5. 47 Number of species by genus and the number of genera and species by family for the Ziphiidae family within the Great West Ozzie Transect. Taxa are summed by global and Australian distributions, as well as their likely occurrence in the northern, central and southern regions of the GWOT.

Taxa	Global	Australia	GWOT	N	C	S
<i>Berardius</i> sp.	2	1	1	0	0	1
<i>Hyperoodon</i> sp.	2	1	1	0	1	1
<i>Indopacetus</i> sp.	1	1	1	1	0	0
<i>Mesoplodon</i> sp.	15	10	5	1	1	5
<i>Tasmacetus</i> sp.	1	1	1	0	1	0
<i>Ziphius</i> sp.	1	1	1	1	1	1
Total						
Genera	6	6	6	3	4	4
Species	22	15	10	3	4	8

Species within the family Ziphiidae likely to be found by region and by species are as follows:

Table A.5. 48 Possible GWOT species in the Ziphiidae family.

Taxa	Common name	N	C	S
<i>Berardius arnuxii</i>	Arnoux's beaked whale	-	-	+
<i>Hyperoodon planifrons</i>	Southern bottlenose whale	-	+	+
<i>Indopacetus pacificus</i>	Longman's beaked whale	+	-	-
<i>Mesoplodon bowdoini</i>	Andrew's beaked whale	-	-	+
<i>Mesoplodon densirostris</i>	Blainville's beaked whale	+	+	+
<i>Mesoplodon grayi</i>	Gray's beaked whale	-	-	+
<i>Mesoplodon layardii</i>	Strap-toothed whale	-	-	+
<i>Mesoplodon mirus</i>	True's beaked whale	-	-	+
<i>Tasmacetus shepherdi</i>	Shepherd's beaked whale	-	+	-
<i>Ziphius cavirostris</i>	Cuvier's beaked whale	+	+	+

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APPENDIX 6: ROMP DISCUSSIONS

To develop my thinking on how to accelerate ocean research, I held a number of discussions with individuals with experience in marine research, industry including marine specialists, intergovernmental organisations, national governments, Community Facing Corporations, and civil society. These discussions broadly provided insights into the challenges of marine research, how to go about building a next-generation vessel, the role it might play internationally and nationally, and how such a vessel would contribute to corporate engagement in oceans and with the conservation sector and civil society more generally.

Table A.6. 1 Identification of organisations and the positions of those with whom discussions were held as part of developing the ROMP concept; an * indicates those also part of development of the ideas for the “Sea the Future”

Company/Organisation	Met with:
<i>Research Institutes</i>	
1 Berkeley University	Scientists at Lawrence Berkeley National Lab
2 CSIRO*	CEO / Director Oceans and Atmosphere
3 European Space Agency	Lead Researcher
4 Georgetown University	Founding Director, Center for Security and Emerging Technology
5 Harvard University	President / George Bemis Professor of International Law, Faculty Director, Berkman Klein Center for Internet and Society
6 KAUST University	Tarek Ahmed Juffali Research Chair in Red Sea Ecology
Lawrence Berkeley National Laboratory Molecular Foundry*	Staff Scientist Molecular Foundry / Project Scientist
7 London School of Economics	Director
8 Massachusetts Institute of Technology*	Director of the MIT Initiative on the Digital Economy / Senior Director, Strategic Initiatives in the Office of External Relations for the MIT Sloan School of Management/Executive Director of the Initiative on the Digital Economy
9 New York University	Provost
10 Oxford University	Professors
11 Peking University*	Professors
12 Stanford University	Provost Emeritus / Deputy Director, Human-Centered Artificial Intelligence Institute/ Professor of Political Science / Director, Digital Civil Society Lab

Company/Organisation	Met with:
13 Tsinghua University*	Chairperson of the Center for International Security and Strategy
14 Turing Institute	Professors
15 University of California - Santa Barbara	Professors
16 University of Cambridge	Vice Chancellor
17 University of Hawaii *	Professors
18 Woods Hole Institute*	Board member and CEO
19 Yale University	Knight Professor of Constitutional Law and the First Amendment at Yale Law School
20 York University	Professors
<i>Community Facing Corporations</i>	
21 Patagonia*	Head of Corporate Development
22 SC Johnson*	Chief Executive Officer
<i>Industry</i>	
23 Maharajah Group*	Group Director / Chairman
24 SABIC*	Chairman
25 BioCellection	Co-Founder and CEO
26 Circulate Capital*	Founder and CEO
27 Greenmantra*	TBC
28 IBM*	IBM Research Senior Manager of Chemistry and Materials
29 Somnio Global	Founder / Title unknown
<i>Marine Industry</i>	
30 AECOM – Steve Morris	Group President - Design & Consulting America
31 ARUP – Clon Ulrick	Global Lead - Maritime
32 Austal Ships	CEO
33 Bechtel – David Wilson	Chief Innovation Officer
34 Boskalis – Peter Berdowski	CEO
35 Fluor Engineering – Allan Boeckmann	President and CEO
36 Fugro*	CEO
37 Jacobs	Global Director - Ports & Maritime
38 Liquid Robotics*	CEO

Company/Organisation	Met with:
39 LOC - Independent Marine and Engineering Consultancy	Group Director, Energy Services
40 LOC Group	President
41 Maersk	CEO - Maersk Supply Service
42 Somnio Global	Project Leader, Somnio SY
<i>Intergovernmental Organisations</i>	
43 United Nations*	Special Envoy for the Ocean
44 United Nations*	President
45 United Nations Environment Programme*	President
<i>National Governments</i>	
46 Government of Australia*	Minister for the Environment
47 Government of Australia*	Prime Minister
48 Government of Norway*	Special Envoy to the High Level Panel on Building a Sustainable Ocean Economy
49 Government of Singapore*	Ministry of Environment
50 Government of the People's Republic of China*	Vice Minister, Ministry of Foreign Affairs
51 United States - Office of Science and Technology*	Director
<i>Conservation Sector and Civil Society</i>	
52 4Ocean*	CEO
53 AREI*	President/Co-Founder
54 Benioff Ocean Initiative*	Director
55 Conservation International*	President / Chairman / CEO
56 Emerson Collective*	Director, Health
57 EYOS Expeditions	Founding Partner
58 Five Deeps Expedition	Chief Scientist
59 Global Citizen*	Global Director of Policy and Advocacy at the Global Poverty Project
60 Global Fishing Watch*	CEO
61 Lonely Whale*	Executive Director
62 Moore Foundation*	Chief Program Officer
63 Nick Hanauer*	Venture Capitalist
64 Ocean Elders*	Founder and CEO

Company/Organisation	Met with:
65 Ocean Voyagers Institute	President & Founder
66 Oceanic Preservation Society*	CEO
67 OECD*	Principal Administrator. Resource Productivity and Waste
68 Open Philanthropy Project	CEO
69 Soul Buffalo*	Co-Founder and CEO
70 The Pew Charitable Trusts*	Senior Vice President
71 Waitt Foundation*	Founder & Board Chairman
72 WWF*	Marine Program Leader
<i>Corporates & Financial Institutions</i>	
73 Berggruen Institute*	Chairman
74 BlackRock*	Chief Investment Officer of the Natural Resources Equity Team
75 Bridgewater Associates	Founder
76 Brookings Institute*	Vice President and Director - Foreign Policy
77 World Bank *	Global Climate Leader
<i>Consultants</i>	
78 Greenmantle*	Founder
79 McKinsey*	Managing Partner (Perth)
80 WSP Parson Brinkerhoff	President and CEO, Aust & NZ
<i>Media Organisations</i>	
81 21 Century Fox*	Newscorp
82 Endeavor Content*	Partner
83 One Community (aka Good Films)*	President and CEO

APPENDIX 7: SEA THE FUTURE DISCUSSIONS

To develop my thinking on developing a Sea the Future to transform our linear plastic economy to a circular plastic economy, I held a number of discussions with individuals with experience in marine research, senior executives in Community Facing Corporations that rely heavily on plastics, senior executives in the oil, gas, petrochemical and plastic producing sectors, leaders in plastics innovation, technology and recycling, marine industry, intergovernmental organisations, national governments, and civil society. These discussions broadly provided insights into the science behind plastic pollution, how industry and community stakeholders would respond to a Sea the Future and how it would be received internationally and nationally. As this is a significant market intervention, I have also held discussions with leaders in the financial, legal and audit sectors. Finally, I have sought thoughts from large media organisations in terms of communicating this global and significant effort.

Table A.7. 1 Identification of organisations and the positions of those with whom discussions were held as part of developing the “Sea the Future” concept; an * indicates those also part of development of the ideas for ROMP.

Company/Organisation	Met with:
<i>Research Institutes</i>	
1 CSIRO*	CEO / Director Oceans and Atmosphere
2 Lawrence Berkeley National Laboratory	Staff Scientist Molecular Foundry / Project Scientist
3 Massachusetts Institute of Technology*	Director of the MIT Initiative on the Digital Economy / Senior Director, Strategic Initiatives in the Office of External Relations for the MIT Sloan School of Management/Executive Director of the Initiative on the Digital Economy
4 Peking University*	Professors
5 Tsinghua University*	Chairperson of the Center for International Security and Strategy
6 University of Hawaii *	Professors
7 Woods Hole Institute*	Board member and CEO
8 NYU – Abu Dhabi	Former Vice-Chancellor
9 NYU	Professor of Ethics and Finance
10 Algalita Marine Research and Education	Founder
11 University of Melbourne	Chancellor

Company/Organisation	Met with:
<i>Community Facing Corporations</i>	
12 The Consumer Goods Forum	Managing Director
13 Patagonia*	Head of Corporate Development
14 SC Johnson*	Chief Executive Officer
15 Buhler	Chairman
16 Coca Cola	Chief Executive Officer & Chief Operations Officer
17 Coles	Chief Executive Online & Corporate Affairs
18 Container Store	Founder
19 Estee Lauder	Senior Vice President, Global Corporate Citizenship, and Sustainability
20 Loreal USA	Head of CSR and sustainability
21 Mars	Board Member / Vice President, Corporate Affairs, Strategic Initiatives & Sustainability
22 Nestle	Board Member / Executive Vice President & Chief Technology Officer / CEO
23 Nestle Waters	CEO, North America / VP, Chief Sustainability Officer at Nestlé Waters North America / CEO Nestle Waters Global
24 Pepsi	Executive Vice President, Communications; President PepsiCo Foundation, Senior VP Government Affairs
25 Qantas	Chief Executive Officer
26 Starbucks	Board Member / Senior Vice President of Global Public Affairs & Social Impact
27 Tesco	Chief Executive Officer
28 Unilever	General Manager - Unilever Indonesia Foundation at Unilever Indonesia / Global CEO
29 Walmart	Chief Sustainability Officer and President of Walmart Foundation
30 Wesfarmers	Chief Executive Officer
<i>Oil, Gas and Petrochemicals and Plastics</i>	
31 Basic Element	Founder
32 USM	Founder
33 Sibur Holdings	CEO
34 Maharajah Group*	Group Director / Chairman
35 SABIC*	Chairman

Company/Organisation	Met with:
<i>Oil, Gas and Petrochemicals and Plastics</i>	
36 ALPLA	Chief Executive Officer
37 Amcor	Chairman
38 Arcelor Mittal	Chairman and CEO
39 Berry	Vice President of Sustainability
40 BP	CFO and COO
41 Chevron	President and CEO of Chevron Phillips Chemical Company
42 Clorex Company	Vice President and General Manager
43 Dow Chemical	Chief Executive Officer
44 Exxon Mobil	Board Member / President of ExxonMobil Chemical Company / Board Member
45 INEOS	Chief Executive Officer of INEOS Technologies
46 Lyondell Basell	Senior Vice President, Research & Development, Technology & Sustainability
47 Royal Dutch Shell	CEO / Executive Vice President, Shell Chemicals / President, Shell France
48 Sealed Air	President and CEO
49 SINOPEC	Director and Deputy Director General
50 National Petroleum Corporation (CNPC)	Vice President
51 Hunan Valin	Deputy GM
52 Jinalong Group	Chairman
53 Baowu Group	Assistant GM
54 Aavishkaar Group and NEPRA	Partner
55 JSW Steel	Chair/Managing Director
56 Saudi Aramco	CEO and Chairman
57 Ferring Pharmaceuticals	Chairman
58 AEPW	CEO/President
59 American Chemical Council	Assistance General Counsel
60 Dow	CEO
61 Cox Enterprises	Vice President of Environmental Sustainability
62 Shell	President Shell France

Company/Organisation	Met with:
<i>Oil, Gas and Petrochemicals and Plastics</i>	
63 Reliance Industries	Chair/Managing Director
64 TOTAL S.A.	CEO
<i>Plastics Innovation, Technology & Recycling</i>	
65 Circulate Capital*	Founder and CEO
66 Greenmantra*	TBC
67 IBM*	IBM Research Senior Manager of Chemistry and Materials
68 Closed Loop	Managing Director, Center for the Circular Economy/ Co-Founder and CEO
69 Croplife International	CEO
70 Novolex	CEO
71 Suez	CTO
72 Loop Industries	Chief Growth Officer
<i>Marine Industry</i>	
72 Fugro*	CEO
73 Subsea Environmental Services	Managing Partner
74 McDonough Innovation	CEO
75 Liquid Robotics*	CEO
<i>Intergovernmental Organisations</i>	
76 World Business Council on Sustainable Development	President and Director of Plastics
77 IUCN	President
78 United Nations*	Special Envoy for the Ocean
79 United Nations*	President
80 UNDP	Country Program Manager
81 United Nations Environment Programme*	President
<i>National Governments</i>	
82 Government of Australia*	Minister for the Environment
83 Government of Australia*	Prime Minister

Company/Organisation	Met with:
<i>National Governments</i>	
84 Government of Australia	Australia's Ambassador to Russia
85 Government of Norway*	Prime Minister
86 Government of Norway	Special Envoy to the High-Level Panel on Building a Sustainable Ocean Economy
87 Government of Norway	Ministry of Foreign Affairs of Norway
88 Government of Singapore*	Ministry of Environment
89 Government of Russia	Special Representative of the President of the Russian Federation on Environmental Activities
90 Government of Japan	Counsellor of Economic Section, Japanese Embassy USA
91 Government of UAE	Minister for Environment and Climate Change
92 Government of the People's Republic of China - Ministry of Foreign Affairs*	Vice Minister
93 China Council for International Cooperation on Environment and Development	Deputy Director
94 Australia - Great Barrier Reef Marine Park Authority	Chairman and Chief Executive
95 Australia - Office of ACT Chief Minister	Director of Budget and Economic Policy
96 Australia - Southern Metropolitan Regional Council	CEO
97 Seychelles Government	President
98 National Energy Administration	Deputy Director
99 National Development & Reform Commission	Deputy Director - Department of Resource Conservation and Environmental Protection
100 United States - Office of Science and Technology (The White House)*	Director
101 Government of United States	Senior Advisor to Secretary of State
102 United States - Environmental Protection Agency	Assistant Administrator

Company/Organisation	Met with:
<i>Civil Society</i>	
103 4Ocean*	CEO
104 AREI*	President/Co-Founder
105 As You Sow	Senior Vice President
106 Benioff Ocean Initiative*	Director
107 Conservation International*	President / Chairman / CEO
108 Emerson Collective*	Director, Health
109 Global Citizen*	Global Director of Policy and Advocacy at the Global Poverty Project
110 Global Fishing Watch*	CEO
111 Greenpeace	Oceans Campaign Director
112 Lonely Whale*	Executive Director
113 Moore Foundation*	Chief Program Officer
114 Nick Hanauer *	Venture Capitalist
115 Nor-Shipping	Director
116 Ocean Elders*	Founder and CEO
117 Oceanic Preservation Society*	CEO
118 OECD*	Principal Administrator. Resource Productivity and Waste
119 Soul Buffalo*	Co-Founder and CEO
120 Skolkovo Foundation	Chairman
121 The Pew Charitable Trusts*	Senior Vice President
122 Waitt Foundation*	Founder & Board Chairman
123 WWF*	Marine Program Leader
124 World Resources Institute	Director, Oceans
125 Wild Life Conservation Fund	President and CEO
126 Oceana	Chief Policy Officer for North America
127 Wildlife Conservation Society	CEO

Company/Organisation	Met with:
<i>Civil Society</i>	
128 Bloomberg Philanthropies	Founder
129 Common Seas	Managing Director
130 Ellen MacArthur Foundation	Chief Executive
131 Environmental Defense Fund	Executive Vice President
132 New York Aquarium	Director of the WCS New York Aquarium
133 Templeton Foundation	Managing Director, Strategic Alliances
134 Global Ghost Gear Initiative	Director
135 Oceans5	Executive Director
136 The Ocean Clean Up	CEO
137 Waste4Change	Founder and Managing Director
138 Skolkovo	Senior Vice President for Innovation
139 World Economic Forum	Specialist, Global Plastic Action Partnership (Circular Economy)
<i>Corporates & Financial Institutions</i>	
140 Chamber of Commerce	Secretary General
141 California Public Employees Retirement System (Calpers)	Managing Investment Director, Sustainable Investments
142 Association of Russian Banks	Executive Vice-President
143 Berggruen Institute*	Chairman
144 BlackRock*	Chief Investment Officer of the Natural Resources Equity Team
145 Imagine	Founder
146 Blackstone	Vice President ESG
147 Bank of America	Global Head of ESG
148 Barclays	Head of global equity capital markets
149 PIMCO	CIO - Core Strategies and Managing Director
150 Brookings Institute*	Vice President and Director - Foreign Policy
151 World Bank *	Global Climate Leader
152 ACCC	Chairman / General Manager, Adjudication
153 Ball Corporation	Chairman, President, CEO

Company/Organisation	Met with:
<i>Corporates & Financial Institutions</i>	
154 Blackstone	CEO
155 Bloomberg Media	CEO
156 Bluemoon	Partner & Co-Founder
157 Bureo	CEO and Co-Founder
158 Capital Global	Investment Analyst
159 Citi	Vice Chairman and Senior International Advisor
160 Credit Suisse	Director / CEO / CEO Impact Advisory and Finance, Credit Suisse
161 Gallup	Chairman & CEO
162 JP Morgan	CEO / Global Chairman for Investment Banking
163 KCA Associates	Founder
164 Preetara	Ex CEO PepsiCo
165 Legal and General	Manager - Sustainability & Responsible Investment
166 Macquarie Bank	CEO
167 Morgan Stanley	MD of Global Sustainability and Chief Sustainability Investing
168 Fidelity	Managing Director of Research, Fidelity Investments
169 State Street	Assistant Vice President; Environmental, Social, and Governance
170 Goldman Sachs	Managing Director and Head of Environmental Markets Group
171 Morgan Stanley	Chief Marketing Officer and Chief Sustainability Officer
172 Capital Group	Director - ESG Investment
173 Prudential	Head of Impact Investing
174 Dow Jones	General Manager of News & Insights
175 Santander	Global Head of Banking
176 Standard Chartered	Global Head Group Sustainability
177 SystemIQ	Co-Founder
178 Credit Suisse	CEO Impact Advisory and Finance
179 Temasek	Head, Sustainability & Stewardship Group
180 SecondMuse	Director, Indonesia

Company/Organisation	Met with:
<i>Legal & Audit</i>	
181 ISCC	Director of ISCC System
182 SGS	Executive VP, Certification and Business Engagement
183 Skaddens	European co-head of Skadden's International Litigation and Arbitration Group /Associate, Antitrust/Competition
<i>Consultants</i>	
184 Greenmantle*	Founder
185 McKinsey*	Managing Partner (Perth)
186 Alpha Beta	Founder and Director
187 EY	Partner
188 Systemiq	Partner
<i>Media Organisations</i>	
189 21 Century Fox*	Newscorp
190 Endeavor Content*	Partner
191 One Community (aka Good Films)*	President and CEO
192 Anton Partner	Creative Director
193 CBS	Vice-Chair
194 Edelman	President & CEO
195 TED	Curator
196 The Economist	Editor-in-Chief
197 Waste Dive	Independent reporter, writer and editor
198 WPP Plc	Founder
199 XTR	CEO

APPENDIX 8: THE CONVERSATION ARTICLES

THE CONVERSATION

Academic rigor, journalistic flair

Arts + Culture Business + Economy Cities Education **Environment + Energy** Health + Medicine Politics + Society Science + Technology

How to keep slave-caught seafood off your plate

November 8, 2018 2.27pm AEDT

Updated November 8, 2018 5.41pm AEDT

Globalised fishing can leave workers vulnerable to exploitation. Shutterstock.com

How would you feel if you knew that slavery had helped provide the fish on your plate? Our [new research](#) reveals that imported seafood raises the risk of Australians consuming fish caught or processed by workers under slave labour conditions by more than eight times, and identifies some of the warning signs to look out for on a global basis.

Our results are consistent with [increasingly widespread reports](#) of modern slavery in the oceans, as highlighted by the recent [Global Slavery Index](#).

[Recent cases](#) record the abuse of Indonesian, Cambodian, and Myanmar nationals subjected to forced labour on vessels from countries including [South Korea](#), [Taiwan](#), and [Thailand](#), in waters as far afield as [New Zealand](#), [Western Africa](#), [Hawaii](#) and the [UK](#).

Authors

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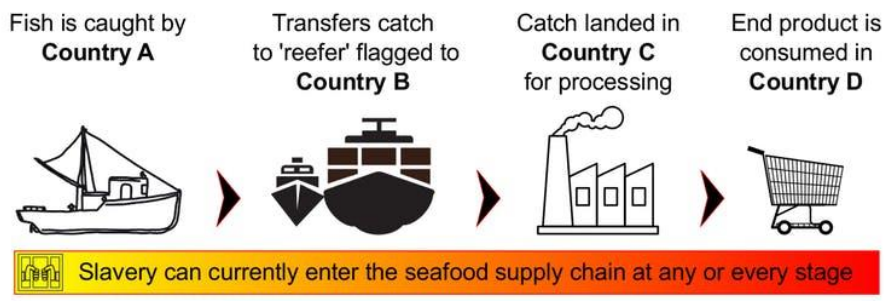
David Tickler
PhD Candidate in Marine Ecology, University of Western Australia

Dirk Zeller
Professor & Director, Sea Around Us – Indian Ocean, University of Western Australia

Forced labour in fisheries is tied to the ongoing depletion of our oceans. Fish catches [peaked in 1996](#) and have since declined. Compared with 1950, fishing fleets now travel [twice the distance](#) to catch a third of the fish. As remaining fish are harder and more expensive to catch, and with rising fuel costs, unscrupulous operators reduce costs by exploiting labour.

The use of refrigerated motherships (“reefers”) allows distant-water vessels to refuel and [tranship](#) catch at sea. Enslaved fishers thus might not see land for years, with the inability to oversee labour practices in offshore conditions providing fertile ground for labour abuses.

A globalised seafood industry with opaque supply chains makes it hard for consumers to avoid slave-caught seafood. The [lack of “net-to-table” traceability](#) compounds the challenge of assessing how prevalent slave-caught seafood might be in our grocery stores and restaurants.



Steps in the seafood supply chain.

Our study, published in [Nature Communications](#), used data on prevalence of modern slavery from the Global Slavery Index alongside fisheries catch data from the [Sea Around Us](#) to determine a set of risk factors that are associated with modern slavery in fisheries. We thus move from anecdotal reports to a global risk assessment.

We found that major fish producing countries with evidence of modern slavery share these characteristics:

- high levels of vessel and fuel subsidies provided by national governments, indicating overcapacity and poor profitability
- poor catch reporting, indicating lack of governance
- dependence on fishing far from home ports and in other countries’ waters, beyond the reach of domestic enforcement
- low catch value, which puts pressure on labour costs.

Slave-caught seafood affects us all

Seafood is the world’s most [highly traded](#) food commodity. To estimate how seafood involving forced labour might reach consumers in ostensibly slavery-free countries, we looked at [trade flows](#) of seafood between countries.

Disclosure statement

Andrew Forrest is a Director of the Minderoo Foundation, which provides philanthropic support to initiatives in education, research, Indigenous affairs, disaster response and the arts.

Dirk Zeller is a member of the Sea Around Us research initiative which receives funding from the Oak Foundation, the Marisla Foundation, the Paul M. Angell Family Foundation, the David and Lucile Packard Foundation, the MAVA Foundation, and Oceana. None of the funders have input in or influence on the material presented here. He is an unpaid advisor on the Scientific Advisory Board of Secure Fisheries, a program of One Earth Future, a privately funded and independent operating foundation.

David Tickler and Jessica Meeuwig do not work for, consult, own shares in or receive funding from any company or organisation that would benefit from this article, and have disclosed no relevant affiliations beyond their academic appointment.

We found that imported seafood in US, European and Australian markets raised the risk of consuming slave-caught or processed seafood more than eight times. Increased vigilance over the provenance of seafood entering these markets is thus urgently required.

How slavery in fishing affects Australian consumers



Prevalence of slave-caught seafood based on domestic fisheries (left) vs combined domestic-caught and imported seafood (right).

Some people would argue that we can avoid forced labour in overseas fisheries by increasing fishing in Australia, but this logic is flawed.

Australia has already lost [30% of its large fish](#) in the past decade. The [annual assessment](#) of 95 Commonwealth-managed stocks finds that roughly 20% are of concern because they are overfished or have uncertain status.

Expanding Australian fisheries is unlikely to reduce our reliance on imported seafood, given that higher overseas prices encourage Australian fishers to [export](#) their product rather than sell it into the domestic market, as is the case with rock lobster and tuna.

Most importantly, modern slavery in the fishing sector is a global scourge that will not be resolved by simply reducing foreign seafood imports. It needs Australian leadership in diplomatic and trading relationships.

Eliminating slavery from your plate

We suggest the following ways in which Australia can contribute to eliminating modern slavery from fisheries:

- Support the federal government's Modern Slavery Act, including the appointment of an independent commissioner to advise seafood companies on minimising risk of forced labour in their products. Australia's regional leadership will help other countries shape their own slavery legislation.
- Help seafood importers, processors and retailers target forced labour through initiatives such as the [Seafood Business for Ocean Stewardship](#), which help and incentivise businesses to improve their supply chains and to evaluate [risk](#).

- Ensure our trading partners have fair labour laws that regulate hiring, payment and treatment of fishing crews. [Thailand](#) in particular has responded strongly to labour issues in its fisheries after being sanctioned by the European Union.
- Support international efforts to eliminate [harmful subsidies](#) and reallocate these resources to rebuilding fisheries through well-enforced management, including [fishing-free zones](#) such as marine parks.
- Choose carefully using [consumer seafood guides](#) that report on social justice along with environmental sustainability.

Very few people would intentionally buy seafood caught by slaves. But the lack of monitoring, transparency and sustainability in fisheries management keeps consumers in the dark and fishing crews vulnerable.

Overfishing damages our environment. Slavery causes immeasurable suffering. We can't tolerate encouraging that by what we put on our plate. Ask your local seafood supplier: "where did this fish come from?" Ask your representative politicians too.



Here is a global solution to the plastic waste crisis - and A\$443 million to get it started

September 25, 2019 2.19pm AES

Informal settlements line a plastic-choked river in Manila, Philippines. newsinfo.inquirer.net

Since the mass production of plastic began, almost six billion tonnes of it - [approximately 91%](#) - has remained in our air, land and water. Plastic production and use is embedded in the global economy, and in our natural environment. This culture of waste is clearly perilous and unsustainable.

Our [paper](#), published today in the journal *Frontiers in Marine Science*, argues that only a global, market-driven intervention can stop the plastic tide.

It is backed by a commitment by the Munderoo Foundation, chaired by the lead author, of up to US\$300 million (A\$443 million) to help establish the scheme and ensure its integrity.

The paper argues that the intervention - a voluntary financial contribution paid by global manufacturers of fossil fuel-based plastic - would drive a system-wide transition to recycled plastic. Our modelling shows that this would lead to a dramatic slowdown in the production of new plastic - creating huge benefits for [marine life](#) and [human health](#).

We must turn off the tap

Plastic takes so long to break down that every piece produced since its inception in [1856](#) still exists today, except the small share we've burned into poisonous gases.

Many strategies to address the plastic problem have been proposed to date, and efforts have been commendable. But we are bailing out a bathtub with a thimble - while the tap is running.

We have identified a simple solution: a voluntary industry contribution for new fossil fuel-based plastic production.

Authors

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Disclosure Statement

Andrew Forrest is the chairman of The Munderoo Foundation Pty Ltd. Munderoo Foundation has been funded principally through the donations of Andrew and his wife Nicola Forrest. Andrew is the founder and former director of Fortescue Metals.

David Tickler works for the Munderoo Foundation's Flourishing Oceans Initiative

Jessica Meeuwig does not work for, consult, own shares in or receive funding from any company or organisation that would benefit from this article, and has disclosed no relevant affiliations beyond their academic appointment.

We believe this technical and financial initiative would level the playing field by making recycled plastic more competitively priced, establishing the right market conditions for a circular plastics economy.

We know from our discussions with industry that this would release technology, in particular [chemical or 'polymer-to-polymer' recycling](#), that is proven today but cannot yet compete economically with new fossil fuel-derived plastic. Increased demand from recyclers would transform plastic waste into a commodity, driving plastic recovery and creating incentives for industry to invest and transition. This is already true for materials like aluminium cans, which are highly recycled because the metal has an inherent value.



Ascension Island is thousands of miles from land, yet even there oceanic wildlife can't escape plastic waste. University of Western Australia – Marine Futures Lab / Ascension Island Government

By mobilising new technology to increase recycling rates, [plastic flows to the ocean and the broader environment would slow](#), and hopefully cease altogether. A circular plastics economy would also significantly reduce carbon emissions created through new plastic production.

Our relationship with plastic is broken

The vast majority of plastics produced to date are derived from [fossil fuels](#). Plastics are made from polymers - long molecular chains comprising smaller carbon-based molecules. Oil and gas are the cheapest materials from which to produce raw polymer resin. This resin is then made into plastic by adding dyes, plasticizers and other chemicals.

Fossil fuel-based plastic has countless uses and is produced very cheaply. Plastic recycling has largely been overlooked because, in the developed world at least, our waste is carted away from our homes and often [shipped overseas](#). This leaves little incentive to tackle our plastic addiction.

But our “out of sight, out of mind” mentality cannot persist.

In 2017, China banned imports of 24 types of solid waste, mainly plastics. This revealed the extent to which developed countries had been sending their waste problem elsewhere. In [Australia](#) this led to recyclables being stockpiled, landfilled or sent to countries ill-equipped to handle them.

Media coverage is also increasingly highlighting the environmental impact of our throwaway culture: [plastic washed up on beaches](#), [filling the guts of endangered marine animals](#) and [accumulating en masse in circular ocean currents](#).

This is an abhorrent market failure, which conservatively costs [US\\$ 2.2 trillion](#) (A\$3.25 trillion) each year in environmental and socioeconomic damages not taken into account by business or the consumer.



A turtle with a plastic bag fragment in its mouth. Plastic waste in the world's oceans is devastating some marine life. Melbourne Zoo

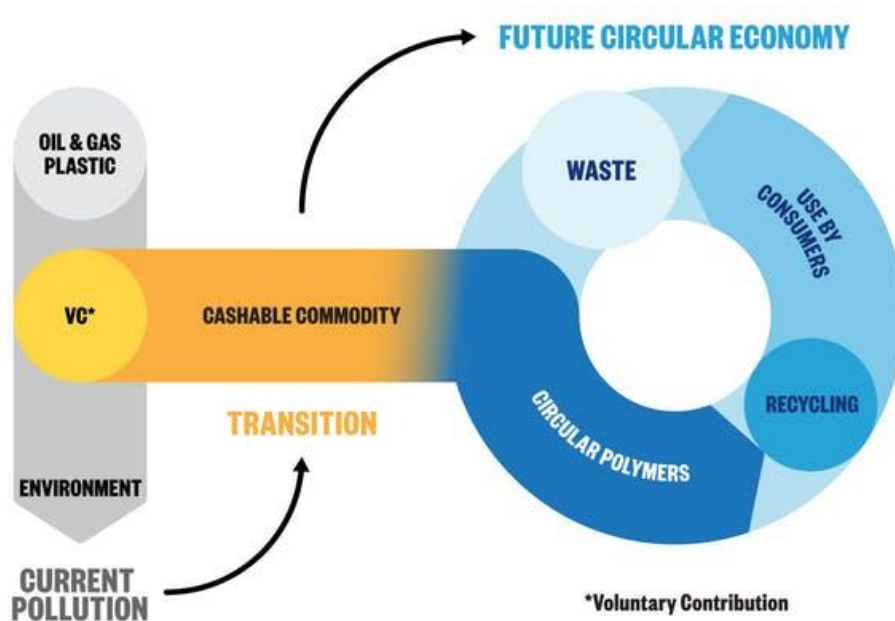
The Sea The Future initiative

We propose an initiative led by global manufacturers in which they make a voluntary financial contribution for each unit of new fossil fuel-based plastic produced. We have dubbed the initiative “Sea The Future”.

Placing a value on plastic both drives its collection and diverts new production away from fossil fuels. The contribution, estimated in our paper as averaging US\$500 (A\$738) per tonne, would be key to encouraging the small number of global resin producers to choose recycled plastic over fossil fuel as their raw material.

The cost would be passed onto consumers via trillions of individual plastic items. The impact would be negligible - say, a few cents on a cup of coffee - and so is likely to gain broad public acceptance.

Anticipating the concerns of regulators that such a move could be perceived as anti-competitive, the lead author has engaged with global law firms to ensure that the initiative is compatible with free market competition law in countries across the world.



The contribution turns plastic waste into a cashable commodity, feeding the circular economy.

The estimated US\$20 billion (A\$29.5 billion) per year raised through the initiative would be used to help establish recycling infrastructure, aid industry transition and remediate the environment. Increased demand and a higher price for recycled material also promises to significantly improve the livelihoods of waste pickers - hundreds of thousands of vulnerable people who currently carve meagre earnings from collecting plastic.

The funds would be administered by a self-regulated global industry body, independently audited to ensure performance, accountability and transparency. To address concerns over governance costs, the Minderoo Foundation has committed to underwrite up to five years' worth of audit fees totalling US\$260 million (A\$384 million), plus cover US\$40 million (A\$59 million) in start-up costs, subject to appropriate conditions.

The future is circular

Public pressure is mounting for action on plastics - and what is bad for the planet is ultimately bad for business. The alternatives to an industry-led approach are less appealing. Plastic bans deny us a useful product upon which our economies rely; taxes typically go directly to general revenue and are unlikely to be applied to plastic waste management. So, tax-derived funds are seldom transferred between nations, ignoring the [transboundary](#) nature of plastic pollution.

Our global discussions with companies throughout the plastics supply chain have revealed that the vast majority recognise the need to move away from a linear plastics economy. They also understand that a global, market-based mechanism is the only path to achieving the system-wide transformation required.

Society discards over 250 million tonnes of valuable polymer, worth at least a US\$ 1,000 per tonne recycled, in plastic waste each year. Soon, if we do nothing, that could grow to 500 million tonnes per annum. What industry would allow half a trillion US dollars of waste each year? Recovering it is simply good business for the environment.