



MONASH University

Evaluating change in the Australian marine protected area (MPA) network: Has growth in area strengthened the capacity to protect biodiversity?

Thesis submitted for the degree of Doctor of Philosophy

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BSc Ecology

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Abstract

Human activities are having a detrimental impact on the world's oceans, including significant decreases in biodiversity and fish stocks. In an effort to halt the global decline of marine biodiversity, conserve ecosystem function, and help promote sustainable fisheries, the establishment of marine protected areas (MPAs) has rapidly increased over the past decade. Despite well documented benefits to fish biomass, spillover effects and ecosystem restoration, only 7% of the world's oceans are currently protected, lagging well behind the 10% target set by international conventions and the 15% protection of terrestrial environments. However, these international targets for protection extend beyond area thresholds to include an ecologically representative and well connected system of MPAs. To effectively evaluate a MPA network's capacity to sustain biodiversity, it is essential to first understand if MPAs are well designed and well placed relative to vulnerable species or habitats.


Australia is well ahead of the global trend with approximately 30% of the Exclusive Economic Zone protected. Over the last two decades, and with the establishment of 44 large, offshore MPAs under federal jurisdiction, Australia has claimed the world's 2nd largest MPA system. The large area protected has been touted as a victory for marine conservation, yet there has been little consideration of whether this growth translates to stronger marine protection. To fill this knowledge gap, spatially explicit data was acquired from around Australia for use in the development of three, high-resolution models of the marine environment. These models were used to evaluate multiple aspects of the MPA system including design over time, representation of different ecosystems, and connectivity via larval dispersal. Additionally, as public involvement is crucial in the MPA planning process, a survey targeting the general public was developed to document the public understanding of marine protection in Australia relative to the current level of protection MPAs provide. This study is the first to systematically evaluate progress in a MPA system by integrating multiple objectives captured by the Convention on Biological Diversity (CBD) Aichi Target 11 and quantifying the MPA system's ability to safeguard Australia's unique biodiversity into the future.

Results of this research reveal that the overall structure and size of the Australian MPA system represents commendable progress toward marine conservation targets. However, the level of protection of many MPAs, especially those within close proximity to the shoreline and therefore in areas of highest anthropogenic impact, is not sufficient to effectively mitigate threats to biodiversity. No-take protection in particular is low and unevenly distributed, with the vast majority targeting tropical environments leaving temperate reefs underrepresented and with lower levels of protection. The placement of MPAs is also crucial, as there are numerous areas around Australia where MPAs are not performing as an integrated network, allowing for the exchange of larvae between populations. Furthermore, the recent downgrades to many of Australia's MPAs, as well as the substantial disconnect between public perceptions of marine protection and the actual levels of protection, suggests there is a clear need to improve education and outreach efforts to ensure the public are informed

about marine conservation policy. Ultimately, the results of this research provide further justification for strengthening the levels of protection for Australia's MPAs and incorporating all aspects of the CBD Aichi Target 11 in future MPA planning.

Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma at any university or equivalent institution and that, to the best of my knowledge and belief, this thesis contains no material previously published or written by another person, except where due reference is made in the text of the thesis.

A black rectangular box redacting the signature of the author.

Signature:

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Date: 26 March 2019

Publications during enrolment

Chapter 2 Roberts, K. E., Valkan, R. S., & Cook, C. N. (2018). Measuring progress in marine protection: A new set of metrics to evaluate the strength of marine protected area networks. *Biological Conservation*, 219, 20-27.

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Thesis including published works declaration

I hereby declare that this thesis contains no material which has been accepted for the award of any other degree or diploma at any university or equivalent institution and that, to the best of my knowledge and belief, this thesis contains no material previously published or written by another person, except where due reference is made in the text of the thesis.

This thesis includes 2 original papers published in peer reviewed journals, 1 paper under review in a peer reviewed journal, and 1 unpublished publication. The core theme of the thesis is the Australian marine protected area (MPA) network and its capacity to protect biodiversity. The ideas, development and writing up of all the papers in the thesis were the principal responsibility of myself, the candidate, working within the School of Biological Sciences under the supervision of Dr Carly Cook.

The inclusion of co-authors reflects the fact that the work came from active collaboration between researchers and acknowledges input into team-based research.

In the case of chapter number 2-5 my contribution to the work involved the following:

Thesis Chapter	Publication Title	Publication Status	Candidate's Contribution	Co-author names	Co-author Monash Student?
2	Measuring progress in marine protection: A new set of metrics to evaluate the strength of marine protected area networks	Published	Design, data collection, data analysis, and writing of manuscript 80%	1. Carly Cook, 15% 2. Rebecca Valkan, 5%	No
3	Bio-physical models of marine environments reveal biases in the representation of protected areas	Published	Design, data collection, data analysis, and writing of manuscript 80%	1. Carly Cook, 15% 2. Grant Duffy, 5%	No
4	Marine protected areas often fail at safeguarding ecological connectivity: An Australian case study	Not Submitted	Design, data collection, data analysis, and writing of manuscript 80%	1. Eric Trembl, 10% 2. Carly Cook, 5% 3. Jutta Beher, 5%	No
5	Evaluating perceptions of marine protection in Australia: does policy match public expectation?	Under Review	Design, data collection, data analysis, and writing of manuscript 80%	1. Carly Cook, 15% 2. Olivia Hill, 5%	No

I have renumbered sections of submitted or published papers in order to generate a consistent presentation within the thesis.

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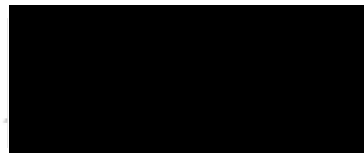


Kelsey E. Roberts

Date: March 26, 2019

The undersigned hereby certify that the above declaration correctly reflects the nature and extent of the student's and co-authors' contributions to this work. In instances where I am not the responsible author I have consulted with the responsible author to agree on the respective contributions of the authors.

Main Supervisor signature:



Carly N. Cook

Date: March 26, 2019

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I've wanted to be a marine biologist for as long as I can remember. I volunteered for my local aquarium all through middle and high school, shadowed numerous biologists on their daily jobs as often as I could, and pestered anyone even remotely connected to the field as to how I could prepare myself for a marine biology career. To all those people that listened and provided advice, there are too many to list here, thank you so much. I am incredibly grateful to have been given the opportunity to pursue my passion for marine science through my PhD and for the chance to do so in Australia. I could not have gotten through these past 3.5 years without the support and encouragement from so many people.

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Chapter **1**

INTRODUCTION

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WHAT IS A MARINE PROTECTED AREA?

Marine environments have been subjected to a wide range of human uses, including commercial and recreational fishing, infrastructure development, pollution, shipping and mining explorations, that have contributed to a steady decline in ocean ecosystem health (Halpern et al. 2007; Klein et al. 2015). The loss of marine biodiversity can have disastrous consequences for food security, resilience to environmental disturbances and the ocean's capacity to act as a global climate regulator (Wales 2008). In recognition of these impacts, governments are increasingly introducing legislation to protect marine ecosystems and promote their recovery. In order to be effective, marine conservation legislation should interface with other legislation that regulates human uses of marine systems like fisheries and tourism. One such mechanism to protect the marine environment is to set aside areas which limit the impact of anthropogenic threats, creating marine protected areas.

A marine protected area (MPA) is defined by the International Union for the Conservation of Nature (IUCN) as: "any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (Day et al. 2012). There is increasing evidence that these ocean sanctuaries are the most effective tool to regulate human pressures on local marine environments (Edgar et al. 2014). Global studies have consistently revealed significant increases in fish biomass, species richness, and diversity in MPAs relative to unprotected areas (Edgar et al. 2014; Halpern 2003).

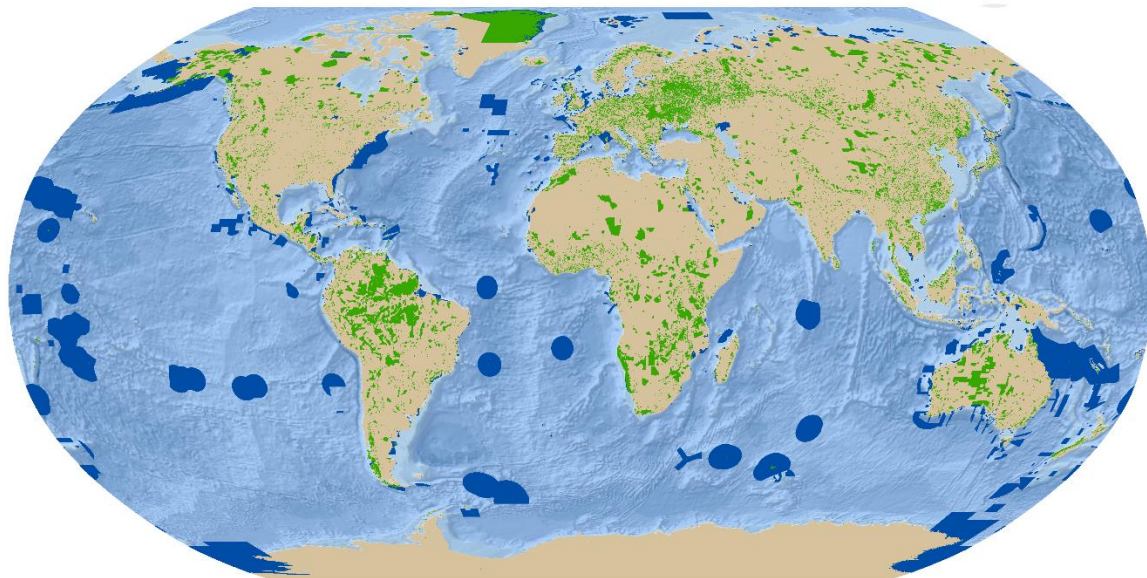
Terrestrial protected areas (PA) have a much longer history than MPAs and thus have far exceeded the area under marine protection (Lindholm and Barr 2001). Although systems of MPAs have been expanding globally, only 7.4% of the global ocean is currently protected by MPAs, compared to 15% of protected area coverage for terrestrial environments (UNEP-WCMC and IUCN 2016; Figure 1). As PAs have grown, there has been more focus on ensuring that growth is strategic. Conservations about ensuring limited funds are spent most effectively, led to the rise of systematic conservation planning, providing a range of tools to ensure new additions to a PA system have the greatest benefit to biodiversity (Margules and Pressey 2000). While these tools developed out of terrestrial PA planning, they have also been applied in marine environments. Given the contentious nature of MPAs, and often

fierce debate over their boundaries, transparent and systematic approaches are particularly important within marine protection.

Systematic conservation planning approaches require sufficient information about the diversity and distribution of threatened species. While these data are generally available in terrestrial systems, information about the marine environment is much more limited. Additionally, knowledge of how well terrestrial protected areas represent biodiversity is acquired through gap assessments, which use species distribution data and vegetation mapping to determine the proportion of species and ecosystems sufficiently represented in protected areas (Kuempel et al. 2016; Venter et al. 2014). Unfortunately, the equivalent data are rarely available for the marine environment, preventing similarly detailed assessment of the representation of marine biodiversity within MPAs. This often creates challenges for assessing the effectiveness of biodiversity conservation since MPA establishment and could contribute to decreasing political support for MPAs in general (Voyer et al. 2012).

In recognition of the important role MPAs play in the conservation of marine ecosystems, targets for protection have been set under the Convention on Biological Diversity (CBD) to expand the area under marine protection to 10% of all coastal and marine waters (Secretariat of the CBD 2011). The primary CBD target that addresses protected areas, Aichi Target 11, recognises the need for ecologically representative, connected and well managed MPAs (Secretariat of the CBD 2011), with total area protected used as the primary metric to measure progress towards this target (Butchart et al. 2010). This focus on total area protected has been criticised because it assumes all MPAs make an equal contribution to marine protection (Collen and Nicholson 2014). Additionally, the total area protected metric disregards the attributes of individual areas, such as their size and the level of protection offered to the marine environment, which have been demonstrated to influence the effectiveness of MPAs (Halpern 2003; Edgar et al. 2014). To gain a comprehensive understanding of progress in marine conservation, it is necessary to take a more nuanced perspective of the value of MPAs to marine biodiversity.

Protected Areas of the world



Source: UNEP-WCMC AND IUCN (2018). Protected Planet: The World Database on Protected Areas (WDPA) [On-line], October 2018, Cambridge, UK: UNEP-WCMC. Available at www.protectedplanet.net



■ Terrestrial protected areas ■ Marine and coastal protected areas



Figure 1. Current area coverage of the global terrestrial and marine protected area estates. Retrieved from: UNEP-WCMC (2018). Protected areas map of the world, October 2018.

EFFECTIVE MPA PLANNING

Implementing systematic conservation planning and measuring progress towards biodiversity conservation targets is not without significant challenges (Klein et al. 2015). Traditionally, progress is measured in relation to the degree to which MPAs comprehensively represent all biodiversity features at levels that adequately ensure their viability. That is, the MPA planning and implementation process should address goals of comprehensiveness, adequacy and representation, which are referred to collectively as the CAR principles (Margules and Pressey 2000). These principles form the foundation of systematic conservation planning, with a clear consensus on the interpretation of each term and easy transferability to any developing or expanding MPA system (Kukkala and Moilanen 2013). A comprehensive MPA system includes the full range of species, ecosystems, and/or habitats (TFMPA 1999). Adequacy refers to the necessary level of protection to retain ecological viability and the integrity of species populations (TFMPA 1999). Measures of species richness and habitat diversity are usually associated with the representativeness principle as a truly representative MPA would capture both common and rare species (Margules and Usher 1981). Additionally, representativeness

focuses on the amount or occurrence of a species within a given area and considers aspects like population range and size (Kukkala and Moilanen 2013). By establishing a network of MPAs guided by the CAR principles, jurisdictions have greater capacity to assess and monitor progress in biodiversity conservation.

In heavily utilised marine areas, there has been considerable interest and numerous attempts to develop a framework that regulates competing uses of the marine environment. The concept of marine spatial planning (MSP) was developed following the creation of several large, multi-purpose MPAs such as the Great Barrier Reef Marine Park and has continued to evolve to become an integral component in ecosystem-based management (Douvere 2008). MSP is broadly defined as: “analysing and allocating parts of three-dimensional marine spaces to specific uses, to achieve ecological, economic, and social objectives that are usually specified through the political process” (Ehler and Douvere 2009). A number of interconnected steps are required to effectively implement MSP. These steps include identifying stakeholders, obtaining financial support and defining existing and future conditions of a particular area (Ehler and Douvere 2009; Figure 2). Due to its flexibility, MSP has already been put into practice in many parts of the world to better direct decisions and resources related to marine conservation.

While in the midst of MPA planning, questions should also be raised and considered relating to opportunity costs, such as those associated with management or the costs accrued by those negatively impacted by MPA establishment (Douvere 2008). While all MPAs are established with different goals in mind, whether it be fisheries management or sociocultural factors, the underlying principle in some capacity is biodiversity conservation (Day et al. 2015). While it is assumed that MPA implementation is always associated with some costs, planning frameworks such as MSP are designed to account for this and arrive at a balance. Devillers et al. (2015) discussed the often perverse outcomes associated with a concerted effort to minimize opportunity costs at the expense of biodiversity conservation. At broad spatial scales, this too often creates ‘residual reserves’, or MPAs that explicitly avoid areas heavily utilised by either fishing or oil/gas interests and are therefore providing protection where biodiversity are not exposed to any threats. Such risks can be reduced by first developing explicit objectives for features at a finer scale and not presuming that MPA establishment equals effective protection (Devillers et al. 2015). Ultimately, the process of

effectively navigating MPA establishment and managing ecological, socio-economic, or political trade-offs requires practitioners to adopt a globally recognised framework, like MSP or systematic conservation planning, capable of both producing favourable biodiversity outcomes and satisfying all stakeholders.

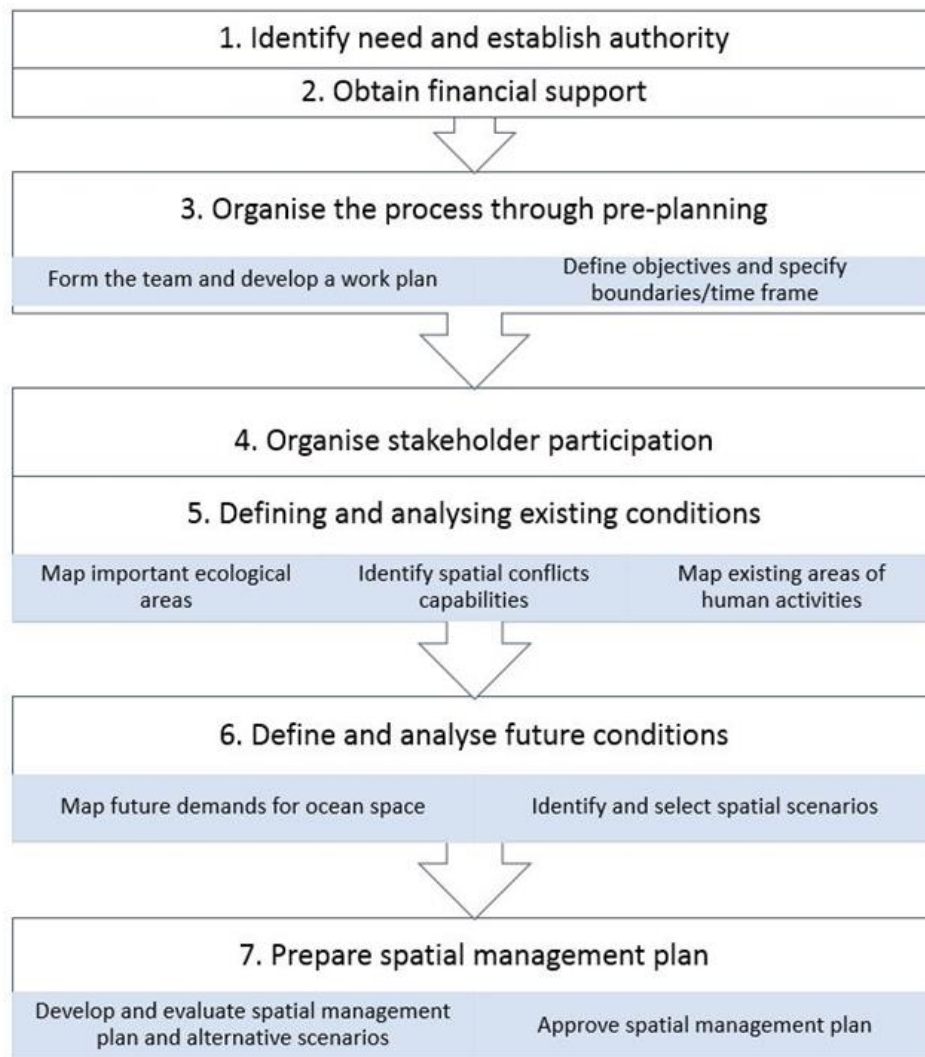


Figure 2. Step-by-step guide to understanding and implementing MSP. Figure adapted from Ehler and Douvere 2009.

THE AUSTRALIAN CONTEXT

Australia is well known for having one of the most diverse assemblages of marine life in the world, with high levels of endemism especially in temperate waters (Butler et al. 2010; Richardson et al. 2009). Ocean resources and marine based tourism in Australia contribute over \$100 billion annually to the local economy (Evans et al. 2017). The health and productivity of Australia’s ocean relies on all aspects of marine conservation at the state,

territory, and federal level. The creation of the Great Barrier Reef Marine Park (GBRMP) established Australia as a pioneer in the field of MPA design and implementation. The GBRMP introduced complex zoning arrangements as well as integrated management into practice after its 1975 establishment and provided increased protection to biodiversity following an updated management plan in 2004 (Day et al. 2002). Australia is now surrounded by a total of 324 MPAs, each varying in their degree of protection from human uses (Roberts et al. 2018; Figure 3).

In Australia, the term MPA encompasses a wide range of reserve types, with varied management objectives and defined zoning schemes offering a range of protection, from strict no-take areas through to multi-use areas that allow natural resource extraction (Boer and Gruber 2010). No-take MPAs (IUCN Category IA and II) have consistently been demonstrated to be the most effective for biodiversity conservation by offering continuous protection to exploited species and recovering fish stocks (Costello 2014; Edgar et al., 2014). The Australian MPA system covers approximately 3.1 million km², the second largest in the world. With significant growth as well as the involvement of many different jurisdictions and legislative frameworks, it is important to assess how well the Australian MPA system acts to protect biodiversity (Roberts et al. 2018). There is growing concern that these MPAs were placed in areas of least political or social objection that are unlikely to be under threat from anthropogenic impacts, calling into question the ability of the network to effectively buffer against the rapid decline of marine biodiversity (Watson et al. 2016). To gain a comprehensive understanding of the value of these MPAs to marine biodiversity, it is essential to know where this growth has occurred and the level of protection offered to critical habitat and species.

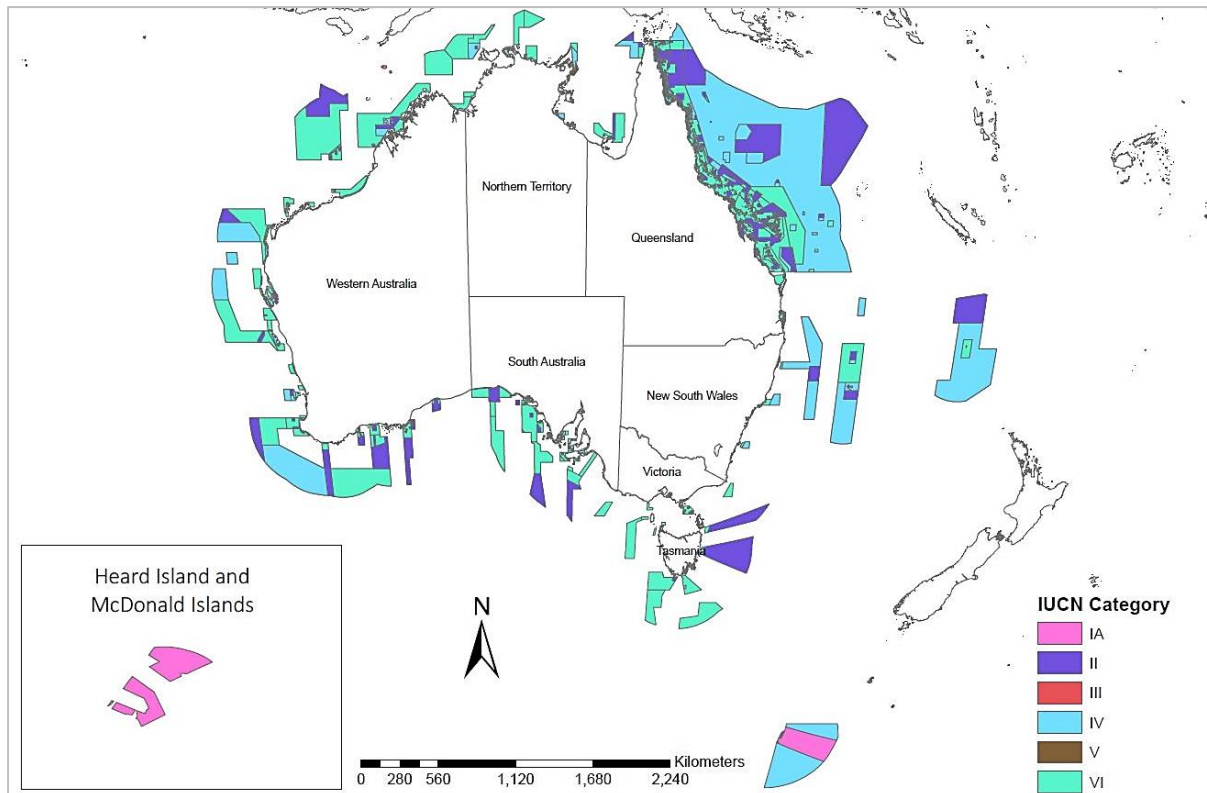


Figure 3. The 2018 Australian MPA system. IUCN Categories are used to define levels of protection in the marine environment, ranging from strict protection (Category IA) to multi-use (Category VI).

RESEARCH AIMS

Australia provides an excellent case study to examine progress in building a robust MPA network because it has more than quadrupled in size over the past two decades to exceed 30% of Australian waters (Roberts et al. 2018). My research aims to systematically assess the structure and design of Australia’s MPAs to determine the network’s capacity to protect biodiversity. To measure progress in the MPA system towards achieving biodiversity conservation targets, chapter 2 aims to develop a suite of metrics that could move beyond total area to evaluate network structure. Chapter 3 then aims to determine how well the MPA network represents Australia’s diverse marine habitats. Chapter 4 aims to determine if functional connections exist among the MPAs. Finally, to assess public perception of marine protection in Australia, chapter 5 aims to determine whether there is a mismatch between the public perceptions of the protection offered by MPAs versus that currently offered under legislation.

This dissertation is written as a ‘thesis including published works’, consisting of two published manuscripts (Chapter 2 and 3), one manuscript currently under review (Chapter 4) and one unpublished manuscript of empirical research intended for a peer-reviewed journal (Chapter 5), detailed below. These four chapters are framed by a general introduction (Chapter 1) to place the work in the relevant theoretical context, and a general discussion (Chapter 6) that synthesizes the overall findings and their contribution to the field of marine conservation.

Chapter 2	Proposes a new suite of metrics that can measure progress towards building robust MPA networks for biodiversity conservation
Chapter 3	Utilises the latest available spatial data to construct a bio-physical model of the distribution of Australian marine environments to understand MPA system coverage and the equality of protection
Chapter 4	Quantifies the degree to which the collection of MPAs in Australia functions as an ecological network
Chapter 5	Developed and distributed a survey to understand how the public perceives the protection offered to the marine environment by Australian MPAs

This research will add substantial knowledge to the field of marine conservation and inform the evaluation of MPA design. It will also assist future MPA design and implementation efforts by providing a clear and comprehensive ‘checklist’ of components that are crucial to review during the planning process.

Chapter 2

MEASURING PROGRESS IN MARINE PROTECTION: A NEW SET OF METRICS TO EVALUATE THE STRENGTH OF MARINE PROTECTED AREA NETWORKS

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ABSTRACT

Marine protected areas (MPAs) have proven to be a valuable tool for both promoting the sustainable use of marine resources and long-term biodiversity conservation outcomes. Targets for marine protection under the Convention on Biological Diversity have seen rapid growth in MPAs globally, with progress judged using targets for total area protected rather than evaluating growth based on the capacity to protect biodiversity. The value of a MPA network to biodiversity conservation depends on a range of attributes of both individual MPAs and portfolios of MPAs, which are not captured by simple area-based targets. Therefore, a clear and efficient set of metrics are needed to effectively evaluate progress towards building MPA networks, considering the representation and adequacy of protection for biodiversity. We developed a universally applicable set of metrics that can evaluate network structure in relation to its capacity to conserve marine biodiversity. These metrics combine properties of effective individual MPAs with metrics for their capacity to function collectively as a network. To demonstrate the value of these metrics, we apply them to the Australian MPA network, the largest in the world. Collectively, the indicators suggest that while Australia has made significant progress in building a representative and well-structured MPA network, the level of protection offered to marine biodiversity is generally low, with insufficient coverage of no-take MPAs across many bioregions. The metrics reveal how the current value of the MPA network could be greatly increased by reducing the prevalence of multi-use zones that allow extractive activities known to negatively impact biodiversity.

INTRODUCTION

Marine protected areas (MPAs) are increasingly being established around the world in an effort to halt the decline of biodiversity and conserve ecosystem function (Klein et al. 2015; O’Leary et al. 2016). These legally protected ocean sanctuaries are widely accepted as the most effective way to regulate human pressures on the marine environment, such as commercial and recreational fishing, shipping, and mining (Metcalfe et al. 2015). With both the frequency and magnitude of impacts on marine ecosystems increasing globally (Halpern et al. 2015), and the area of marine protection lagging well behind terrestrial protection (UNEP-WCMC and IUCN 2016), the strategic expansion of effective marine conservation is increasingly urgent.

MPAs often have multiple objectives, including socioeconomic objectives, such as maintaining public support, and preventing loss of income for local communities (Rossiter and Levine 2014; Watson et al. 2014). However, the primary objective of MPAs is protecting biodiversity, such that a MPA cannot be deemed successful unless it first achieves these biological objectives (Agardy et al. 2011; Fox et al. 2012). The performance of protected areas is dependent on many elements, including their design and management, and the broader context within which they exist (Barnes et al. 2016). Similarly, many factors contribute to MPA effectiveness, with impact evaluation studies around the world identifying several design and management features that correlate with better biological outcomes, such as increased biomass of exploited species. Important design features include larger MPAs, and those more isolated from human activities, being more effective for protecting reef fishes (Edgar et al. 2014). Management features include excluding extractive uses (i.e., no-take reserves) (Halpern 2003; Edgar et al. 2014), actively enforcing restrictions (Edgar et al. 2014) and sufficient resources for management (e.g., staff, equipment) (Gill et al. 2017).

There is growing evidence that to benefit biodiversity, MPAs must also function collectively to protect the full range of marine ecosystems. The broader-scale goals of MPAs, such as protecting ecosystem function, rely on functional connectivity among MPAs to support the ecological and evolutionary processes necessary to enable species to persist over time (Horigue et al. 2015). While often difficult to quantify, a detailed understanding of the range of anthropogenic pressures on marine ecosystems is critical to effective management and conservation (Ban et al. 2010). Likewise, for MPAs to be effective at mitigating threats to

biodiversity, they must be situated in areas of high pressure and biodiversity value, rather than in areas with minimal opportunity costs (Devillers et al. 2015). Therefore, the structure of MPA networks must consider the distribution of key pressures to marine biodiversity in order to design MPAs that offer appropriate levels of protection from threats.

To accommodate MPAs with a broad range of objectives, the International Union for Conservation of Nature (IUCN) recognises different types of MPA, ranging from those primarily for biodiversity conservation (i.e., no-take) to those that promote a range of extractive, recreational and commercial activities (i.e., multi-use) (Day et al. 2012). While jurisdictions apply the IUCN categories to their MPAs, the IUCN provides guidelines for how to assign different categories of protection based on MPA management objectives, with the goal of presenting a globally standardised way for MPAs to be compared across jurisdictions (Day et al. 2012). Given that destructive fishing is considered the most significant threat to marine environments after increasing sea temperature (Halpern et al. 2007), many argue that no-take MPAs are the only legitimate MPAs (e.g., Costello and Ballantine 2015; Miller and Russ 2014). This argument is supported by the empirical evidence for increased species richness, biomass and density of fishes (e.g., Edgar et al. 2014; Lester et al. 2009; Micheli et al. 2004; Starr et al. 2015), as well as increased spill-over effects into surrounding fishing areas (e.g., Gell and Roberts 2003; Halpern et al. 2009) of no-take relative to multi-use areas.

In recognition of the important role MPAs play in the conservation of marine ecosystems, targets for protection have been set under the Convention on Biological Diversity (CBD) to expand marine protection to 10% of all coastal and marine waters by 2020 (Secretariat of the CBD 2011). However, recent reviews have concluded that the 10% target, while ambitious, is unlikely to meet all of the objectives for MPAs (O'Leary et al. 2016). While these targets recognise the need for ecologically representative, connected and well-managed MPAs, the primary metric used to measure progress, total area protected (Tittensor et al. 2014), assumes all MPAs make an equal contribution to marine protection. This focus on the total area protected disregards the attributes of individual areas, such as their size and level of protection from human activities, which have been demonstrated to be important to the effectiveness of MPAs (Edgar et al. 2014; Gill et al. 2017; Halpern 2003).

With increasing recognition of the need to design functioning MPA networks (Krueck et al. 2017), it is critical to develop metrics that can provide more meaningful measures of progress

in marine conservation. These metrics must capture factors we know to be important for the effectiveness of individual MPAs, but also the structure of portfolios of MPAs that reveal their collective contribution to protecting vulnerable marine habitats and species. To better understand how growth in MPAs over time has influenced their capacity to protect biodiversity, we propose a set of metrics that can measure progress towards building robust MPA networks for biodiversity conservation. We then demonstrate these metrics using a case study of the Australian MPA network, the largest in the world (Devillers et al. 2015), using long-term, spatially explicit data. We illustrate how shifting the emphasis from total area protected to evaluating features associated with successful MPAs can provide deeper insights into whether growth in marine protection has improved the strength of the MPA network. These metrics provide a template for improving global efforts to evaluate progress in marine protection, and identifying how to strengthen the value of existing MPA networks for biodiversity conservation.

METHODS

Proposed indicators of the capacity of MPAs to protect biodiversity

To understand how a MPA network has changed over time, and how these changes have influenced the protection for biodiversity, we propose a series of indicators of change:

1. Trend in the number of MPAs;
2. Trend in the total area protected;
3. Trend in size class distribution of MPAs;
4. Trend in the level of protection for marine species;
5. Trend in biodiversity representation;
6. Trend in management effectiveness;
7. Trend in level of connectivity; and
8. Trends in pressures on the marine environment.

Within the context of biological objectives for MPAs, these indicators capture both existing (e.g., 1, 2, 5) measures of progress toward Aichi Target 11 (Secretariat of the CBD 2011) and

those without agreed indicators (e.g., 6, 7, 8; Tittensor et al. 2014). Additionally, they include measures that track features of effective MPAs, as revealed by the impact evaluation literature (e.g., 3, 4, 8; Edgar et al. 2014; Halpern et al. 2010; Klein et al. 2015). Current progress in building MPAs is evaluated based on the 1st and 2nd indicators, which disregards important attributes of successful MPAs.

Given larger MPAs have been shown to offer greater benefits for biodiversity (Edgar et al. 2014), the 3rd indicator, size class distribution, provides a measure of how the area protected is distributed among MPAs. Likewise, the benefits of no-take MPAs for biodiversity are well-documented and extensive relative to multi-use areas (Costello and Ballantine 2015; Edgar et al. 2014; O'Leary et al. 2016). Tracking the distribution of the level of protection for biodiversity offered by MPAs (4th indicator) can help reveal the equality of protection across biodiversity, and the degree to which other objectives for MPA establishment can be accommodated within the network. To avoid residual reserves (i.e., Devillers et al. 2015), it is essential to know which habitats and species are protected within MPAs (5th indicator). When combined with information about the size (3rd indicator) and level of protection offered to biodiversity (4th indicator), this indicator provides a powerful picture of whether MPA expansion is leading to a more robust network than change in representation alone.

In addition to elements of the design of MPAs, effective management (6th indicator) is a critical variable in MPA success or failure (Barnes et al. 2016; Gill et al. 2017). Protected area management effectiveness evaluations offer an opportunity to calculate a numerical indicator of effective management, as proposed by Leverington et al. (2010). Repeat evaluations of MPAs can therefore provide a measure of trends in management effectiveness. Beyond the effectiveness of individual MPAs, species persistence also relies on functional connections (i.e., ability to disperse between areas; Santini et al. 2016) between MPAs (7th indicator). Functional connectivity can be estimated using the protected connected (PC) metric (e.g., Santini et al. 2016), which measures the percentage of species with a specified proportion of their distribution included in connected PAs, using species specific information about dispersal capability and matrix permeability (e.g., ocean currents; Krueck et al. 2017). This metric can therefore estimate trends in functional connectivity of MPAs over time. Finally, understanding the distribution of threats across the marine environment (8th indicator) provides critical information about whether MPAs have levels of protection

and management effort well matched to the threats they experience. Multiple pressures on marine biodiversity can be combined into a threat index (Ban et al. 2010). When represented spatially, change in the threat index can be used to assess trends in the pressures surrounding MPAs, revealing whether MPAs are located in areas of pressure on biodiversity with protection matched to threats.

Analysing all indicators as trends allows for change to be assessed, potentially revealing negative trends, such as Protected Area Downgrading, Downsizing, and Degazettement (PADDD; Mascia and Pallier 2011), using indicators 1, 2, 3, and 4. While each indicator alone provides valuable information, collectively they provide a meaningful assessment of MPA progress, identifying strengths and weaknesses in how area targets are being met. While MPAs can be evaluated based on an endless list of factors (e.g., social, governance), the indicators presented here provide a balance between essential information, as set out by the Aichi Targets, and feasibility.

Case study of change in the Australian Marine Protected Area Network

Australia has the largest MPA network in the world (3.1 million km²), which has more than quadrupled in size over the past two decades to exceed 30% of Australian waters (Devillers et al. 2015). This growth makes it an excellent case study to evaluate how expansion of marine protection has influenced the capacity of the MPA network to protect biodiversity. While the large area protected has been touted as a victory for marine conservation through improved representation of biodiversity (Barr and Possingham 2013), there has been little consideration of whether growth has been well targeted to strengthen marine protection.

To understand the changes in marine protection within the Australian MPA network we used the Collaborative Australian Protected Area Database (CAPAD), which has compiled spatially explicit records of all MPAs, and marine components of terrestrial PAs (Appendix A), on a biennial basis since 2002 (CAPAD 2014).

Trend in the number of MPAs and area protected

We examined every area that came in or was excised from the Australian National Representative System of Marine Protected Areas (NRSMPA) over time to determine the change in the number and size of MPAs throughout the study period.

To report changes in area, we adopt the terminology of PADD with losses in area occurring through *downsizing* (a decrease in the area of an existing MPA) or *degazettement* (total loss of protection status). In keeping with these definitions, we describe increases in area protected as *upsizing* (area added to an existing MPA), or *new* MPAs (whole areas that have newly acquired protection).

Trend in size class distribution of MPAs

To examine trends in the size of MPAs across the network, for each time step we categorized every MPA in the Australian network into size classes using a log scale (e.g. <1, 1-10, 10-100 km² etc.) and calculated the proportion of MPAs in each size class. As large MPAs (greater than 100km²) are more effective in conserving fish biomass (Edgar et al. 2014), we identified the number of MPAs in Australia that have reached or exceeded that size.

Changes to the level of protection for marine biodiversity

Australian MPAs are divided into zones that permit different activities (Agardy et al. 2011), making it necessary to evaluate the level of protection at the scale of zone. Changes to the level of protection for an MPA can occur in two ways: 1) an amendment in the legislation that alters the activities permitted within all MPAs or protection zones of a particular type (termed a policy change); and 2) a change to how a MPA is zoned that moves areas within the MPA into a zone with higher or lower protection (termed a zone change).

To examine changes in the level of protection, we evaluated both policy and zone changes across the network. A decrease in legal restrictions on the number, magnitude, or extent of human activities within a PA is termed a *downgrade* in protection (Mascia and Pallier 2011). In accordance with this definition, we define an *upgrade* in protection as a legal change to the human activities permitted within a PA to reduce harmful impacts. Policy changes were extracted from MPA management plans, government reports and legislation, peer-reviewed literature, media reports and discussions with experts. We verified any changes identified with the relevant agencies across Australia. Zone changes were identified using the CAPAD data (see Appendix A).

Standardising level of protection across jurisdictions

Each jurisdiction in Australia has its own zoning scheme and preliminary analyses revealed that the IUCN categories are not applied consistently, making it difficult to compare the level of protection across jurisdictions. Therefore, to achieve consistent categorisation across the Australian NRSMPA, we conducted a comparative analysis that organised the zones for each jurisdiction according to the activities they permit based on MPA management plans. We then applied the IUCN Best Practice Guidelines (Day et al. 2012) to determine the appropriate IUCN category based on the highest impact activity permitted within a zone (see Table A1 for a detailed breakdown of how these criteria were applied to each jurisdiction). We used this standardised classification of the level of marine protection across Australia to assess trends in area of the MPA network within each IUCN category.

Change in representation and level of protection for biodiversity

While ideally this indicator would be assessed using information on the distribution of marine species and ecosystems, the lack of network-wide habitat mapping and biodiversity monitoring data prevented a detailed assessment of the trend in representation for the NRSMPA. Therefore, we assessed trends in representation of broad marine ecotypes, which provides a coarse measure of the representation of marine biodiversity (Barr and Possingham 2013). To determine how changes to the NRSMPA influenced the representation of Australia's marine ecosystems, we used the Integrated Marine and Coastal Regionalisation of Australia (IMCRA) classification system. IMCRA classifies the marine environment of Australia's EEZ into broad bioregions according to a range of ecosystem attributes (e.g., geomorphic features, climate characteristics) and distributions of selected species (e.g., demersal fishes; IMCRA Technical Group 1998). We calculated the proportion of each IMCRA bioregion with any form of marine protection, presenting the area protected on a categorical scale (e.g. <1%, 1-10%, 10-25%, etc.). We use the 10% target for protection set by the CBD (Secretariat of the CBD 2011) as a reference point to estimate "adequate" protection and quantify change in the level of representation over time.

We were also interested in the equality of protection offered to the different bioregions. Using our standardised IUCN classification, we grouped protection categories into no-take (IUCN Category IA-III) and multi-use zones (IUCN Categories IV-VI) based on whether

extractive uses were permitted. We then calculated the proportion of each bioregion within each of these protection categories. This enabled us to evaluate the change in the equality of protection offered to biodiversity over time.

Trend in management effectiveness, connectivity and pressures on the marine environment

Unfortunately, we were not able to identify the necessary data to populate the proposed indicators for management effectiveness, connectivity and pressures for the Australian MPA network. While some data were available on management effectiveness and selected pressures, these data were generally patchy and not available at the national scale. Therefore, these indicators were omitted from further analysis.

RESULTS

Changes to the number of MPAs and total area protected

Between 2002 and 2014, 107 new MPAs were established in Australia's waters, bringing the total number to 324 (262 MPAs and 62 marine components of terrestrial PAs). No new marine components of terrestrial PAs were added during the study period. More than 2.6 million km² were added, bringing the total area protected to 3.26 million km² (Figure 1). Forty-seven of the additions were small MPAs in coastal waters, accounting for approximately 2% of the area added since 2002. The vast majority of new area (98%) was the result of 60 large, offshore (3-200 NM from coastline; Vince et al. 2015) MPAs within Commonwealth waters, totalling 2.37 million km². These offshore MPAs now account for 71% (by area) of the total MPA network (Appendix A, Figure A1).

We identified very few changes to the size of existing MPAs, with only one MPA, Ningaloo Marine Park in Western Australia, being upsized (350 km² added in 2004) during the study period. No MPAs were degazetted or downsized in Australia between 2002 and 2014.

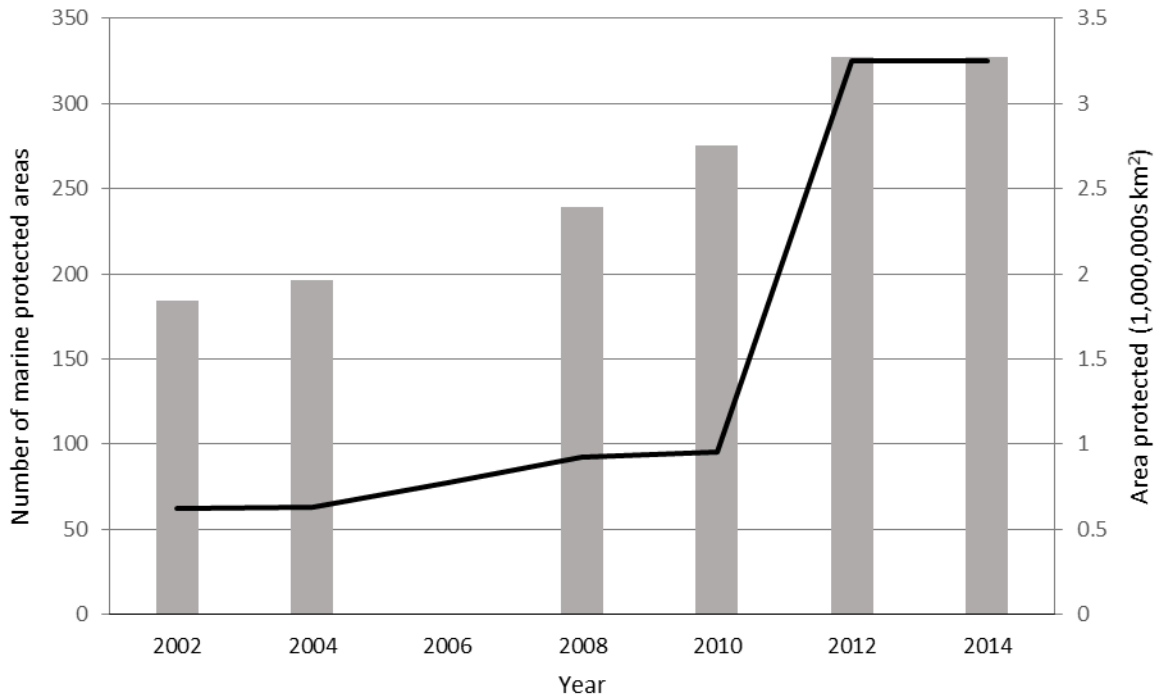


Figure 1. The area (black line) and number (grey bars) of MPAs in Australia from 2002-2014. No data were reported to CAPAD in 2006.

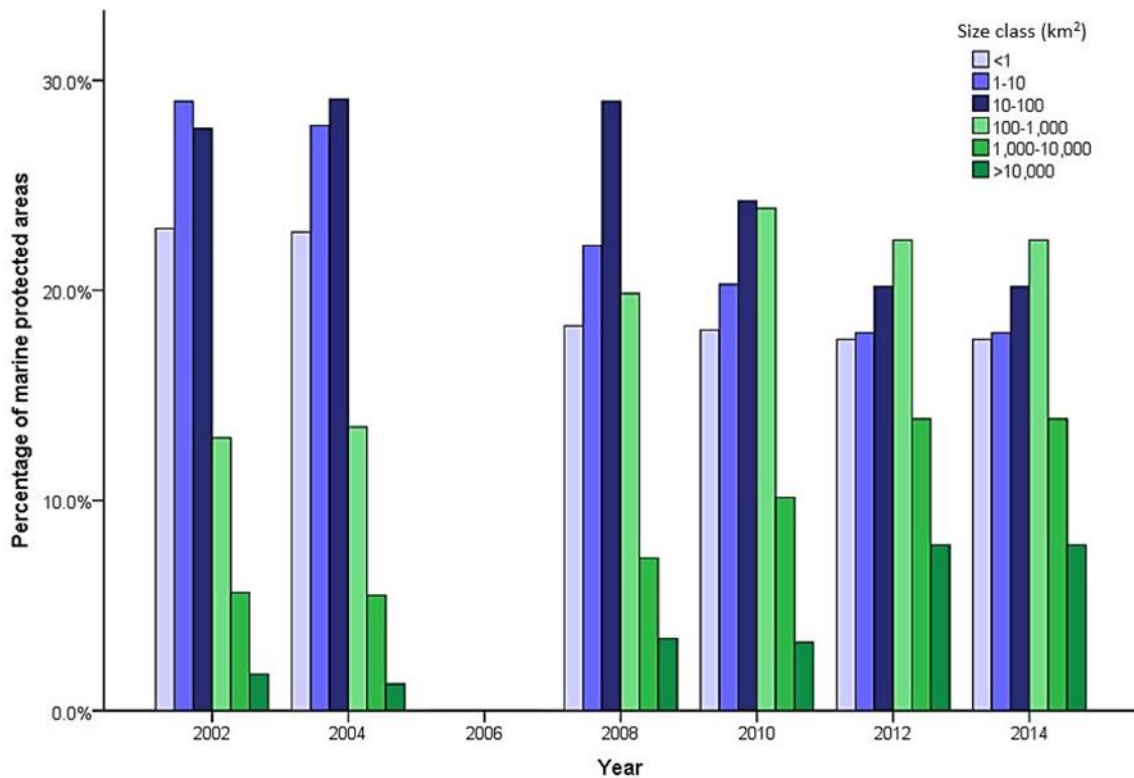


Figure 2. Size class distributions of MPAs in Australia over time. No data were reported to CAPAD in 2006.

Size distribution of MPAs

We found that over time, the size class distribution of MPAs shifted toward larger MPAs, with the greatest increases in the two largest size classes (Figure 2). The shift in size class distribution was driven by the 60 large, offshore MPAs. The overall median MPA size increased from 12.5 km² in 2002 to 46 km² in 2014, while the median size of offshore MPAs jumped from 665 km² to 3,725 km² between 2002 and 2014. By 2014, approximately 45% of the NSRMPA exceeded the definition of a large MPA (>100km²) associated with better biodiversity outcomes by Edgar et al. (2014).

Changes to the level of protection for marine biodiversity

The protection for Australian MPAs ranges from strict protection (Category IA and II) to multi-use areas (Category IV and VI) (Table 1). No Australian MPAs are classified as Category IB (wilderness areas) or Category III (natural monuments). The majority of the Australian NRSMPA is multi-use (65%), with the strongest growth in these categories occurring in 2012 (Figure 3) when the offshore MPAs were established. There has been a modest increase in the proportion of the MPA network made up of no-take areas (Categories IA and II); however, these areas comprise only about one third of the MPA network (Figure 3A), with the majority of this increase in Category II protected areas (Figure 3). There has been almost no growth in area within Category IA (strict nature reserve) (Figure 3A), such that this category has decreased as a proportion of the total network (Figure 3B). Using the strict interpretation of MPAs (as excluding all extractive uses), the Australian MPA network covers a total of 1.23 million km² out of the total 3.3 million km² network (15% of the EEZ) (Appendix A, Figures A2 & A3).

In total, 8 MPAs (affecting 110,388 km²) had their protection upgraded as a result of rezoning or small reserves being encompassed within the boundaries of new MPAs (Appendix A, Table A2). We found no evidence of policy changes that increased the protection of MPAs. A total of 33 MPAs (affecting 2,839 km²) had their protection downgraded during the study period, although 20 of these downgrades were subsequently reversed (Appendix A; Table A2). Downgrades occurred both as a result of formal rezoning and policy downgrades (Category IA sanctuary zones changed to permit recreational shore fishing).

Table 1. Designated zones across Australian MPA network with corresponding IUCN category labels and activities permitted within each IUCN category. ✓ Indicates activities permitted within the zone. ✓♦ indicates discrepancies between jurisdictions regarding whether the activity is permitted within the zone. See Tale A1 for full breakdown by jurisdiction of activities permitted in each zone.

ACTIVITY	Sanctuary Zone	Restricted Access Zone	Sanctuary Zone	Marine National Park Zone	Recreational Use Zone	Marine Park Zone	Habitat Protection Zone	Special Purpose Zone	General Use Zone
Assigned IUCN	IA	IA	II	II	IV	IV	IV	VI	VI
Scientific collecting/research*	✓	✓	✓	✓	✓	✓	✓	✓	✓
Anchoring a boat			✓	✓	✓	✓	✓	✓	✓
Boating			✓	✓	✓	✓	✓	✓	✓
Scuba/snorkel			✓	✓	✓	✓	✓	✓	✓
Commercial line fishing						✓	✓	✓	✓
Trapping						✓	✓	✓	✓
Longline fishing							✓♦		✓♦
Dropline fishing									✓♦
Spearfishing					✓♦	✓	✓	✓♦	✓
Recreational fishing					✓♦	✓	✓	✓	✓
Recreational collecting						✓	✓	✓♦	✓
Aquaculture and pearling							✓♦	✓♦	✓♦
Netting							✓♦	✓♦	✓♦
Trawling									✓♦
Mining operations*								✓♦	✓♦

*Permitted under permit

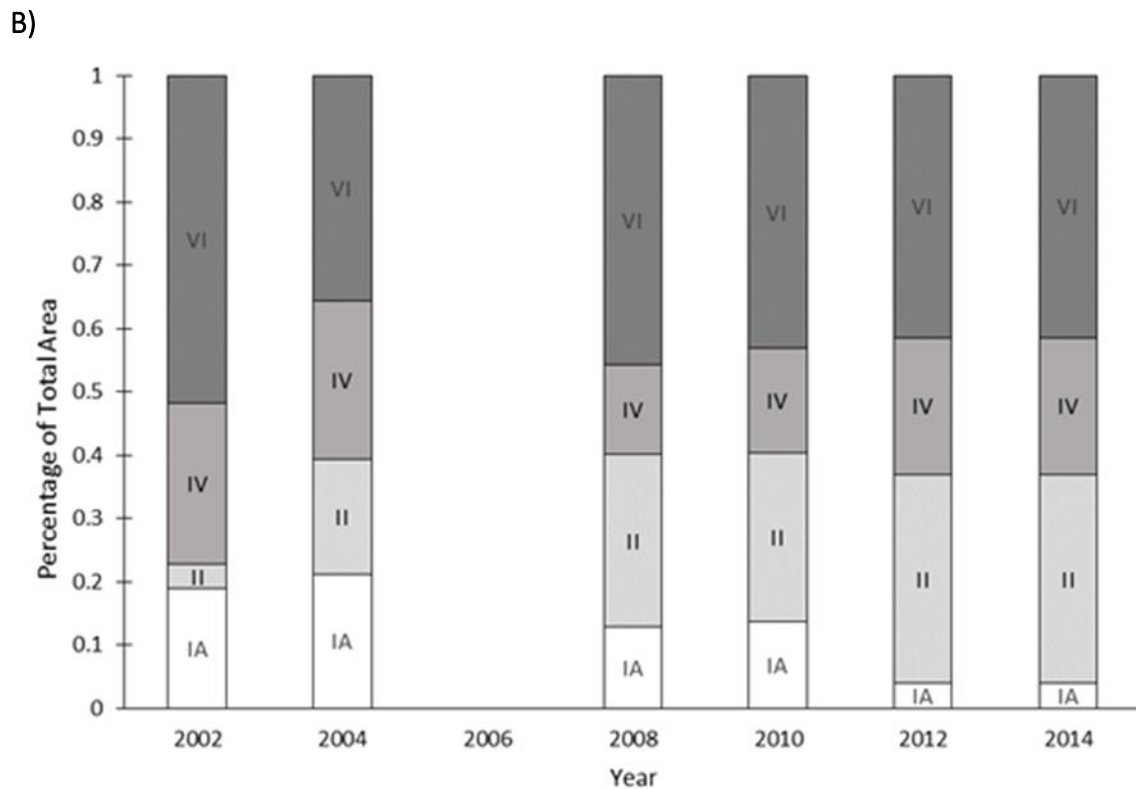
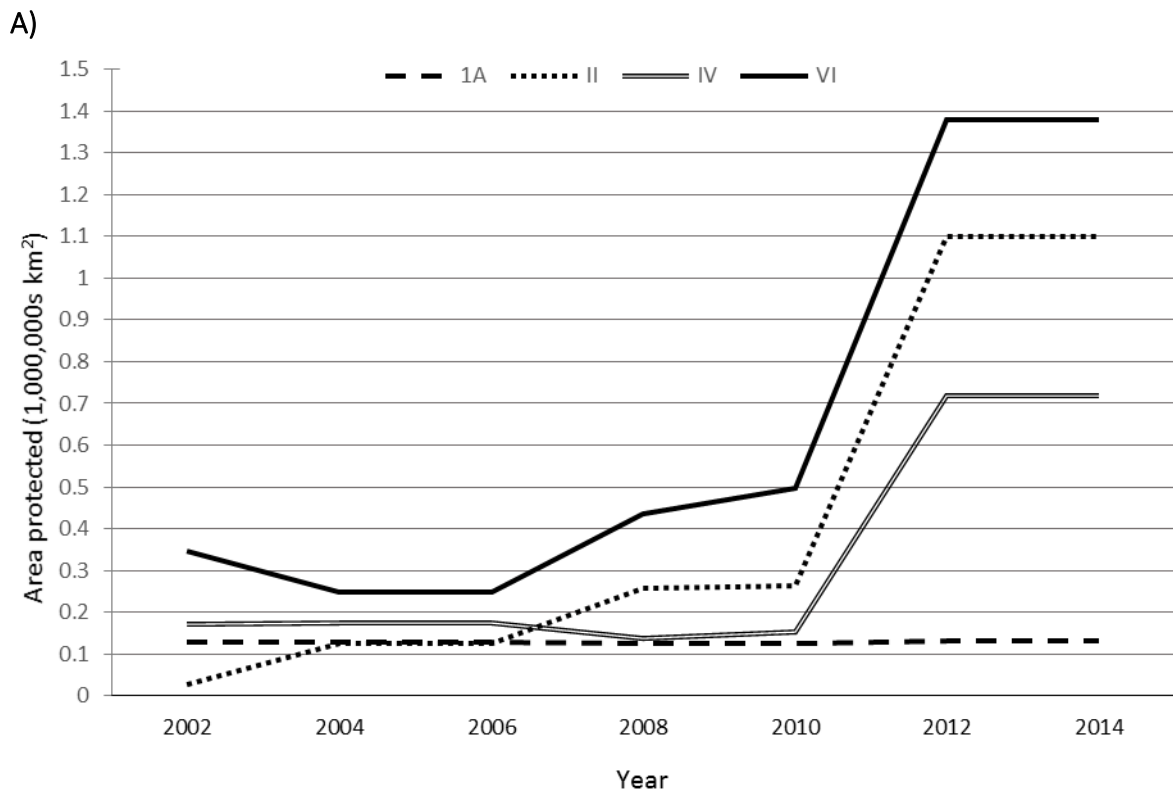


Figure 3. Change in the level of protection across the Australian MPA network between 2002-2014 presented as A) total area (millions of km²) of the network in each IUCN category over time; B) the proportion of the network within each IUCN category. No data were reported to CAPAD in 2006.

Change in the representation and level of protection of marine biodiversity

Growth in the Australian MPA network over time has resulted in a consistent trend of increasing the coverage of poorly represented bioregions (Figure 4A). Some progress was made in increasing the protection for poorly represented bioregions between 2002 and 2010, with the number of bioregions below the 10% threshold for adequate coverage reduced from 31 to 24 (Figure 4A). However, a major leap forward in achieving the CBD minimum 10% coverage target was achieved through the addition of the offshore MPAs in 2012, reducing the number of poorly represented bioregions to five (Figure 4A). These new MPAs also increased the protection for already well protected bioregions, moving eight bioregions to 50% or more of their area protected (Figure 4A). There is a distinct geographic bias in protection, with the most well-represented bioregions located around the Great Barrier Reef, off the north Queensland coast, and the poorest protected occurring in south-eastern Australia (Figure A4).

While most bioregions have reached the target of at least 10% of their area protected, the level of protection for these environments is not evenly distributed (Figure 4B & C). Much of the increase in representation occurred within multi-use zones, where extractive uses are permitted (Figure 4C). We found that there has been less progress achieving strict protection for bioregions, with the majority of bioregions (61%) having less than 10% of their area within no-take zones (Figure 4B). Therefore, while there has been significant improvement in the overall representation of bioregions (Figure 4A), only a fraction of this growth offered ecosystems protection from extractive uses.

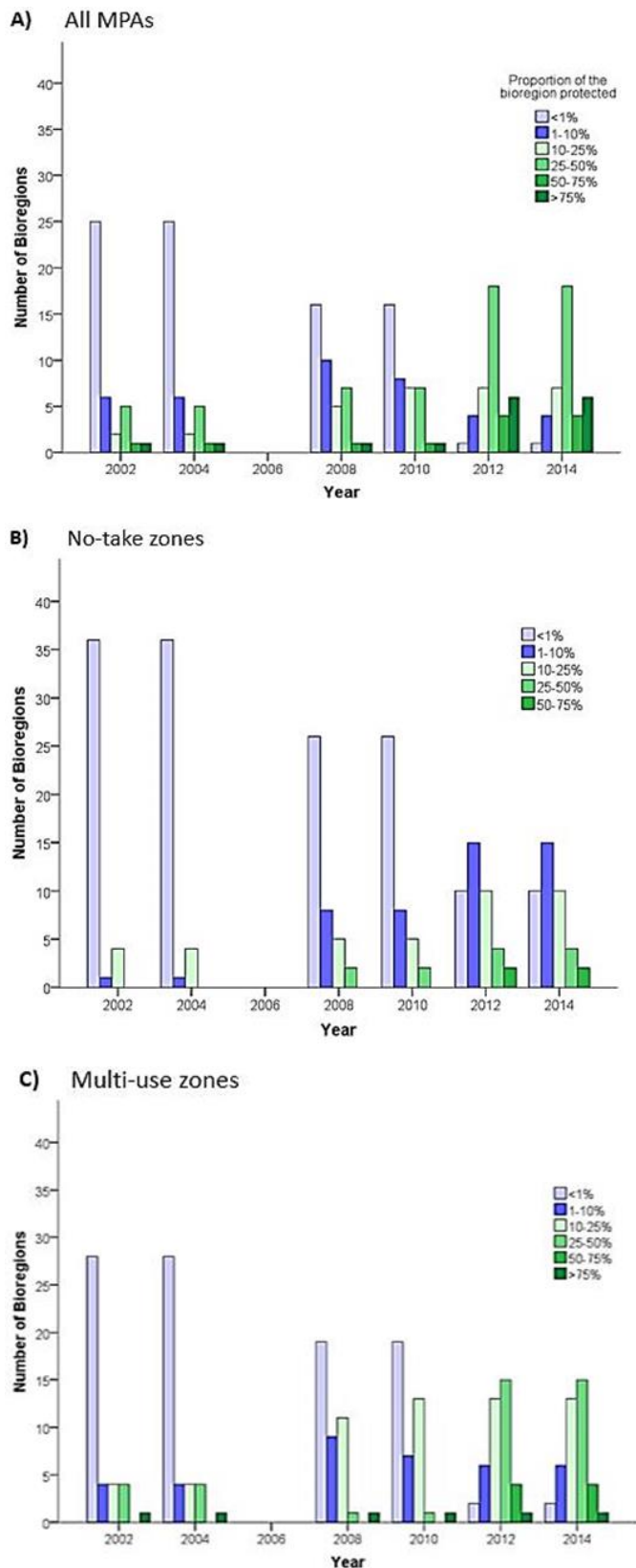


Figure 4. Change in the number of IMCRA Provincial Bioregions represented within the MPA network over time according to the proportion of their area protected for: A) all MPAs; B) no-take zones (IUCN Category IA & II); C) multi-use zones (IUCN Categories IV & VI). Purple bars indicate bioregions below the 10% threshold for adequate protection and green bars are above this threshold. No data were reported to CAPAD in 2006.

DISCUSSION

The total area under protection is frequently used to track conservation progress (Tittensor et al. 2014), despite being criticised as treating all PAs as equal (Collen and Nicholson 2014). Yet, as our data demonstrate, using the total area protected as the sole indicator can obscure trends in other important metrics of the strength of MPA networks, which provide a more detailed picture of how growth contributes to marine conservation. In particular, our data reveal that considering trends across indicators can be important to understand the structure of an MPA network, where major progress in increasing the area protected (Figure 1) and broad representation of biodiversity (Figure 4A) may be compromised if not accompanied with protection from key threats to biodiversity (Figure 3B).

Properties of effective MPAs

Effective MPAs are those where the size, placement and level of protection (Edgar et al. 2014; Gill et al. 2017) are well matched to the distribution of biodiversity and key threats to biodiversity (Devillers et al. 2015; Halpern et al. 2007). Given the interaction between these factors (e.g., high levels of protection in areas of low biodiversity and/or low threat offer little conservation value) it is essential that measures of progress in marine protection can evaluate these trends. The lack of data on pressures and the distribution of species and habitats, along with the effectiveness of MPA management, limited our ability to provide such a detailed evaluation of progress for the Australian NRSMPA, highlighting the difficulty obtaining sufficient data to genuinely measure progress in marine protection.

While the distribution and intensity of threats could not be quantified, fishing pressure has been identified as one of the greatest pressures on marine environments (Halpern et al. 2015) and is often concentrated in coastal waters, where vulnerable and commercially valuable biodiversity are disproportionately located (Klein et al. 2015). This would suggest that the trend in large, often strictly protected offshore MPAs and smaller, more disjointed coastal MPAs with very few no-take zones within Australian waters is not well-matched to the distribution of vulnerable biodiversity or key threats (i.e., residual reserves; Devillers et al. 2015). While large offshore MPAs can target important ecological assets (e.g. whale migratory route) or other habitats at high risk, our result highlights the importance of interrogating trends in the placement of MPAs to ensure large MPAs located in areas that

minimise opportunity costs are not used as a convenient way to reach protection targets with little benefit to biodiversity (Wilhelm et al. 2014). The indicators reveal that in order to strengthen protection for vulnerable biodiversity, future MPA coverage in Australia should be targeted towards large, no-take areas in coastal environments (Costello and Ballantine 2015; Edgar et al. 2014).

The vast majority of the Australian MPA network is designated as multi-use, permitting various forms of fishing. While some multi-use zones can protect against the most destructive forms of fishing, such as demersal trawling, they have specific objectives focused on recreational and commercial usage of marine biodiversity (Day et al. 2012). The variability in activities permitted in multi-use zones across Australian MPAs (Table 1) makes it difficult to estimate their value for biodiversity. Given the clear indications that multi-use MPAs have fewer ecological benefits (e.g., for reef fishes; Edgar et al. 2014), the significant proportion of MPAs within these lower protection categories raise questions about the long-term capacity of the Australian NRSMPA to conserve marine biodiversity. Nevertheless, given other indicators suggest progress has been made in representation and size class distribution, rezoning exercises could be used to improve the equity of protection for biodiversity to significantly strengthen the Australian MPA network. However, it should be acknowledged that rezoning exercises can lead to downgrades in protection, such as those currently proposed for many of Australia's large, offshore MPAs (reducing existing no-take zones by 40-50%; Director of National Parks 2017). These findings demonstrate the value of a more nuanced set of indicators for identifying opportunities to strengthen the value of MPAs without necessarily increasing the area protected.

Our spatially explicit approach to quantifying change in the level of protection for MPAs revealed the vulnerability of these areas to changes in political ideology, as seen elsewhere in the world (e.g., Bernard et al. 2014). The examples we documented of downgrades in protection, some reversed after a change in government, reflect the global tension between protection for the intrinsic value of biodiversity versus a more utilitarian perspective on the value of the natural environment (Eagles 2002). Changes in protection for MPAs can make enforcement difficult (Halpern et al. 2010) and contribute to decreased public support of MPAs in general (Voyer et al. 2012). Despite some observed changes in protection level, Australian MPAs have been more stable than terrestrial PAs in Australia, where over 1,500

PADDD events have been documented impacting one third of the network (Cook et al. 2017). This relative stability may reflect the challenge of negotiating the establishment of MPAs, given conflicts with many stakeholder groups (Ruiz-Frau et al. 2015), but emphasises the need to ensure that systematic conservation planning principles, and evidence for the attributes of effective MPAs, are taken into account at the time of establishment, as there may be little political will or scope for change in the future.

Properties of MPA networks

Ultimately, MPAs should be designed to function as an integrated network to protect biodiversity. While ideally MPA networks provide adequate representation of species and ecosystems (Sutcliffe et al. 2015), the lack of habitat mapping data for marine environments, means representation must often be judged on the basis of coarse ecoregion classifications (e.g., IMCRA). While habitat mapping data in Australia may be better than that of many other countries, data are still patchy and not available network-wide. Our longitudinal perspective of change clearly demonstrates the success of systematic conservation planning to achieve extensive progress toward minimum protection targets (10%) for bioregions (Figure 4). Nevertheless, there are inherent risks associated with planning MPAs based on coarse ecoregions rather than finer scale data on the distribution of species and ecosystems, as can be achieved for terrestrial PAs (e.g., representation of threatened species, ecosystems and key biodiversity areas; Butchart et al. 2015; Venter et al. 2014). Likewise, the focus of impact evaluation studies on exploited fishes (e.g., Edgar et al. 2014) leaves gaps in our understanding of how well these species represent the response of a broader range of biodiversity. There is clearly a need to capture biodiversity data across a wide range of habitats and taxonomic groups to ensure measures of progress provide the most comprehensive estimates possible.

Given studies have found that enforcement of protections is critical to positive biodiversity outcomes from MPAs (Edgar et al. 2014), the lack of management effectiveness data leaves a significant gap in our ability to determine whether legislative protection translates to genuine protection for biodiversity. Importantly, small, well-enforced MPAs may ultimately be more effective at conserving important biodiversity if resources are not provided to enforce larger areas (Gill et al. 2017), making management effectiveness data a critical element of interpreting trends in the structure of MPA networks.

In addition to gaps in biodiversity, pressure and management data, it was not feasible to assess the level of connectivity of MPAs, which is necessary to determine whether they function as a network to support the ecological and evolutionary processes required for persistence (Krueck et al. 2017). Therefore, while our study reveals the value of evaluating a more nuanced set of progress indicators, it also reveals the need to fill critical gaps in our ability to assess the strength of marine protection. Future research should also consider the social and political factors that may influence the capacity to strengthen existing MPA networks, particularly given evidence that governments are not prioritising management of MPAs (Addison et al. 2015). Nevertheless, our findings reveal that assessing both individual and network-wide metrics of effective MPAs can reveal simple changes (e.g., increasing the level and equity of protection) to existing MPAs and strategic expansion of MPAs (e.g., within heavily exploited environments; Halpern et al. 2015; Klein et al. 2015) that could significantly strengthen of the network.

Conclusion

Our systematic assessment of change in an MPA network demonstrates the value of a broader set of metrics to evaluate progress in marine protection, going beyond total area protected. Collectively, these metrics can track progress in the development of a MPA network and help guide efforts towards strengthening conservation of marine biodiversity. Encouragingly, the application of these metrics to the Australian NRSMPA revealed opportunities to build on the strong progress towards a robust system of MPAs by extending no-take areas. However, it is critical that governments invest in improving data on the distribution of biodiversity, threats, connectivity and management effectiveness to provide the most meaningful assessment of progress in marine conservation.

Chapter 3

BIO-PHYSICAL MODELS OF MARINE ENVIRONMENTS REVEAL BIASES IN THE REPRESENTATION OF PROTECTED AREAS

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ABSTRACT

The significant shortfall in global marine protection targets is likely to continue to drive rapid growth in marine protected areas (MPAs). Systematic conservation planning to fill gaps in marine protection requires sufficient knowledge of both the distribution of biodiversity and the threats to species and ecosystems. Yet such data are lacking for much of the marine environment, creating significant challenges for planning effective marine protection. In the absence of habitat mapping data, critical environmental variables associated with species' distributions can be used to model the spatial distribution of different environments. While this approach has been used in some jurisdictions to assist MPA planners, the increased availability and resolution of spatial data now provide an opportunity to improve assessments of MPA representation. Capitalizing on advances in spatial data, this study uses a range of biological and physical environmental attributes to model the distribution of Australian marine environments. Given many Australian MPAs were implemented without knowledge of the distribution of species and benthic habitats, this bio-physical model is used to assess MPA coverage and equality of protection for Australian marine environments. Results of the bio-physical model revealed that Australian MPAs over-represent warm, offshore waters (such as the Coral Sea) and under represent temperate environments. Furthermore, the distribution of protection in Australian MPAs is heavily skewed, with no-take protection disproportionately targeting tropical environments, leaving major gaps in the protection of both temperate and nearshore habitats. Without comprehensive habitat mapping, the representativeness and adequacy of a MPA system cannot be accurately evaluated, nor can the required expansion of MPAs be planned effectively. In the interim, the biological and physical attributes chosen for this model provide useful proxies to assist in efforts to better target current and future protection based on the most up-to-date knowledge.

INTRODUCTION

A key conservation strategy to protect and manage marine biodiversity is the implementation of marine protected areas (MPAs; Klein et al. 2015). MPA regulations vary greatly, and accordingly the level of protection awarded to the marine environment is not uniform. Nevertheless, significant benefits, such as increased fish biomass (Edgar et al. 2014; Halpern et al. 2009), spill-over effects into unprotected sites (Campbell et al. 2017), and reduced impacts of invasive species (McCook et al. 2010), have all been linked to MPAs around the world. In recognition of the important role MPAs play in the conservation of marine ecosystems, targets for protection have been set under the Convention on Biological Diversity (CBD), and recently integrated into Sustainable Development Goals (SDG 14.5), to systematically expand the area under protection to 10% of all coastal and marine waters (Secretariat of the CBD 2011; Diz et al. 2017). These targets recognize the need for an effectively managed, ecologically connected, and representative system of MPAs where protection is targeted at vulnerable biodiversity. Despite the recent surge in MPA declaration, just over 5% of the global ocean is currently protected, leaving significant gaps in protection for many ecosystems (UN Economic and Social Council 2017). The majority of this growth is in large, offshore MPAs, which contribute substantially to reaching global targets for MPA coverage but too often provide minimal value to biodiversity conservation (Devillers et al. 2015; Roberts et al. 2018).

Systematic conservation planning outlines a range of design principles to maximise biodiversity outcomes (Ban et al. 2014; Margules and Pressey, 2000). The basis for systematic conservation planning is knowledge of both the distribution of biodiversity, and ideally, the threats to species and ecosystems. On land, this is generally achievable as the distributions of major ecosystems and species have largely been mapped (UNEP-WCMC and IUCN 2016), along with the spatial distribution of multiple threats (Human Pressure Index; Geldmann et al. 2014). While comparable data are lacking for much of the marine environment, surrogates for the distribution of marine ecosystems can greatly assist systematic conservation planning effort to evaluate the representation of existing protection and prioritize new MPAs (Gleason et al. 2006; Klein et al. 2015). Previous research has also demonstrated the importance of moving beyond the representation element of international targets to also evaluate the degree of connectivity within a MPA system (e.g. the degree to which MPAs function as a

network), and which areas should be targeted to improve ecological connectivity (Magris et al. 2017). While such assessments are critical to the effective design of the global network of MPAs, they are currently lacking for much of the world's oceans.

In the absence of data on the distribution of biodiversity, biological and physical environmental attributes (hereafter referred to collectively as bio-physical attributes), such as temperature, bathymetry and primary productivity, can be used as surrogates to assist with conservation planning due to their broad geographic coverage and strong correlation with species distributions (Briones et al. 2009). Numerous global biogeographic classifications, such as the Marine Ecosystems of the World (MEOW; Spalding et al. 2007) and the Global Open Oceans and Deep Seabed (GOODS; Briones et al. 2009) classifications, have been developed using spatially explicit surrogate variables to delineate major ecosystems in the ocean. The Essential Ocean Variables (EOV) framework also relies on physical and biogeochemical surrogate variables to provide a cost-effective and feasible method of monitoring biological patterns in marine systems on a global scale (Miloslavich et al. 2018). While marine ecosystems can be studied using countless oceanic variables, previous research demonstrates the relevance of a few key surrogate variables (i.e. temperature, depth, salinity, and benthic substrate) to aid in the development of biogeographic classifications (Constable et al. 2016; Howell 2010; Miloslavich et al. 2018).

Surrogate variables map environmental conditions associated with the limits to species distributions (e.g. thermal minima and maxima, depth, etc.) on the assumption that they will help define different ecosystems (Lewandowski et al. 2010). While biotic interactions are important in determining the distribution of biodiversity, often abiotic factors can help explain much of the variation in community assemblages (McHenry et al. 2017). For example, the distribution of terrestrial communities can generally be predicted based on the combination of soil type, rainfall, slope, aspect, and elevation (Stephenson 1998). Likewise, in marine environments, while most species have natural adaptive features, a range of bio-physical attributes have a strong impact on where marine organisms occur (Witman et al. 2008). For example, high levels of species diversity and complexity are found in shallow, coral reef environments due to the high levels of primary productivity produced by these ecosystems (Rogers et al. 2015). Additionally, levels of species diversity will vary significantly according to benthic substrate, with harder substrates often associated with higher diversity

(i.e. rocky or coral reef) due to a greater habitat complexity and stability than softer substrates (Piacenza et al. 2015). Combining multiple variables known to be associated with species distribution into a bio-physical model can provide a detailed assessment of how well a MPA system (a collection of MPAs within a jurisdiction) represents the spectrum of marine ecosystems, and whether protection is well matched to areas of high biodiversity.

Australia declared its commitment to conserve marine ecosystems through the establishment of the National Representative System of Marine Protected Areas (NRSMPA) in 1991, which has evolved into the second largest MPA system in the world (UNEP-WCMC and IUCN 2016). While Australia's progress beyond the CBD 10% target is praiseworthy, many MPAs were implemented despite significant knowledge gaps about the distribution of species assemblages, benthic habitats, threats to marine biodiversity or connectivity among sites, limiting the ability to identify areas that would benefit most from MPA protection (Lawrence et al. 2015). In 2012, Australia established a series of Commonwealth Marine Reserves (CMRs), which significantly increased protection for 24 poorly represented broad scale bioregions (Roberts et al. 2018). Excluding the Great Barrier Reef Marine Park, habitat mapping data (defined here as mapping the distribution and extent of communities to obtain complete coverage of the seabed) were generally not available, or not in a comparable format, to inform the design and placement of the CMRs (Lawrence et al. 2015). To date, only about 12% of the habitats within the CMRs have been mapped (Lawrence et al. 2015). Given the lack of habitat mapping data, the design and planning process for MPAs is heavily reliant on surrogate measures for the distribution of biodiversity (e.g. topographic or ecological features; Sutcliffe et al. 2015; Williams et al. 2009), as was the case for the design and planning of the CMRs. While these surrogate measures fulfilled a much-needed role in the absence of national habitat mapping data, the increased availability of spatial data on important bio-physical attributes of the marine environment (e.g. Bio-ORACLE; Assis et al. 2017) provides an opportunity to model a suite of fundamental environmental attributes, based on the best available evidence for the factors that influence the distribution of biodiversity.

Without comprehensive data on the diversity and distribution of marine biodiversity, the performance of MPAs in protecting the marine environment cannot be measured effectively. The current study uses the latest available spatial data to construct a bio-physical model of

the distribution of Australian marine environments to understand MPA system coverage and the equality of protection for those environments. This modelling approach provides a significant advance on the approach used to plan the Australian NRSMPA (i.e. Integrated Marine and Coastal Regionalisation of Australia: IMCRA; Commonwealth of Australia 2006), using a broader set of variables and finer scale data for a core set of biological and physical parameters from IMCRA (Appendix B, Table B2), along with a transparent methodology. The global availability of these data means that, in the absence of detailed habitat mapping data, this approach can provide a valuable spatial planning tool for MPA planners around the world. With sufficient knowledge of the distribution of vulnerable biodiversity, the gaps in protection for marine biodiversity can be identified and associated with the necessary levels of protection to mitigate important impacts on the marine environment.

METHODS

Study area

The Australian Exclusive Economic Zone (EEZ) is one of the largest in the world at approximately 9 million km², with an additional 2 million km² of Antarctic Territory waters (Butler et al. 2010). The responsibility for managing the EEZ is divided between state (shoreline to 3 NM) and federal governments (3 to 200 NM) (Vince et al. 2015). Each jurisdiction has legislation to regulate the use of the marine environment, including fisheries management and declaring and managing MPAs (Boer and Gruber 2010). With eight jurisdictions, each with their own governance structures, the term MPA in Australia encompasses a wide spectrum of reserve types, with varied management objectives and protection zones offering strict protection for biodiversity through to multi-use areas that allow natural resource extraction (Boer and Gruber 2010).

Australia's marine environment is home to a wide range of unique species, with especially high levels of endemism in temperate waters (Butler et al. 2010). Much of Australia's marine life is now recognised as threatened due to anthropogenic impacts, such as overfishing, pollution, and climate change (Evans et al. 2017). Despite the area covered by MPAs in Australia having more than quadrupled in the past decade (Roberts et al. 2018), Australia, like many other countries, lacks comprehensive habitat mapping for its marine environment (Lawrence et al. 2015). While habitat mapping was completed for some MPAs after their

establishment (i.e. South Australia), mapping has generally not been conducted for the broader marine environment outside the MPAs. Therefore, large swaths of the Australian coastline and offshore waters have no mapping of their habitat characteristics, raising questions and concerns about how well Australian MPAs represent marine habitats.

The IMCRA classification was developed in 1998 as a surrogate for habitat mapping and formed the primary spatial framework utilized in the design and planning process for the CMRs (Barr and Possingham 2013). IMCRA classifies the Australian EEZ into broad bioregions based on a range of environmental attributes, such as geomorphic features and climate characteristics, along with the biogeography of over 5000 demersal fish species (Commonwealth of Australia 2006; Last et al. 2005). Various statistical and graphing methods, alongside expert opinion, were used to identify the boundaries of the bioregions, although the criteria and process used were not explicitly outlined (Commonwealth of Australia 2006). IMCRA was based on the most comprehensive data available at the time to inform MPA planning for the representation of marine biodiversity (Devillers et al. 2015).

Bio-physical attributes

An extensive literature review was completed, along with consultation with experts in the Australian marine environment, to identify the key bio-physical attributes associated with the distribution of Australian marine species. The attributes most commonly reported to be important in the distribution of marine biodiversity were: temperature (generally measured as sea surface temperature; SST), salinity (generally measured as sea surface salinity; SSS), bathymetry (depth), primary productivity (Chlorophyll a), and benthic substrate. These five attributes, which align with the bio-physical attributes of the global EOVs (FOO 2012), have been used to accurately identify marine habitat distribution on multiple scales (McArthur et al. 2010; Riginos et al. 2016), and most marine species exist within an optimal range of each of these attributes. Therefore, it was important to individually assess the distribution of these attributes throughout the Australian EEZ, as well as collectively, through the development of a bio-physical model.

Sea surface temperature or temperature at depth (i.e. benthic or water column) are important to include because a species' thermal tolerance will limit their distribution to certain water temperatures, and their abundance will be greatest within thermal optima

(Sunday et al. 2012). Likewise, marine species vary in their biological tolerance to salinity, and the presence of a salinity gradient can be critical to the life cycle of many fish and invertebrate species (Fenchel and Finlay 2004; Larson and Belovsky 2013). Temperature, sunlight penetration and levels of hydrostatic pressure vary with depth, so bathymetry and the variability in bathymetry (i.e. rugosity) can be important determinants of species distribution and abundance (Smith and Brown 2002). While the general patterns of primary productivity for the global ocean varies seasonally, tropical, shallow water systems tend to be nutrient poor in comparison with cold, deeper waters (Woolley et al. 2016). The benthic substrate often defines the types of marine species that are dominant in a community (Sebens 1991). Hard substrate communities, such as reefs, have been found to have higher diversity than soft sediment habitats, although the latter are by far the most common marine environments on earth (Taylor and Wilson 2003; Wilson 1990). While the distribution of biodiversity could be evaluated based on an endless list of proxy variables, this analysis was limited to the five aforementioned attributes due to the evidence for their relationship with species distributions, broad scale data availability and computational efficiency. Future analyses could expand on this core set of commonly used surrogate attributes to explore the influence of additional variables, such as ocean surface heat flux or current velocity.

Spatial layers were identified for each of the attributes. Mean annual SST and SSS were derived from NOAA's World Ocean Atlas and downloaded at a 1 km² spatial resolution from MARSPEC (Sbrocco and Barber 2013). Mean annual benthic temperature (temperature at max depth) was retrieved from Bio-ORACLE at a 9.2 km² spatial resolution (Assis et al. 2017; Tyberghein et al. 2012). Mean annual primary productivity was retrieved from NASA's Global Ocean Color website and downloaded at a 1 km² spatial resolution (<https://oceancolor.gsfc.nasa.gov/data/aqua/>). Bathymetry data for the Australian EEZ were sourced from Geoscience Australia at a 250 meter spatial resolution (Whiteway 2009). No national dataset is available for benthic substrate, so data were sourced directly from each of the state, territory and federal agencies. Due to inconsistent classifications between jurisdictions, the data were standardized using an ordinal scale of softest to hardest substrate so it could be incorporated into the bio-physical model (Appendix B, Table B1). Layers for all attributes were imported into ArcGIS as raster data, resampled to a 1 km² spatial resolution and clipped to the Australian EEZ, which yielded 6,149,339 observations. Spatial joins were

performed in ArcGIS to extract the bio-physical attributes of each unique 1 km² cell. Grid cells were designated as MPA or non-MPA using boundaries from the Collaborative Australian Protected Area Database (CAPAD 2016). When a single grid cell was split between a MPA and non-protected area, a 60% threshold value was used to designate the cell as protected.

Although the Australian EEZ encompasses Antarctic territory and several remote, sub-Antarctic offshore islands (i.e. Heard and McDonald Islands, Macquarie Island), insufficient data were available to incorporate these areas into the model, excluding 233,000 km² of largely no-take MPAs from the analyses. Considering these areas are sub-Antarctic and Antarctic environments, they are expected to be very different to the rest of the EEZ and their remoteness is likely to mean they are under little pressure. Ideally, as they are a significant portion of the Australian MPA system, future analyses would include these areas. Hereafter, reference to the EEZ does not include Heard, McDonald and Macquarie Islands, or the Australian Antarctic Territory, unless explicitly stated.

Statistical analysis

Prior to further analysis, the bio-physical attributes were examined for collinearity using Pearson's correlation coefficient. A strong correlation was present between benthic temperature and bathymetry (0.79). Therefore, benthic temperature was excluded from all analyses as bathymetry displayed only a moderate correlation with SST (0.345). While moderate correlations were seen between several variables (i.e. bathymetry and substrate; Appendix B, Table B3), all bio-physical attributes incorporated into the model were validated in multiple ways to determine if each provided a genuine contribution to the model output (see Model validation section).

Univariate approach

The distribution of each of the bio-physical attributes was examined around the Australian EEZ. Generalized linear models were used to assess whether protection status (MPA or not) was distributed evenly relative to: (a) sea surface temperature (square transformation; Gaussian distribution, identity link), (b) sea surface salinity (Gaussian distribution, identity link), (c) depth (Negative binomial distribution, log link), and (d) chlorophyll a (Poisson distribution, log link). The distribution of protection (MPA or not) across different types of substrate was assessed using a non-parametric Mann-Whitney U-test due to ordinal data.

Given the large sample size (n=6,149,339), effect sizes were calculated for all significant differences. Due to the skewed distribution of bio-physical attribute data, medians were selected as the preferred descriptive statistic for central tendency of these data (though means were also calculated as a complementary measure). For the parametric tests, effect sizes were calculated as the standardized mean difference. For the non-parametric tests, effect size was calculated as an additional parameter of the Mann-Whitney U-test.

Multivariate approach

To understand how the different bio-physical attributes vary in relation to one another, a principal components analysis (PCA) was used to reduce the five attributes to a smaller number of dimensions in order to generate an uncorrelated multivariate state space. PCA is also well suited to situations where there is some collinearity among variables (Vaughan and Ormerod 2005). While PCA assumes all of the variables are normally distributed, it is generally considered robust to violating that assumption for some of the variables (Jolliffe 2002; see 2.4 Model validation below). The resulting principal components (PC) were used to assess whether MPAs represent the spectrum of environments in the EEZ using a Mann-Whitney U test. The level of protection within MPAs (based on data from CAPAD 2016) was also assessed to determine how protection (i.e., the IUCN category associated with each protection zone) was distributed among different environmental conditions using a Kruskal-Wallis H test. All unprotected areas, along with IUCN categories III (natural monument) and V (protected seascape), which are rarely used to classify MPAs in Australia, were omitted from this analysis. Finally, as an alternative technique to visualize differences and patterns in the data, the principal components were integrated into a k-means cluster analysis and displayed as a bi-plot with a pre-determined selection of four clusters (Appendix B, Figure B1). MPA locations per cluster were highlighted as an additional method of evaluating MPA representation of Australian marine environments. Uni- and multi-variate analyses were conducted in R (R Core Team 2017).

Model validation

The bio-physical model was validated in three separate ways. First, a random forest model was generated to determine the relative importance of all attributes to the overall model. This analysis was necessary due to moderate correlations between several of the bio-physical

attributes (Appendix B, Table B3). Random forest analysis is a powerful, model-averaging technique that provides a relative importance value for each variable to determine whether they genuinely contribute to the model through comparison with randomly generated numbers (Johnstone et al. 2014). The random forest was created by generating 1,000 trees from a randomly selected sample of 10% of the full dataset (i.e. 600,000 records) with an additional three columns of randomly generated numbers. To ensure confidence in the values for the relative importance of each attribute, results were compared for three separate random forest runs with different random numbers.

Second, spatial layers for PC1 and PC2 were generated using the principal components tool in ArcGIS to produce a single, multiband raster layer for each of the two components (Figure 1). While coverage of habitat mapping is patchy, some habitat types, such as seagrass, mangrove, and some algae communities, have fine scale resolution mapping across multiple jurisdictions. Therefore, the spatial layers for PC1 and PC2 were overlaid with available habitat mapping data to assess whether the bio-physical model was consistent with the distribution of mapped habitats around Australia.

Lastly, PC values were compared across the different IMCRA provincial bioregions to determine if the model was broadly consistent with this coarser evaluation of the marine environment. For each bioregion, the mean values of PC1 and PC2 for both MPA and non-MPA cells were plotted and used to visually assess whether the bio-physical model could broadly distinguish between IMCRA bioregions.

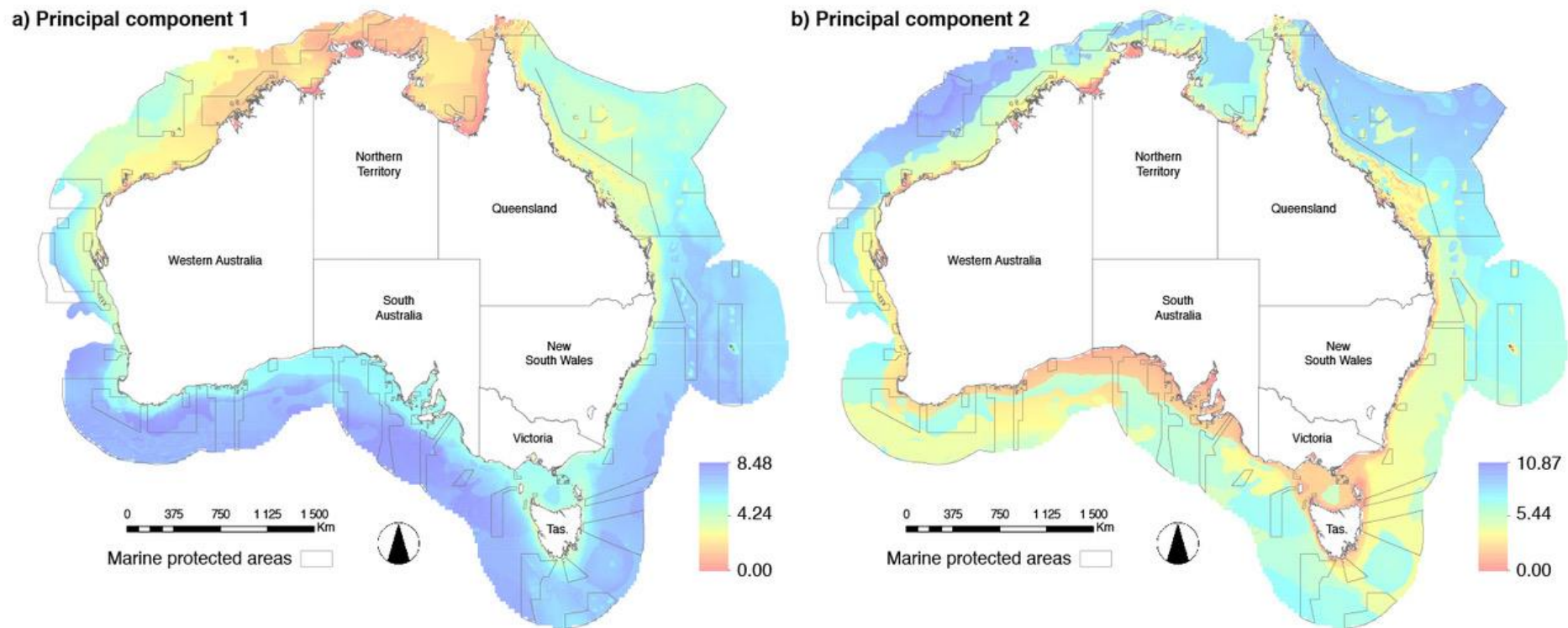


Figure 1. Spatial distribution of A) principal component 1 (PC1) and B) principal component 2 (PC2) using the principal components tool in ArcGIS. MPA boundaries are outlined in black.

RESULTS

The Australian marine environment

Australia's marine waters span tropical and temperate regions, with a median sea surface temperature of 17.4 °C (Table 1). Bathymetry is characterized by shallow inshore waters along the continental shelf, but most of the EEZ is deep water (median depth of 2,000 m) and can be as deep as 6,000 m on the abyssal plain (Table 1). Substrates around Australia range from silt and mud to rocky reef but are predominantly soft sediments (Table 1). Much of Australia's EEZ is unproductive waters (median chlorophyll *a* of 0.2 mg m⁻³), although chlorophyll values range from 0.1 to 9.2 mg m³ (Table 1). There is a modest salinity gradient across the EEZ, with sea surface salinity ranging between 33.9 and 36.2 PSU (Table 1).

When examining the distribution of the bio-physical attributes relative to one another, the PCA analysis revealed that two principal components explain approximately 66% of the variation among the predictor variables across the EEZ (39.9% for PC1 and 25.9% for PC2; Figure 2). The factor loading values show a strong positive correlation between values of PC1 and SST, and a strong negative correlation with SSS. There is also a modest negative correlation between PC1 and bathymetry. These loadings suggest that larger values of PC1 represent warmer, less saline and shallower water environments, while smaller values represented cooler, more saline and deeper water environments. These results would concord with PC1 representing an approximate separation between tropical and temperate waters.

PC2 showed a strong positive correlation with productivity and substrate, and a modest negative correlation with depth. Therefore, larger values of PC2 represent more productive, shallow water environments with harder benthic substrates, while smaller values are less productive, softer sediment environments in deeper waters. This component would be consistent with a separation between subtidal reef environments and the deeper, offshore environments.

The distribution of values of PC1 for the EEZ span tropical and temperate environments, with a slightly greater abundance of cooler, saltier, deeper water environments (Figure 4F). The distribution of values of PC2 suggest that while the EEZ has some highly productive, reef environments, much of the waters are deep, unproductive soft sediment areas (Figure 4F).

This concurs with what is known about the Australian marine environment, where the harder substrates on the continental shelf are a small proportion of the area relative to the offshore waters. When the two components are plotted against one another, a clear distinction is evident between the less abundant temperate and tropical reef environments (top half of Figure 3B), and the predominance of deeper soft sediment environments (bottom half of Figure 3B), and the abundant warmer, nutrient-poor waters (left half of Figure 3B) relative to the cooler, more productive waters (right half of Figure 3B).

Table 1. Descriptive statistics of the five Australian seascape attributes chosen for the bio-physical model: Sea surface temperature (SST), Sea surface salinity (SSS), Substrate, Chlorophyll *a* (Chl_a), and Depth

	SST	SSS	Substrate	Chl_a	Depth
Mean	16.99671	35.20421	2.87	0.314933	-2269.06
Std. Error of mean	0.001411	0.000174	0.001	0.000186	0.793
Median	17.4143	35.28	2	0.208469	-1999
Std. Deviation	3.499433	0.431348	1.806	0.461008	1966.704
Minimum	8.832801	33.97	1	0.072753	-6426
Maximum	23.9903	36.27	8	9.225864	0

MPA representation of the marine environment

When considering MPAs, significant differences were found between how protection is distributed relative to the bio-physical attributes of the Australian EEZ. In general, MPAs span a similar distribution of values for each of the bio-physical attributes but tend to over-represent some environments, such as warmer water environments, relative to the rest of the EEZ ($\chi^2=268715.43$, $df=1$, $p<0.001$, $d=0.43$; Figure 4A), probably due to the influence of the Great Barrier Reef Marine Park. They also tend to be in deeper ($\chi^2=1930.82$, $df=1$, $p<0.001$, $d=0.04$; Figure 4C), less productive ($\chi^2=4083762.80$, $df=1$, $p<0.001$, $d=0.2$; Figure 4B) waters with softer substrates ($U=5.13 \times 10^{12}$, $p<0.001$, $d=0.2$; Figure 4E), which likely reflect the size of the 40, offshore Commonwealth Marine Reserves established in 2012. While

MPAs tend to be in slightly less saline waters ($\chi^2=8129.96$, $df=1$, $p<0.001$, $d=0.1$; Figure 4D), this is only a minor deviation from the rest of the EEZ.

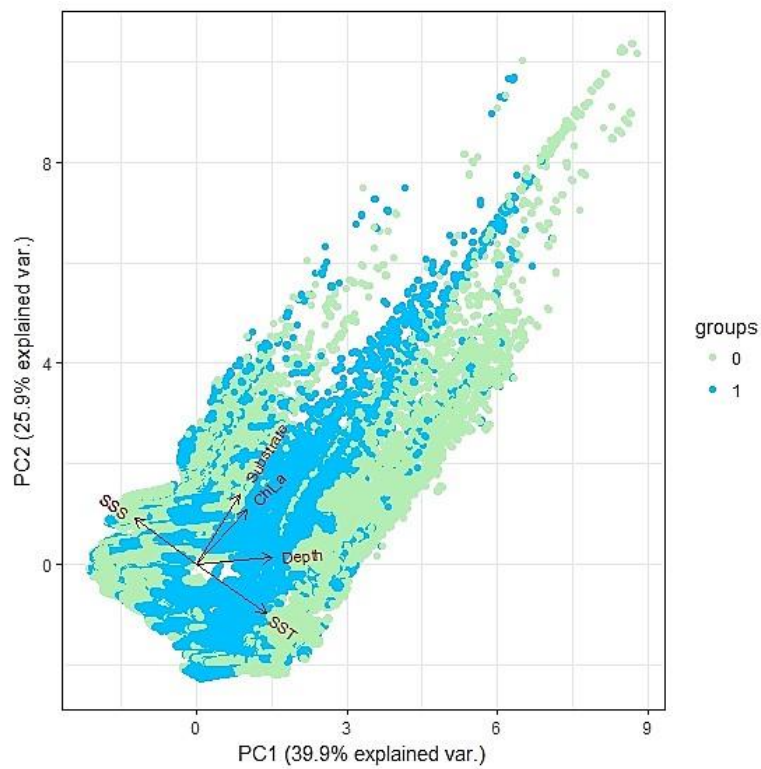


Figure 2. Biplot indicating PCA-based loadings of Australian seascape attributes chosen for the bio-physical model from all grid cells colour-coated by MPA presence (blue dots) or absence (green dots).

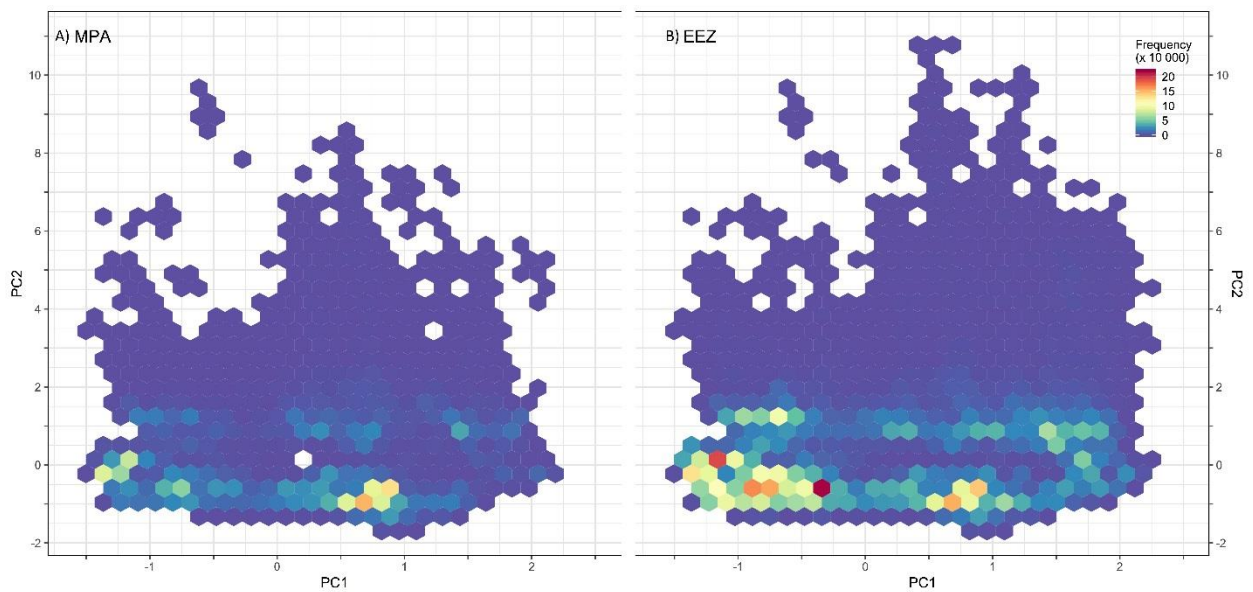


Figure 3. Density plot for A) MPA-only grid cells and B) the whole EEZ showing the frequency distribution of principal component 1 (PC1) and principal component 2 (PC2).

While the PCA suggests that MPAs in Australia span the distribution of component values (Figure 3A), there are significant differences in the distribution of environments represented within MPAs relative to the seascape (PC1: $U=3.9217 \times 10^{12}$, $p < 0.001$ and PC2: $U=5.5037 \times 10^{12}$, $p < 0.001$; Figure 4F). Consistent with the univariate analysis, median values for PC1 place MPAs in warmer, less saline waters while median values of PC2 indicate MPAs tend towards deeper, less productive waters with soft benthic substrate.

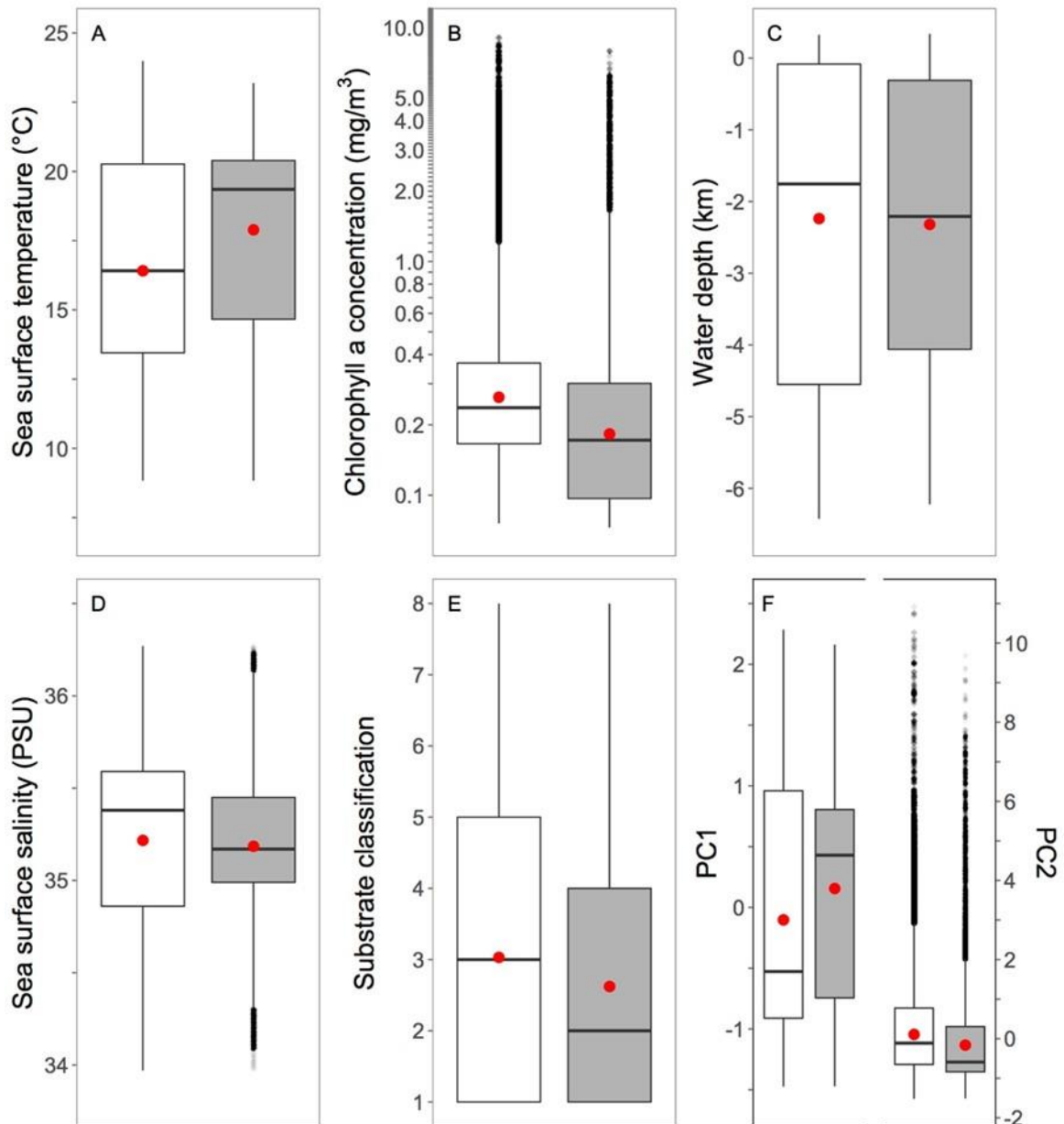


Figure 4. Distribution of attributes chosen for the bio-physical model for MPA and non-MPA grid cells: (A) Sea surface temperature (SST); (B) Depth; (C) Chlorophyll a; (D) Substrate; (E) Sea surface salinity (SSS); and (F) PC1 and PC2. Grey boxes represent only MPAs while white boxes represent the whole EEZ. Red dots on each box represent the mean value.

Levels of protection for MPAs

Within MPAs, the level of protection is not evenly distributed across marine environments (PC1: $H=63098$, $df=3$, $p<0.001$; and PC2: $H=282120$, $df=3$, $p<0.001$; Figure 5). Strict protection (IUCN Category IA) is extremely low or absent across all environments included in the model (Table 2). While temperate reefs tend to be underrepresented in MPAs, the areas protected are offered relatively high levels of no-take protection (IUCN Category II), as are offshore tropical waters (Table 2). Nevertheless, most protected tropical and temperate reefs, and temperate offshore environments, are in the lowest protection category (multi-use zones; IUCN Category VI), that offer little or no protection from extractive activities. Habitat protection zones (IUCN Category IV), which restrict destructive activities such as mining and trawling, form less than 10% of the protection for both temperate and tropical reefs (Table 2).

Model validation

The results of the random forest showed that all attributes made a significant contribution to the model, with the relative importance values all positive for each variable indicating no variable was less predictive than random. The random forest results were consistent with the proportion of variance explained by the different components (e.g. PC1 explains 36% of the variance and the random forest found SST and SSS were the strongest predictors in the model). The spatial distribution of the component values were consistent with the small amount of available, ground-truthed habitat mapping data (Appendix B, Figure B2 & B3). For example, mapped locations of seagrass were represented by higher values of PC1 (i.e. shallow, less saline and warmer waters) and lower values of PC2 (i.e. soft substrate and less productive), which is generally consistent with known habitat requirements for seagrass settlement (Hemminga and Duarte 2000). Additionally, the spatial distribution of PC1 and PC2 aligned reasonably well with the IMCRA bioregions (Appendix B, Figure B4). Four bioregions surrounding the Great Barrier Reef were able to be broadly distinguished from each other due to differences in environmental parameters within those regions (Appendix B, Figure B5).

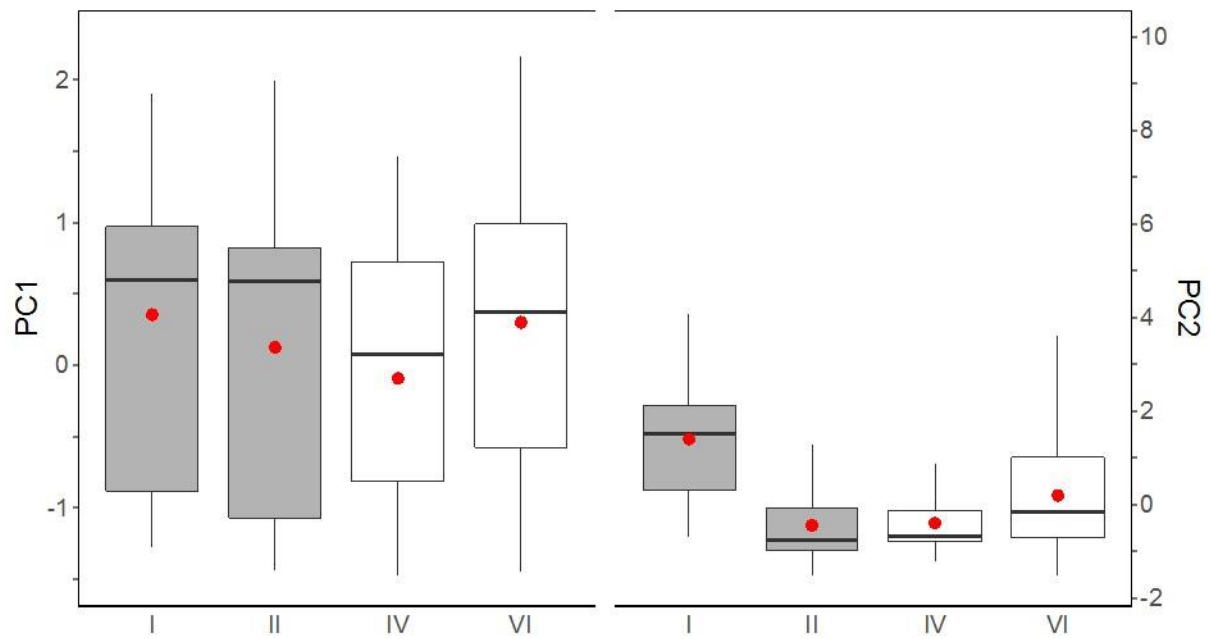


Figure 5. Distribution of the levels of protection designated by IUCN category as indicated by the principal component analysis. Grey boxes represent no-take MPAs. Red dots on each box represent the mean value.

Table 2. The proportion of the Australian Economic Exclusion Zone^a (EEZ) and of marine protection that falls within the different environments, as indicated by the principal components analysis.

	Temperate reef	Tropical reef	Offshore temperate waters	Offshore, tropical waters
	PC1 < 0; PC2 > 0	PC1 > 0; PC2 > 0	PC1 < 0; PC2 < 0	PC1 > 0; PC2 < 0
Breakdown across the Exclusive Economic Zone ^a	18.5%	19.7%	37.1%	24.7%
Breakdown across marine protected areas	11.7%	18.6%	25.9%	43.7%
Proportion of the EEZ within MPAs	24.2%	37.3%	28.6%	69.8%
Breakdown of protection within environments				
IUCN Category Ia ^b	0.1%	0.1%	0.0%	0.0%
IUCN Category II ^b	42.9%	17.6%	27.9%	48.6%
IUCN Category III	NA	NA	NA	NA
IUCN Category IV	8.0%	9.0%	34.5%	20.9%
IUCN Category V	NA	NA	NA	NA
IUCN Category VI	49.0%	72.7%	37.6%	30.5%
Total	100%	100%	100%	100%

^a Excludes the Australian Antarctic Territory, and sub-Antarctic Islands (Heard, McDonald and Macquarie Islands)

^b These IUCN Categories represent no-take marine protected areas.

DISCUSSION

Despite being one of the largest MPA estates in the world, planning for the Australian NRSMPA has been done in the absence of detailed information on the distribution of marine species and ecosystems. This study provides an advance on high level planning regions (IMCRA bioregions) that have been the best available data to guide MPA planning to date, but they lack critical bio-physical attributes (e.g. benthic substrate) and where boundaries were drawn based on subjective decisions. Using fine scale, spatially explicit data on key environmental attribute, yielded a robust, bio-physical model of the distribution of Australian marine environments that provides valuable insights into the distribution and equality of protection among different marine environments. The model reveals key gaps in the levels of protection awarded to biodiversity, particularly in areas that have been shown to be under sustained pressure, such as inshore reefs (Klein et al. 2015). This research could provide a valuable tool for managers to evaluate the representation of different marine environments within the NRSMPA, and priorities for new or upgraded protection to benefit vulnerable biodiversity. However, truly assessing the performance of MPAs in Australia and priority areas for protection will require comprehensive assessment of the distribution of key threats and patterns of ecological connectivity. These are critical next steps to inform marine spatial planning in Australia.

MPA representation of marine environment

Encouragingly, MPAs span the full distribution of environments in Australia (Figure 3A), suggesting commendable progress towards the representation principle of systematic conservation planning (Margules and Pressey 2000). However, in accordance with the assessment made under IMCRA bioregions (Barr and Possingham, 2013), this analysis demonstrates the underrepresentation of many marine environments, along with the substantial overrepresentation of some, which leave significant gaps in the capacity to protect biodiversity. The underrepresented environments (e.g. more productive, temperate waters, which tend to have high levels of endemism; Butler et al. 2010) support increasing concerns that Australian MPAs are more successful in reducing opportunity costs than protecting vulnerable biodiversity (i.e. residual reserves; Devillers et al. 2015).

The bio-physical model suggests that MPAs are mostly protecting tropical, deep water environments, likely heavily influenced by the Great Barrier Reef and Coral Sea, which account for 40% of the NRSMPA. The analysis also shows that the significant growth in marine protection associated with the declaration of 40, large Commonwealth Marine Reserves (Roberts et al. 2018) increased the representation of deep, offshore environments of the EEZ. Beyond protecting important migratory routes and ecological features, such as trenches and canyons (Davies et al. 2007), there is little information about the habitats and species these reserves are protecting. While offshore environments are now provided with extensive, but generally low level protection, most vulnerable biodiversity exists in inshore waters, where anthropogenic pressures are highest (Klein et al. 2015). Previous research has demonstrated that commercially important game species, such as tuna and trevally, frequent shallow reef environments around Australia, as these habitats are important for the juvenile stage of their life cycles (Hobday et al. 2015). Additionally, these habitats are also critical for some overexploited species of shellfish and crustaceans, such as rock lobster (*Panulirus Cygnus*) and abalone (*Haliotis laevigata*). While shallow, reef environments make up a relatively small percentage of the entire EEZ, if MPAs are targeted at the areas of highest biodiversity, then subtidal reef environments should be more strongly represented. This is crucial, not just for the protection of vulnerable biodiversity, but also for the viability of commercial fisheries that rely on the preservation of fish stocks (Weigel et al. 2014).

Level of protection

It is not enough to simply declare a MPA. The levels of protection must be well matched to the biodiversity that require protection and the distribution of pressures on the marine environment (Devillers et al. 2015). No-take MPAs (IUCN Category IA and II) have continuously been demonstrated to be the most effective for biodiversity conservation by offering continuous protection to exploited species and recovering fish stocks (Costello 2014; Edgar et al. 2014; Halpern et al. 2009). Previous studies have found that the most vulnerable biodiversity (i.e. coastal species and ecosystems) have not been given appropriate protection in Australia (Devillers et al. 2015). Unfortunately, the distribution of protection in Australian MPAs is considerably uneven, with the majority of areas falling into the lower protection classes (i.e. multi-use; IUCN category IV-VI; Roberts et al. 2018). The highest level of protection, IUCN Category IA, is so minimal that it barely visible as a proportion of the overall

MPA system. The vast majority of Category IA protection in Australia (128,951 km²; approximately 95%) is given to the offshore, territory islands of Heard, McDonald and Macquarie, which could not be included in the model.

MPAs designated as IUCN Category II are also no-take and cover significantly more area in Australian waters than Category IA (Roberts et al. 2018). The majority of these areas (~72%) occur in tropical waters, further highlighting the disproportionate amount of protection in tropical versus temperate environments, largely due to the dominance of the Great Barrier Reef and Coral Sea (although no-take protection for the Coral Sea has recently been significantly reduced). The temperate reefs of Australia are considered a global biodiversity hotspot for many species of marine invertebrates, with levels of endemism up to 80% for some taxonomic groups and levels of anthropogenic impacts almost matching that of tropical reefs (Bennett et al. 2016). Of the total area protected in temperate reefs, approximately 43% is designated no-take, while the remaining 57% is split between multi-use (49%) and habitat protection (8%) zones (Table 2). Nevertheless, temperate reefs are often poorly represented within MPAs, such as those areas in south-eastern Australia with high levels of endemism (Roberts et al. 2018). Therefore, temperate reefs should be considered priority areas for new MPAs, and as part of rezoning exercises to boost the levels of protection for existing MPAs. Excluding the Great Barrier Reef Marine Park and Coral Sea Marine Reserve (significant areas of which were IUCN Category II at the time of this analysis), this study has revealed some major gaps in no-take protection (IUCN Categories IA and II) throughout the rest of the Australian coastline and EEZ. While this study did not focus on the distribution of threats to marine biodiversity, other studies have shown where biodiversity are most vulnerable, such as inshore, reef habitats, which are heavily utilized by commercial and recreational fisheries, shipping traffic, and tourism-related activities, relative to deep, offshore environments where the biodiversity present is unclear and pressures are generally lower (Klein et al. 2015). Therefore, while strict protection should be targeted towards these areas, there are clear gaps in protection for these environments.

MPAs designated as IUCN Category IV, termed habitat protection zones in Australia, are primarily covering offshore habitats in temperate and tropical waters. These habitat protection zones can protect against the most destructive impacts to marine environments, such as benthic trawling through sponge beds, or mining. Additionally, these zones can

provide protection to migratory species. Given this zone allows a wide range of extractive uses, Category IV should be targeted at offshore environments with lower endemism. Multi-use zones (IUCN Category VI) make up the vast majority of protection for Australian marine environments (Table 2). Given that multi-use MPAs produce fewer ecological benefits relative to no-take MPAs (Campbell et al. 2017; Edgar et al. 2014), this calls into question the long-term capacity of the NRSMPA to safeguard biodiversity. The NRSMPA was established around the principle of developing a comprehensive, adequate, and representative system of MPAs (Lourie and Vincent 2004). However, without knowledge of the distribution of biodiversity the representation and adequacy of MPAs is difficult to measure, and without also having knowledge of the distribution of threats, the appropriate levels of protection for biodiversity cannot be determined. This study represents an advancement over a coarse scale ecoregion analysis and further highlights deficiencies in the data available to assess MPA effectiveness in general. Quantifying the comprehensiveness, adequateness, and representativeness of MPAs in Australia cannot fully be achieved until comprehensive habitat mapping data are available at the required scale. Until this time, bio-physical assessments remain the best available information to inform marine conservation planning in the interim.

Study limitations

The data incorporated into the model was the best available at the time of this study. The spatial resolution of 1 km² chosen for the model was the finest scale available for all bio-physical attributes. However, a significant amount of variability can exist in the marine environment within a 1 km² grid that could not be accounted for in this study. This is particularly true for benthic substrate, where the best available data was so coarse that it had to be converted into an ordinal measure. Likewise, while bathymetry captures some variability in the water column, the vertical heterogeneity and protection of the pelagic environment is difficult to model and often overlooked in marine spatial planning. Future models could therefore be utilized in conjunction with a 3D modelling approach (i.e. Venegas-Li et al. 2018) to better capture specific biodiversity features at depth for more efficient conservation prioritization. To enable a more accurate assessment of the NRSMPA, comprehensive, benthic habitat mapping, should be conducted to remove the need for environmental surrogates. The bio-physical model was designed to help assess the representation of marine environments within Australian MPAs. However, the lack of data

prevented an assessment of other aspects of Aichi Target 11, such as connectivity patterns, the distribution of threats to the Australian marine environment and the effectiveness of management. In order for Australia to truly assess progress toward the CBD target, these assessments should be a priority and used to guide improvements to marine protection into the future.

Conclusion

This study demonstrates that bio-physical models can provide useful tools to assist in efforts to better target future protection based on the most up-to-date knowledge. The approach presented here is easily transferrable to other regions, provided adequate spatial data are available, and can be extended to include more complex parameters and objectives. As remotely sensed data to capture ocean variability become more widely available, bio-physical models in two or three dimensional space can provide a valuable contribution to plan future expansions of MPAs or identify priority sites to increase the level of protection awarded to vulnerable species and ecosystems. Although bio-physical models are a valuable tool in the absence of habitat mapping data, jurisdictions should ideally continue to work towards developing a similar understanding of species and ecosystem distributions as exist to support terrestrial conservation planning.

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Chapter 4

MARINE PROTECTED AREAS OFTEN FAIL AT SAFEGUARDING ECOLOGICAL CONNECTIVITY: AN AUSTRALIAN CASE STUDY

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ABSTRACT

The continued persistence of marine biodiversity requires the establishment of marine protected areas (MPAs). While the area protected within MPAs is growing, the movement of individuals (or larvae) among MPAs, termed connectivity, is often neglected or oversimplified in the planning process. For building population persistence, it is important to ensure that protected areas within a system are functionally connected by dispersal or adult movement. Here, we present a multi-species model of larval dispersal for the Australian marine environment to determine if the extensive system of MPAs truly functions as a network, a common assumption of MPA systems. We show that connectivity among Australian MPAs is highly variable depending on location and species, and that the majority of MPAs outside the Great Barrier Reef Marine Park, are not functionally connected. Our approach highlights the benefits of integrating multi-species connectivity into conservation planning to ensure the configuration of MPAs is capable of producing long-term, sustainable biodiversity outcomes.

INTRODUCTION

In an effort to halt the global decline of marine biodiversity, conserve ecosystem function, and help promote sustainable fisheries, the establishment of marine protected areas (MPAs) has rapidly increased over the past decade. In addition to well-documented benefits, such as increased biomass (Edgar et al. 2014), spillover effects and ecosystem restoration (Campbell et al. 2018), MPAs have been shown to boost resilience to many anthropogenic stressors for species exploited by commercial or recreational fisheries (Costello et al. 2010; Magris et al. 2018). To be effective, MPAs rely on an effective design process guided by systematic conservation planning principles. This process can ensure that MPAs collectively represent the species and ecosystems in need of protection in order to maximise biodiversity outcomes (Margules and Pressey 2000). The long-term success of MPAs also requires that they function as an interconnected ecological network, rather than a collection of isolated parks, ensuring the exchange of individuals between populations (Santini et al. 2016).

While a primary goal of an effective protected area planning process is to ensure species persistence, the movement of individuals (or larvae) among protected areas, termed connectivity, is often neglected, despite playing a crucial role in species' persistence (Hastings and Botsford 2006). Within marine systems, connectivity is most often neglected at broad scales and for plans with multi-species objectives (Kool and Nichol 2015). Connectivity contributes to persistence through ecological dynamics, such as self-recruitment and colonisation, and leads to evolutionary outcomes, such as the flow of adaptive genes in the face of environmental change (Hoffmann and Sgrò 2011; Matz et al. 2018). The exchange of individuals between distinct populations typically occurs during the larval stage for many fish and invertebrate species, which largely depends on ocean currents and larval characteristics to determine likely settlement sites (Tremblay et al. 2008). Empirically, population connectivity is difficult to measure as it occurs at multiple spatial and temporal scales and varies with the life history traits of species (Kool and Nichol 2015). Therefore, models of dispersal have been developed that can account for such complexity across broad spatial scales (e.g., Cowen and Sponaugle 2009; Tremblay et al. 2015).

Connectivity among MPAs, and the role it plays in protecting ecological and evolutionary processes is crucial to achieving the benefits of MPAs. To function as a tool for supporting fisheries, MPAs must protect adult fish populations (Weigel et al. 2014). However, it is equally

important they protect the reproductive potential of sites and subsequent recruitment into critical habitat. For example, a MPA placed in an area with numerous strong connections with nearby habitats will likely have higher persistence and resilience to disturbances (Chollett et al. 2017), and greater potential to support downstream populations. The need to consider multiple species with different dispersal capabilities, as well as the complexities of dispersal modelling, has resulted in connectivity generally being ignored or over-simplified in the design of MPAs (Berumen et al. 2012; Magris et al. 2016). Achieving fisheries objectives should prioritise locations for MPAs with high levels of recruitment and that are net sources of larvae to surrounding fished sites (Krueck et al. 2017). While fisheries management is often only one of many objectives for MPA establishment, integrating multi-species connectivity into conservation planning can ensure a configuration of MPAs capable of producing favourable fisheries and biodiversity outcomes.

While collections of MPAs are often referred to as a network, the term ‘network’ implies that individual protected areas are connected through the exchange of individuals between populations (Minor and Urban 2007; Treml et al. 2008). If the goal of protection is to create a functioning ecological network of protected areas, then it is important to ensure the collection of MPAs function together, through the dispersal of larvae, to ensure population persistence (Krueck et al. 2017). Here, we quantify the degree to which the MPA system (collection of MPAs) in Australia functions as an ecological network. We do this by: i) quantifying multi-species connectivity among all habitat patches, ii) evaluating how much connectivity is safeguarded within the system of MPAs, and iii) map where, and to what degree, the MPAs in the system function as ecological networks. We model multi-species connectivity among all reef habitat patches throughout Australia, including those within the national system of MPAs, the world’s second largest collection of MPAs, using life history parameters from four dispersal phenotypes to explore a range in life history and dispersal capacities.

METHODS

Study area

A biophysical modelling approach (Treml et al. 2012) was used to quantify potential larval dispersal between habitat patches within the Australian Exclusive Economic Zone (EEZ),

excluding the Australian Antarctic Territory (Figure 1). The model includes shoreline data, reef habitat (rocky or coral), MPA boundaries, and ocean current data (~ top 10 m; HYCOM; Chassignet et al. 2007). Reef habitat data was acquired from state and federal jurisdictions and limited to the continental shelf (up to 200 meters deep). To capture the range of temporal variability within the seascape, all years of available current data were utilised (1993-2012). A spatial resolution of 10 x 10 km was used for the model to match that of the hydrodynamic data. While a significant amount of variation can exist within a 10 x 10 km spatial grid, a coarse-scale resolution was necessary to maintain computational efficiency given the extensive model domain. The model domain of the Australian EEZ was then divided into ecoregions using the marine biogeographic classification system defined by Spalding et al. (2007). All analyses were completed or summarised at the ecoregion scale to show broad-scale patterns around Australia (Appendix C).

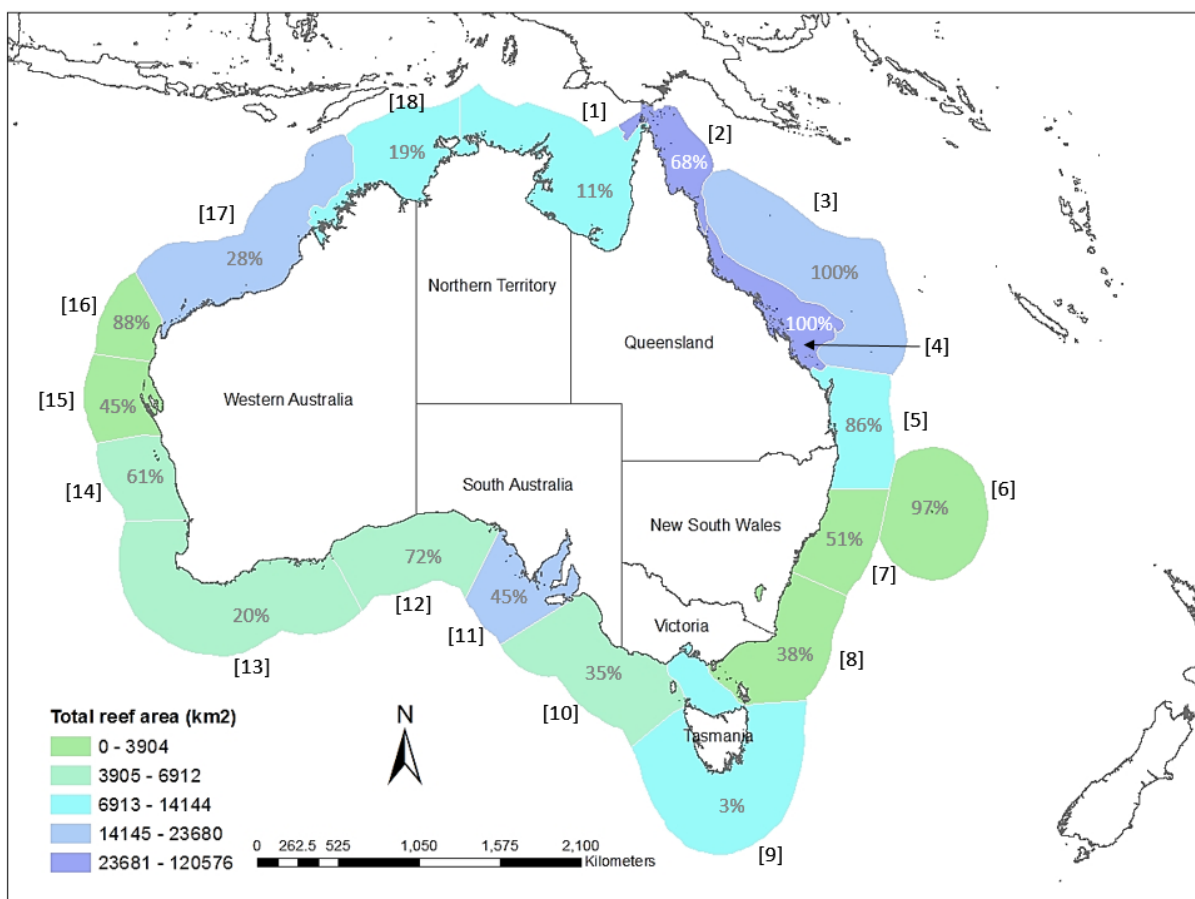


Figure 1. Map of the study area and surrounding ecoregions. Colour gradient represents the area of reef habitat in each ecoregion. Overlying percentages represent the proportion of reef habitat protected within marine protected areas for each ecoregion. Numbers in square brackets correspond with ecoregion name in Table 1.

Model taxa

Movement between habitat patches, while strongly influenced by ocean currents, is partly a function of a species' life history traits. The duration of the larval phase (maximum pelagic larval duration – PLD) has been identified as a strong predictor of dispersal distance (Trembl et al. 2012). The time it takes for a larvae to develop sufficiently to be able to settle on a substrate (competency window) influences the minimum distance they can travel before they could successfully settle on a suitable reef. Together, these two parameters can help estimate how the distance between reefs might influence the capacity for larvae to successfully disperse to new habitats (Trembl et al. 2012). While extremely variable in marine species, it is important to account for daily larval mortality (e.g. due to predation, starvation), which can heavily influence post-settlement survival rates (Trembl et al. 2015). When and how often marine species release larvae (spawning window) is also an important determinant of post-settlement success. Therefore, these important life history parameters (PLD, time to competency for settlement, larval mortality and spawning window) were used to develop dispersal phenotypes with which to model connectivity (Trembl et al. 2012; Trembl et al. 2015).

While behaviour has been shown to impact the local retention and connectivity of larvae for some taxa (Paris et al. 2007), there is both empirical (Gerlach et al. 2007) and model-based research (Trembl et al. 2015) that suggests the impact of behaviour on connectivity is highly variable, dependent on larval biology and seascape context (i.e., where the larvae are), and most-often explored at very small scales (metres). While most larvae have behaviour, incorporating this information into a dispersal model is difficult due to a lack of empirical data and low confidence in how to parameterise behavioural traits (when does behaviour develop, how strong is swimming, what are the vertical swimming strategies, what are the sensing capacities, etc.).

To populate the model, we compiled a database of life history characteristics for a range of marine invertebrates and fishes across different body sizes and trophic levels, capturing species with wide versus narrow distributions and encompassing species targeted or not targeted by fisheries. Based on the life history characteristics, we developed dispersal phenotypes that characterise long range versus short range dispersers. Short range dispersers were selected with broad versus localised distributions. Long range dispersers tend to have large distributions, so phenotypes reflected smaller and larger bodied taxa. This

resulted in four dispersal phenotypes (Table 1), where life history parameters were based on the mean values for approximately 10 indicative species (Appendix C, Tables C1-C4). A 20% larval mortality per day was applied for all model simulations as a compromise between the lower end (10% mortality) for larger, more sturdy larvae (i.e. trevally) and the upper end (30% mortality) for smaller, fragile larvae (i.e. damselfish).

Table 1 . Model input parameters used for each dispersal phenotype

Dispersal phenotype and indicative taxa	Short range, widespread	Short range, localised	Long range, smaller bodied	Long range, larger bodied
Indicative taxa	urchin	damselfish	wrasse	trevally
Life history parameters				
Mean pelagic larval duration (PLD; days)	6	22	30	40
Mean competency window (days)	4	7	13	26
Spawning window	seasonal spawning	seasonal spawning	annual	annual

Dispersal model

Each dispersal simulation consisted of releasing a cloud of larvae over a habitat patch (the quantity proportional to patch area) and allowing it to be transported downstream on ocean currents according to the larval traits. As larvae came into contact with suitable habitat, the total number of competent larvae settling was recorded at each time-step throughout the duration of the simulation. This was repeated for all habitat patches (284 total), for all spawning dates (1st and 15th of each month during the spawning window), for all 20 years and for each modelled species on a high-performance computing cluster (Lafayette et al. 2016). The computing wall-time estimate for a single species was approximately 45 days (250 simulations per species, with four days per simulation). Therefore, to maintain computational efficiency, only four dispersal phenotypes were chosen for this analysis. The final connectivity matrix used for network analysis quantified the proportion of settlers to each habitat patch that came from each original source habitat patch, accounting for mortality, ocean parameters, and biology (Crandall et al. 2014; Samsing et al. 2017). A connectivity strength threshold of 1% was applied to all model outputs to remove weak connections that

contribute less than 1% to a receiving population. Additionally, despite some uncertainty in these estimates (Zamborain-Mason et al. 2017), the model analysis maintained self-seeding connections by incorporating local retention and self-recruitment. Ecological connectivity networks were then built for all species using the resulting matrices and habitat information (e.g. reef location, area) to quantify the connectivity in the marine environment of the Australian EEZ and the proportion captured by MPAs (Figure 2).

ECOREGION NAME	REEF HABITAT				NUMBER OF COMPONENTS			
	LOCATION ON FIGURE 1	# of HABITAT PATCHES	# of PROTECTED PATCHES	AREA of HABITAT (km ²)	URCHIN	DAMSELFISH	WRASSE	TREVALLY
ARNHEM COAST TO GULF OF CARPENTARIA	1	18	7	9,792	6	5	5	3
BASSIAN	9	34	5	11,008	5	3	3	2
BONAPARTE COAST	18	21	11	14,144	3	3	2	1
CAPE HOWE	8	9	2	3,328	3	2	1	1
CENTRAL AND SOUTHERN GREAT BARRIER REEF	4	17	17	120,576	1	1	1	1
CORAL SEA	3	19	19	19,904	4	1	1	1
EXMOUTH TO BROOME	17	25	10	23,680	5	4	3	2
GREAT AUSTRALIAN BIGHT	12	10	10	5,248	3	3	3	2
HOUTMAN	14	7	3	5,952	1	1	1	1
LEEUWIN	13	18	6	6,912	3	2	2	2
LORD HOWE AND NORFOLK ISLANDS	6	3	3	768	1	1	1	1
MANNING-HAWKESBURY	7	3	1	1,920	2	1	1	2
NINGALOO	16	4	4	3,648	1	1	1	1
SHARK BAY	15	9	5	3,904	3	2	1	1
SOUTH AUSTRALIAN GULFS	11	41	26	17,472	2	1	1	1
TORRES STRAIT - NORTHERN GREAT BARRIER REEF	2	12	8	79,616	1	1	1	1
TWEED-MORETON	5	7	7	9,920	1	1	1	1
WESTERN BASSIAN	10	16	6	6,016	2	2	2	1
TOTAL					47	35	30	25

Table 2. List of all ecoregions surrounding Australia with reef habitat details and the number of components per species.

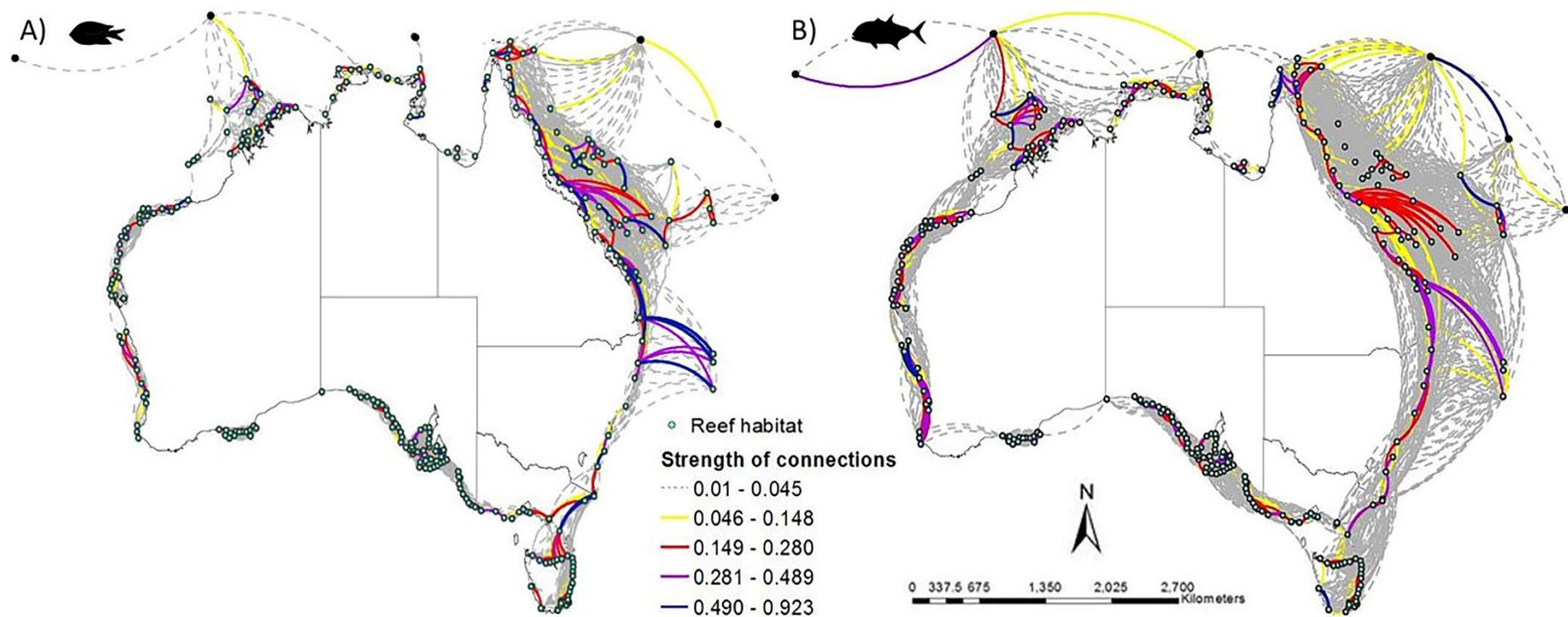


Figure 2. Networks for: A) damselfish (22 day PLD, 7 day competency period) and B) trevally (40 day PLD, 26 day competency period) derived from the final connectivity matrix. Light blue dots represent reef habitat and arcs represent ecological connectivity. Strength of connectivity depicted using the relative inflow matrix. Black dots at northern end of study area represent reef habitat outside of Australia’s EEZ and were excluded from analysis.

Model analysis

To quantify multi-species connectivity among habitat patches, we compared modelled estimates of potential connectivity within the Australian EEZ for each of the four phenotypes. To evaluate how much connectivity is safeguarded within the system of MPAs, we calculated two critical products of population connectivity: 1) the rescue potential of an MPA to surrounding habitat (Kininmonth et al. 2011) and 2) the capacity of an MPA to act as a source habitat (Crowder et al. 2000). Rescue potential refers to the degree to which a reef is supported by the inflow of larvae from upstream reefs, while source habitats are a measure of the larvae that flow out of a reef to support downstream reefs. Therefore, rescue potential was calculated as larval inflow between MPAs relative to the total flow per ecoregion and the degree to which a reef is a source of larvae was calculated as larval outflow between MPAs relative to the total flow per ecoregion. These metrics are often described as dispersal flux (in or out), which represents the relative contribution of each habitat patch to surrounding patches by using the probability of dispersal and the source strength of the donor habitat patch (Urban and Keitt 2001). Areas of high outflow are an important source of larvae to surrounding areas and indicate a productive habitat patch that could bolster downstream recruitment (Minor and Urban 2007). Where these sites support populations of exploited species, the establishment of no-take MPAs could significantly contribute to rebuilding fish stocks through adult spillover (Harrison et al. 2012). Areas of high inflow are receiving larvae from nearby sources, often supporting higher levels of genetic diversity and increasing resilience to disturbance (Minor and Urban 2007).

To map where, and to what degree, the MPAs in the system function as a network, we: a) quantified network fragmentation by calculating the number of components, and b) calculated the proportion of MPAs that were ecologically connected within a component (and therefore functioning as a network). We identified the natural breaks in the network by calculating the number of components per ecoregion. A component is defined as a group of habitat patches in which any two patches are connected through links, with no movement possible between two distinct components (Minor and Urban 2007). The number of components can therefore provide a measure of the fragmentation of the system and may represent isolated metapopulations (Minor and Urban 2007; Trembl et al. 2008). The proportion of MPAs which are ecologically connected as a network, per component, was

calculated using a shortest path algorithm. We considered two MPAs to be ecologically connected as a network if they were linked directly or through one stepping-stone patch (i.e., shortest path of two or less), regardless of its protected status. Metrics for all phenotypes and ecoregions were calculated using the igraph package in R (Csardi and Nepusz 2006).

RESULTS

Connectivity safeguarded within MPAs?

The proportion of both inflow and outflow of larvae protected within MPAs varied considerably across phenotypes, implying that the current configuration of MPAs does not provide the same benefits for all species (Figures 3 and 4). The variation in geographic patterns between phenotypes is a result of the interplay between ocean currents, biological traits and the distribution of habitat. Ecoregions surrounding the Great Barrier Reef, as well as the Ningaloo and Great Australian Bight ecoregions, performed well across all model phenotypes because they contain large amounts of dense, reef habitat with high levels of protection, and therefore both act as strong sources of larvae and have high rescue potential. Across all four phenotypes, the Bassian ecoregion, surrounding Victoria and Tasmania, consistently had the lowest proportions (<15%) of larvae protected, suggesting many larvae do not reach protected reefs and that the rescue potential for these MPAs is low (Figures 3 and 4). This is likely due to very low levels of reef habitat protected (3%) protected in this ecoregion within few, small MPAs, compared to the rest of the Australia's ecoregions. Where differences in the proportions of the inflow and outflow of larvae between protected reefs were observed among dispersal phenotypes, as expected the long-range dispersers showed higher proportions of larvae protected. Only a few ecoregions displayed the reverse pattern. The proportions of outflow (source of larvae) protected within MPAs was consistently higher than the proportions of inflow (larvae settling) for all phenotypes, suggesting that MPAs function as sources of larvae but are vulnerable to catastrophe due to the low rescue potential (Figures 3 and 4).

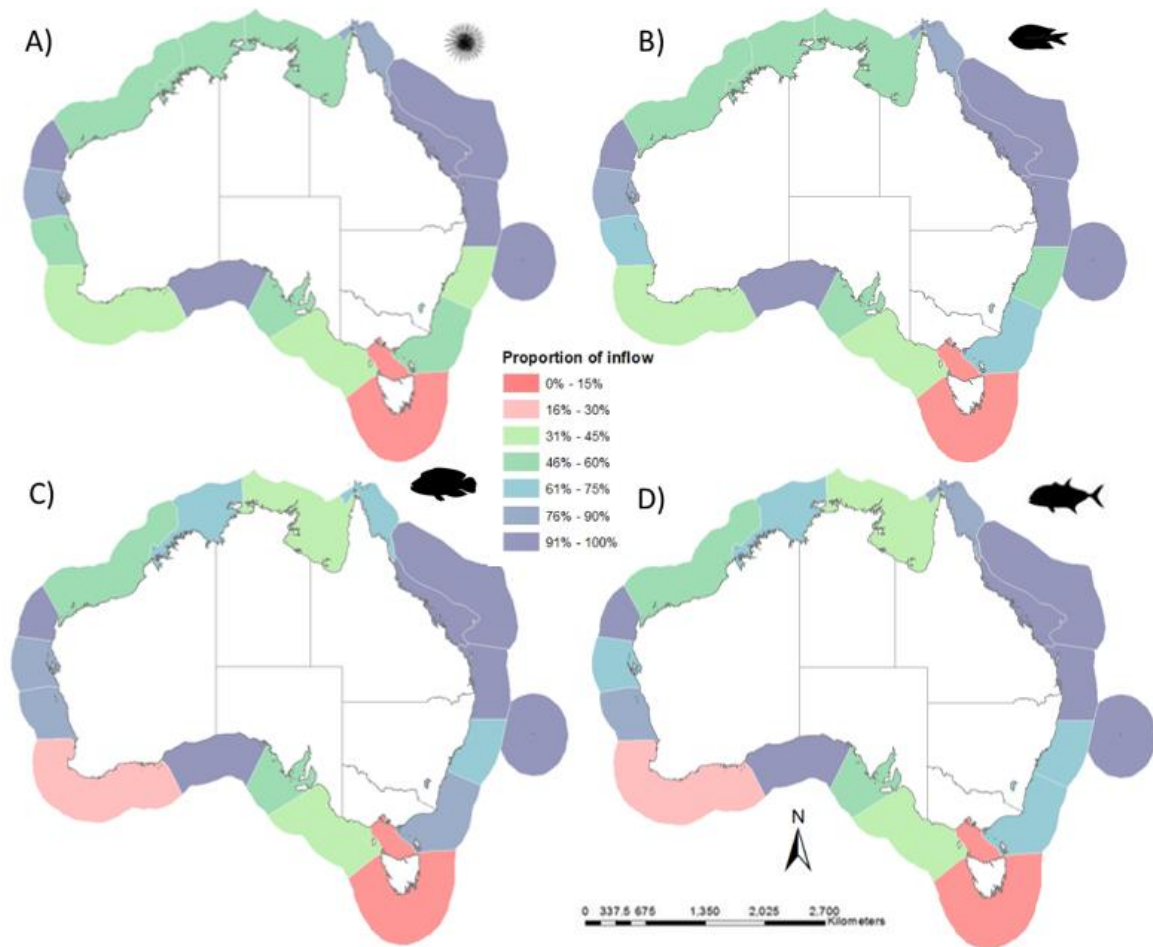


Figure 3. Proportion of the total inflow protected per ecoregion for each modelled phenotype: A) urchin; B) damselfish; C) wrasse; and D) trevally. A habitat patch with a high inflow value indicates a high immigration potential (i.e., a site benefiting from strong upstream sources).

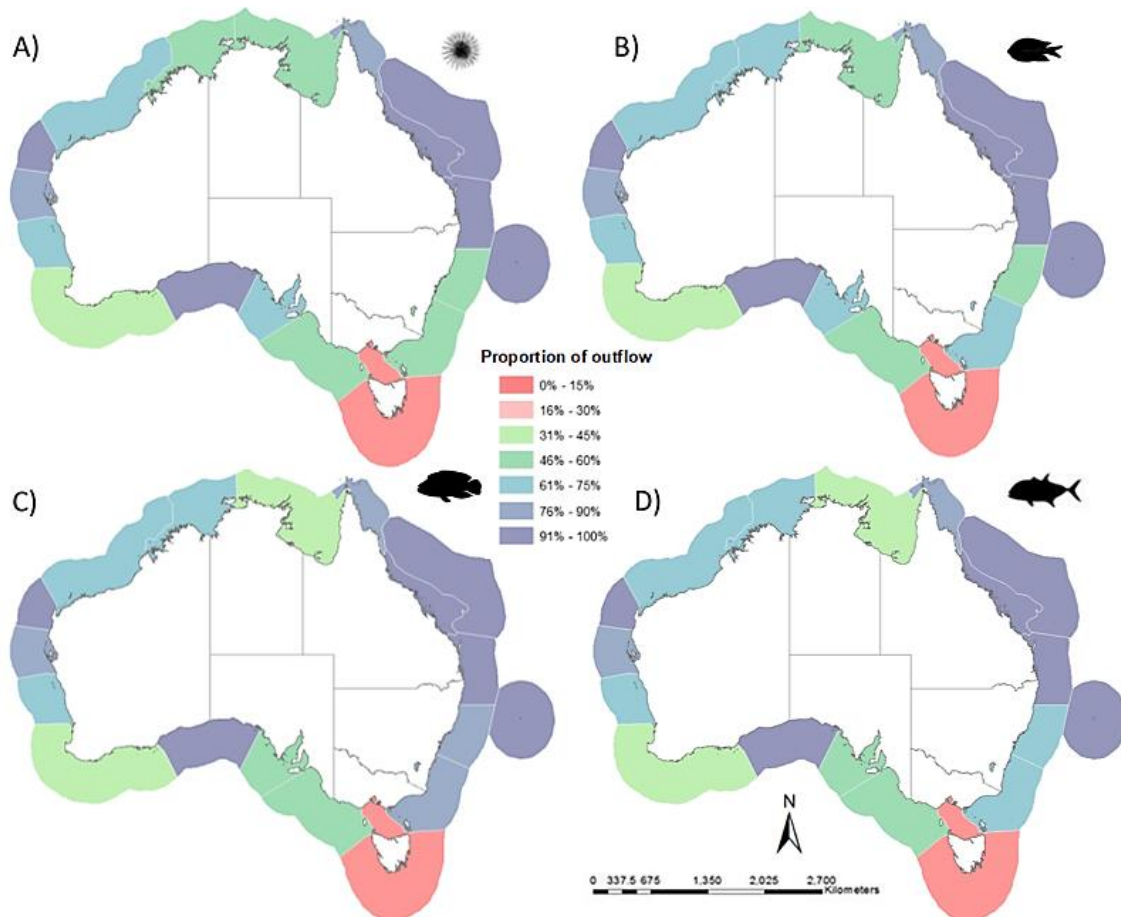


Figure 4. Proportion of the total outflow protected per ecoregion for each modelled phenotype: A) urchin; B) damselfish; C) wrasse; and D) trevally. A habitat patch with a high outflow value indicates the site serves as a strong source to downstream areas.

Are MPAs ecologically connected within a functional network?

The number of components across ecoregions were negatively correlated with dispersal potential, measured through the PLD and competency period (Table 2). These results suggest a more fractured network of habitat and lower connectivity for species with shorter PLDs, such as urchins or damselfish, and more cohesion and greater connectivity for species with longer PLDs, such as wrasse and trevally. A substantial difference was seen in four ecoregions, where the number of components differed by 50% or more between the urchin and trevally model (Table 2). This had significant implications for the proportion of MPAs that were considered to be functionally connected. The Great Barrier Reef and Ningaloo ecoregions are the only ecoregions whose MPAs are working as a functional network, with above 75% of MPAs ecologically connected, regardless of the phenotype (Figure 5). Outside of these well-connected ecoregions, the proportion of functionally connected MPAs generally

ranged from 0% and 50%. The trevally model, which represents the greatest dispersal potential, had 16 of 25 total components with above 50% of MPAs functionally connected (Figure 5). The four ecoregions that had few functionally connected MPAs for the trevally were located in the tropical north around the Northern Territory, the temperate waters around New South Wales, Victoria, and Tasmania, or on the border of South Australia and Western Australia.

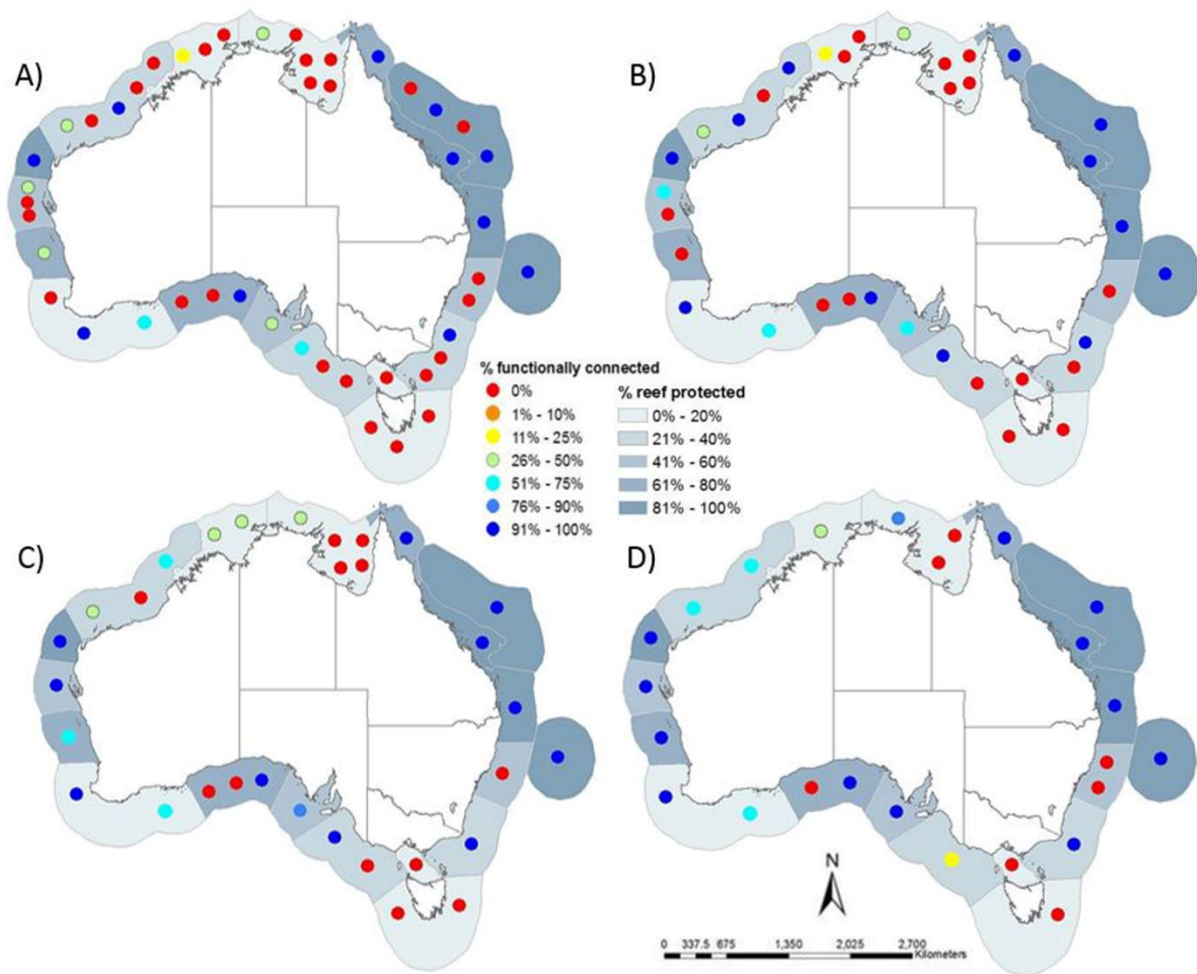


Figure 5. Proportion of functionally connected MPAs per ecoregion per modelled phenotype: A) urchin; B) damselfish; C) wrasse; and D) trevally. Each colored dot within an ecoregion represents an individual component's collection of MPAs.

DISCUSSION

While the global expansion of MPAs in the last few decades has been labelled a conservation success, there are growing concerns that many MPAs have been established without sufficient quantitative data, such as the distribution of biodiversity (Jantke et al. 2018) or the

connectivity of suitable habitat (Schill et al. 2015), to genuinely support the persistence of marine populations. Recent studies have highlighted the importance of integrating connectivity via larval dispersal into MPA design and management, and that including connectivity as an objective in marine spatial planning has the potential to alter the optimal design or configuration of MPAs (Krueck et al. 2017; Magris et al. 2018). Our results lend strong support to the idea that Australia's system of MPAs cannot be considered as a single network (Figure 5). As the number of components demonstrate, Australia's EEZ is built up of numerous networks delineated by natural breaks in the connectivity of reef habitat.

Depending on the dispersal phenotype of species, there are between 25 and 47 individual networks across all ecoregions (Table 2). Therefore, MPAs should be configured to account for this natural fragmentation, leveraging functional connections to ensure they make the greatest contribution to connectivity. Our analysis demonstrates the importance of placing a greater emphasis on the configuration of MPAs when planning or evaluating marine protection, rather than the current focus on the size or shape of areas (O'Leary et al. 2018).

Ideally, MPAs would be large enough to support self-sustaining populations, but also be sufficiently connected to enable dispersal that can buffer against demographic stochasticity and facilitate gene flow that can support genetic diversity and adaptive potential (Magris et al. 2018). The rescue potential of a habitat patch (larval inflow) is crucial to protect against catastrophic events, such as cyclones or disease outbreaks, which have the potential decimate a reef population that was previously self-sustaining. If there is little or no connectivity from nearby habitat patches, then that habitat patch will be less likely or far slower to recover after such a bottleneck. Therefore, sufficient connectivity from outside MPAs as well as inside is important for long term species persistence. The flow of larvae from outside habitat patches is even more crucial for habitats where fishing is permitted. These habitats are less likely to produce as much larvae as no-take areas due to fewer and smaller species present to produce larvae (Barneche et al. 2018).

While important habitat can be identified using numerous metrics, such as representation and the distribution of threats, integrating measures of larval inflow and outflow into the design of MPA systems can contribute to effective management from both fisheries and ecological perspectives. For MPA planning, protecting both the inflow and outflow of larvae should be a priority, as both are crucial for population persistence. However, where

compromise is necessary, an MPA with high inflow of larvae could potentially compensate for low outflow into no-take MPAs because these areas can support larger fish (Edgar et al. 2014), which make a significantly higher contribution to reproductive output (Barneche et al. 2018). While these results are presented as a proportion of larvae protected at the ecoregion level, there will naturally be variation within habitat patches (nodes) as some reefs represent larger proportions of flow compared to others within an ecoregion and will therefore be more valuable. While outside the scope of this analysis, examining the inflow and outflow of larvae at the patch scale could be an important next step in quantifying connectivity among specific MPAs and be used to inform decisions on how to target future areas for protection.

Across all phenotypes, connectivity is well protected within the Great Barrier Reef, Ningaloo Reef (Western Australia), and for sections of the South Australian coastline (Figure 5). For highly mobile larvae (i.e. wrasse and trevally), connectivity is easier to accommodate (Figures 5C & 5D). For those species with limited dispersal capacity and mobility as larvae, the extent of connectivity protected is very limited (Figures 5A & 5B). Results from the urchin and damselfish models revealed that 49-58% of components (natural networks) across all ecoregions have no functional connections between MPAs, such that no larvae move between protected reefs (Table 2 & Figure 5). Our results also demonstrate that the MPA configuration for some ecoregions perform poorly regardless of what dispersal phenotype is being considered and therefore could benefit from additional MPAs to boost connectivity. However, a focus on area protected alone will not achieve a functioning network of MPAs. To successfully integrate connectivity into MPA planning, managers need to be strategic regarding the placement of the MPAs. For example, in some ecoregions (i.e. Great Australian Bight, Manning-Hawkesbury), protecting 51-72% of reef habitat can still result in components with little to no connectivity among MPAs depending on which reefs are protected. In contrast, ecoregions with comparatively less habitat protected (20-38%) can achieve greater than 50% of MPAs functionally connected within natural networks (components) for the majority of species (i.e. Leeuwin, Cape Howe; Table 2 & Figures 1 & 5).

The application of a threshold of less than two links between MPAs (i.e., allowing for one stepping stone between protected reefs) to be considered functionally connected ensures that suitable, protected habitat is able to be connected within two generations. Additional unprotected links between MPAs could, in time, have negative consequences for population

dynamics, such as resulting in extremely isolated populations and potentially higher local extinction rates. Establishing such a threshold, or a baseline of connectivity within MPA planning, can also assist in the identification of priority sites where an additional MPA could significantly increase the functional connectivity for a wide range of species. Our research identified several ecoregions around Australia (i.e. Bassian, Arnhem Coast to Gulf of Carpentaria, & Bonaparte Coast) that should be a priority for MPA establishment due to few reefs being protected and poor connectivity among protected reefs. To improve multi-species connectivity in these areas, future marine spatial planning should ensure that the natural connectivity of habitat is accounted for to ensure the configuration of MPAs achieves functional connections that maximise the connectivity protected across the network. Future research should also parameterize the larval dispersal model to account for higher reproductive output from MPAs known to contain higher size classes of target species and/or higher overall biomass (Edgar et al. 2014). As previous research suggests that larger females within a population contribute disproportionately more to overall replenishment than smaller females (Barneche et al. 2018), no-take marine reserves should be prioritised in future MPA planning and management to support and potentially replenish fisheries (Roberts et al. 2001).

The model developed here was designed to be inclusive of a wide range of species and easily modified to accommodate different dispersal strategies. Therefore, it is easily transferrable to other species and areas with an existing or proposed system of MPAs. Our results suggest that neglecting connectivity in MPA design could hinder its effectiveness in achieving conservation objectives, like species persistence. Furthermore, these results highlight the need to first assess the underlying natural connectivity of a study area prior to implementing new MPAs, rather than assuming that a MPA system can automatically function as a network. As estimates of ecological connectivity from dispersal modelling increasingly become more feasible and available, MPA planners should incorporate connectivity objectives alongside other conservation goals, particularly from a multi-species perspective, to better safeguard species' persistence.

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Chapter 5

EVALUATING PERCEPTIONS OF MARINE PROTECTION IN AUSTRALIA: DOES POLICY MATCH PUBLIC EXPECTATION?

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ABSTRACT

A key conservation strategy to protect and manage marine biodiversity is the implementation of marine protected areas. The level of protection from human activities offered to biodiversity by MPAs is not uniform but varies according to the type of MPA, as well as by jurisdiction. The IUCN embraces this diversity and has developed a classification scheme that recognizes multiple categories of MPA from strict protection to multiple use areas based on the level of resource extraction permitted. This diversity in the activities permitted within MPAs means that reporting total area of marine protection does not reflect the level of protection offered to biodiversity. As such, there is not only the potential for public confusion surrounding what is permitted or prevented within any one MPA, but also for the overall perception of the adequacy of marine protection. Public support is a vital component of an effective MPA planning and management process to ensure these areas achieve their goal of biodiversity conservation. Therefore, it is critical to determine the degree to which the general public understand the activities permitted within MPAs, and how this accords with the actual protection these areas offer to biodiversity. To do this, we conducted an anonymous survey to understand the expectations of the general public as it relates to marine protection in Australia and, specifically, the activities permitted or prohibited within MPA boundaries. Results indicate that there is a clear mismatch between the perceived and actual level of protection afforded to MPAs under current legislation, with the overwhelming majority of respondents (63%) believing that Australia's system of MPAs restrict fishing, when this is only true for 25% of the total area protected. This study identifies a significant gap in the public awareness surrounding marine conservation issues in Australia, suggesting a strong need for an explicit conversation between policymakers, scientists and the general public about whether marine protection needs to be increased to better align with public expectations.

INTRODUCTION

Human pressures on the marine environment, including overfishing, pollution, and coastal development, are causing ongoing and persistent declines in marine biodiversity on a global scale (Halpern et al. 2007). In an effort to restore ecosystem function, as well as stabilise declining fish stocks, there has been a massive surge in the establishment of marine protected areas (MPAs) over the past decade (Gaines et al. 2010; Edgar et al. 2014). This growth can predominantly be attributed to the Aichi Target 11 set by the Convention on Biological Diversity (CBD), which called for at least 10% of global ocean protection by 2020 (Secretariat of the CBD 2011). Despite the impending deadline, MPAs currently cover only 7.4% of the marine environment, lagging well behind the 15% coverage of terrestrial protected areas (UNEP-WCMC and IUCN 2016). Additionally, not all MPAs are created equal. Marine protection varies significantly across MPAs, ranging from strict protection for biodiversity where no exploitation is permitted ('no-take MPAs'; IUCN Categories IA and II) to MPAs that allow for a range of extractive uses, such as commercial fishing ('multi-use MPAs'; IUCN Categories IV and VI; Day et al. 2012). Despite well documented benefits of no-take marine reserves over multi-use areas (Costello 2014; Edgar et al. 2014), only 0.2% of the global ocean offers no-take protection, raising questions and concerns about the capacity of MPAs to safeguard marine biodiversity into the future (UNEP-WCMC and IUCN 2016).

The establishment of MPAs is driven by numerous ecological, social, economic or political factors, all of which can impact on the eventual effectiveness of marine protection. Marine spatial planning (MSP) provides a framework with which to manage competing uses of the marine environment and balance ecological and social objectives (Douvere 2008). This process calls for the involvement of different stakeholders and urges the development of a consensus among all sectors impacted by marine protection prior to the implementation of a management regime (Pomeroy and Douvere 2008). Previous research in MSP as it applies to MPAs primarily focuses on the ability of marine protection to achieve ecological objectives (Agardy et al. 2011; D'Aloia et al. 2017). Social factors are often overlooked or simplified in the MPA planning and management process, despite increasing recognition that they play a crucial role in the long term success of a MPA (Martin et al. 2016; Strickland-Munro et al. 2016). Failing to account for the many human dimensions of MPAs can often result in a wide range of conflicts including opposition to displacement of recreational or commercial

fisheries (Martin et al. 2016), disturbance to unique cultural contexts of the area, or an unequal distribution of the costs accrued by those affected by MPA establishment (Charles and Wilson 2009). For example, the prevalence of multi-use areas around the world (UNEP-WCMC and IUCN 2016) suggests opposition to no-take MPAs from recreational and commercial fisheries as these groups are often successful at influencing MPA planning.

MSP can greatly assist with the placement of MPAs but their governance often varies with jurisdiction, which leads to diverse definitions and regulatory regimes. Furthermore, MPAs are declared in a variety of shapes and forms. As a result of numerous definitions for a MPA, often varying within a single jurisdiction, it is often difficult to determine the activities permitted within boundaries and therefore the exact levels of protection offered to biodiversity (Roberts et al. 2018). This frequently extends to the lack of specificity in the management plan, progress reports, or even the MPA legislation. Restrictions on recreational or commercial activities can be specified either by MPA type or by the zones within an individual MPA, which are subject to change depending on the jurisdiction. This variation understandably creates public confusion, challenges for MPA enforcement (Halpern et al. 2010), and may lead to decreased public support for MPAs in general (Voyer et al. 2012). The marine environment is defined by increasingly diverse uses and complex governance/regulatory arrangements. It is therefore necessary to evaluate public awareness and understanding of the levels of protection for the marine environment in order to determine whether public perceptions align with actual protection.

Previous research conducted in New Zealand and the United States revealed significant inconsistencies and large knowledge gaps in public perceptions of marine conservation relative to actual marine policy. In New Zealand, the general public believed that the coverage of protection for their marine environment is 31%, two orders of magnitude higher than the actual value of 0.3% coverage (Eddy 2014). Public knowledge about marine biodiversity and ocean policy has been shown to be similarly low in the United States, with researchers suggesting better dissemination of information to the public, such as through advertisements rather than television, to increase understanding of marine conservation (Steel et al. 2005). Developing strategies to better direct efforts for marine conservation campaigns or, ideally, drive positive policy changes, first requires a clear understanding of what the general public knowledge is concerning marine public policy.

The Australian MPA system (collection of MPAs in the EEZ) provides an ideal case study to assess public perceptions of marine protection. While Australia has the 2nd largest MPA system in the world, the levels of protection and total area of the system have seen significant fluctuations over the past 15 years (Roberts et al. 2018). Australia has several internationally significant MPAs, such as the Great Barrier Reef in Queensland and Ningaloo Reef in Western Australia, and welcomes millions of visitors every year that contribute billions of dollars to the local economy (Fernandes et al. 2005). Australia is also well known for its active beach culture, with approximately 85% of all Australians living within 50km of the coast. Yet we know little about the perceptions of Australians about marine protection. The Australian MPA system has been examined extensively in terms of its ecological and economic benefits (Watson et al. 2014; Mellin et al. 2016). However, it is unclear what people understand about MPAs or whether the public's perceptions of the protection offered by MPAs matches that currently offered under legislation. Therefore, this study aimed to document the public understanding of marine protection in Australia relative to the current level of protection within MPAs. We also sought to understand the degree to which the public understand the marine regulations they must comply with when in a MPA, which may influence their support for MPAs. This information can help develop recommendations about where transparency and communication with the general public could be improved.

METHODS

Study area

Australia's marine jurisdiction within the Exclusive Economic Zone (EEZ) covers approximately 9 million km², with an additional 2 million km² of Antarctic territory (Butler 2010). Spanning almost 40° of latitude, the Australian marine environment encompasses a large diversity of seascapes, including tropical, temperate, and Antarctic ecosystems, with high levels of endemism found in temperate waters (Evans et al. 2017). Australia's ocean policy, developed in 1998, distributed management of the EEZ across state, territory and federal jurisdictions (Vince et al. 2015). State and territory jurisdiction extend seaward from the shoreline out to 3 nautical miles, while the Commonwealth Government assumes control from 3-200 nautical miles offshore (Vince et al. 2015). Each jurisdiction has a least one piece of legislation which enables them to declare MPAs, along with other regulatory requirements relating to the management of the marine environment (Boer & Gruber, 2010).

The Australian MPA system covers approximately 3.1 million km² and therefore about a third of Australia's ocean territory (Devillers 2015), making it the 2nd largest MPA system in the world. A large proportion of this area (71%) can be attributed to the Commonwealth Marine Reserves declared in 2012, which added approximately 2.37 million km² of protection in federally controlled, offshore waters (Roberts et al. 2018). Each jurisdiction in Australia has developed its own MPAs that are often broken up into zones, which distinguish where commercial or recreational activities are permitted. The activities permitted in each zone are defined in the relevant legislation with little if any consistency among jurisdictions (Roberts et al. 2018) creating a complex national context with the potential to create significant confusion.

Public survey

To assess the public's perception of the protection offered by MPAs in Australia, we developed a one page, anonymous survey to be distributed to members of the general public (Appendix D). The survey was designed to determine which activities respondents believed were permitted or prohibited by MPAs. The survey was presented to visitors of the SEALIFE Melbourne Aquarium in Melbourne, Australia from January-April 2017 and from August-October 2018. These dates were selected to coincide with peak visitation periods. Visitors who were over 18 were invited to complete the survey during their visit. We targeted the general public at an aquarium to ensure the participants had at least some interest in the marine environment and potentially some knowledge about marine protection.

A small amount of background material was provided with the survey, such as the definition of a MPA from the Australian Department of the Environment as well as descriptions of some lesser known marine activities, such as longline and dropline fishing. Respondents were asked to provide their gender and level of education. The Aquarium attracts national and international visitors, allowing us to access a broad cross section of the community. We asked respondents to provide their postcode in order to determine which country and state they were from. The survey presented respondents with the full list of activities permitted in at least one MPA in Australia. Respondents were asked to indicate which of the activities they believe are permitted or prohibited within a MPA.

Data analyses

As respondents were from different parts of Australia subject to different regulatory frameworks for MPAs (Boer & Gruber, 2010) it was important to consider their answers in relation to the activities permitted within MPAs in different states. Given the lack of consistency in the level of protection offered to biodiversity by MPAs across the world, the IUCN has developed a classification scheme to standardise the description of MPAs, ranging from strict protection to those that permit sustainable resource use (Day et al. 2012). The variability in different activities permitted in different protection zones across Australia required an approach to compare the level of protection across jurisdictions and the inconsistent application of IUCN categories across Australia meant that it was necessary to develop an alternative method of interpretation. The protocol developed by Roberts et al. (2018) provided a framework to standardise the classification of Australian MPAs across jurisdiction according to the activities they permit or prohibit (Appendix D, Table D1).

Based on the activities respondents indicated as permitted within an MPA, each survey was assigned an IUCN category, with Categories IA and II being classified as 'no-take' and Categories IV or IV classified as 'multi-use'. All surveys were assessed relative to the state the respondent indicated they lived to ensure a fair comparison with the regulatory environment they may be most familiar with and if these perceptions varied according to state of residence. Surveys completed by international respondents were analysed separately. As the Commonwealth (federal government) zoning arrangements are relevant to all jurisdictions, the federal data were compared with all the Australian respondents.

To compare public perceptions with current levels of protection, the proportion of the MPA system in each state that fell into each IUCN category was calculated (Appendix D). This was completed for the 2014 and 2018 zoning arrangements to reflect a 2018 policy change to the Commonwealth Marine Reserve Network, which downgraded over 400,000 km² of area across 13 MPAs from IUCN Category II (no-take protection) to Category IV and VI (multi-use; Figure 1). Survey results were collated into a single database for analysis in IBM Statistical Package for the Social Sciences (SPSS Version 23).

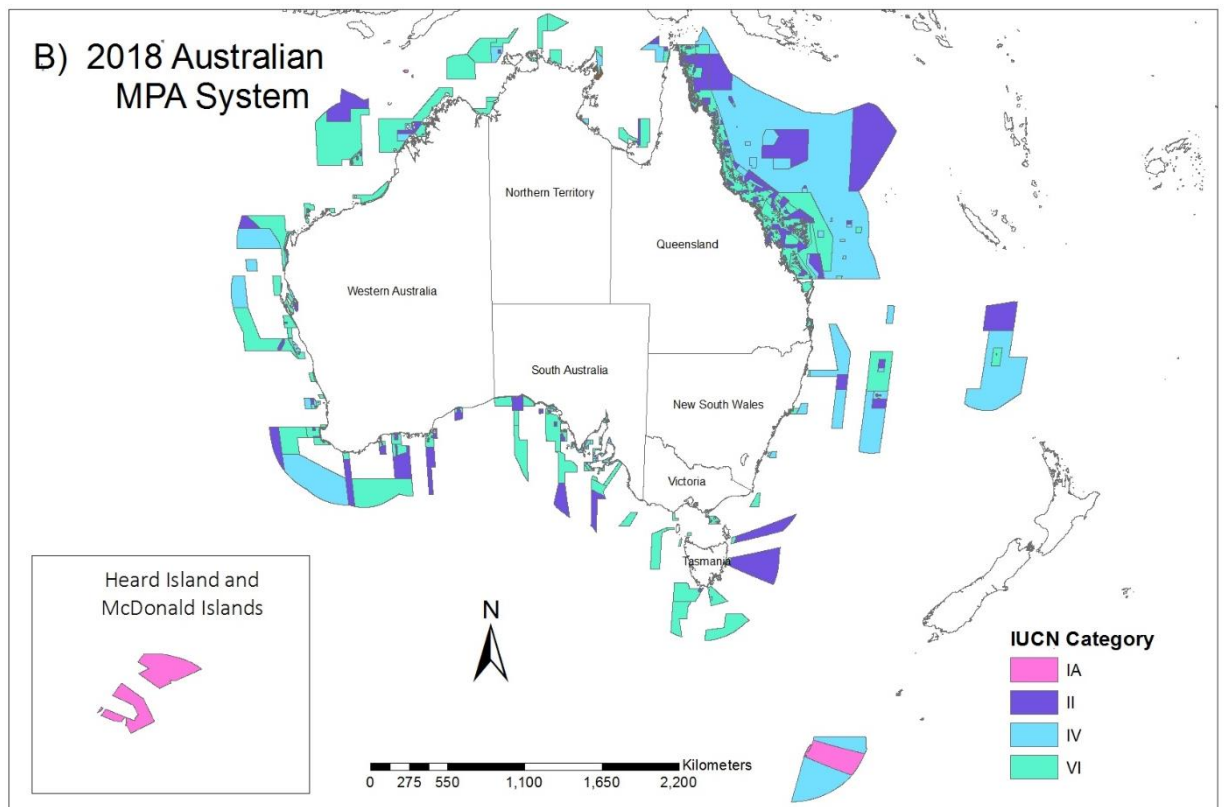
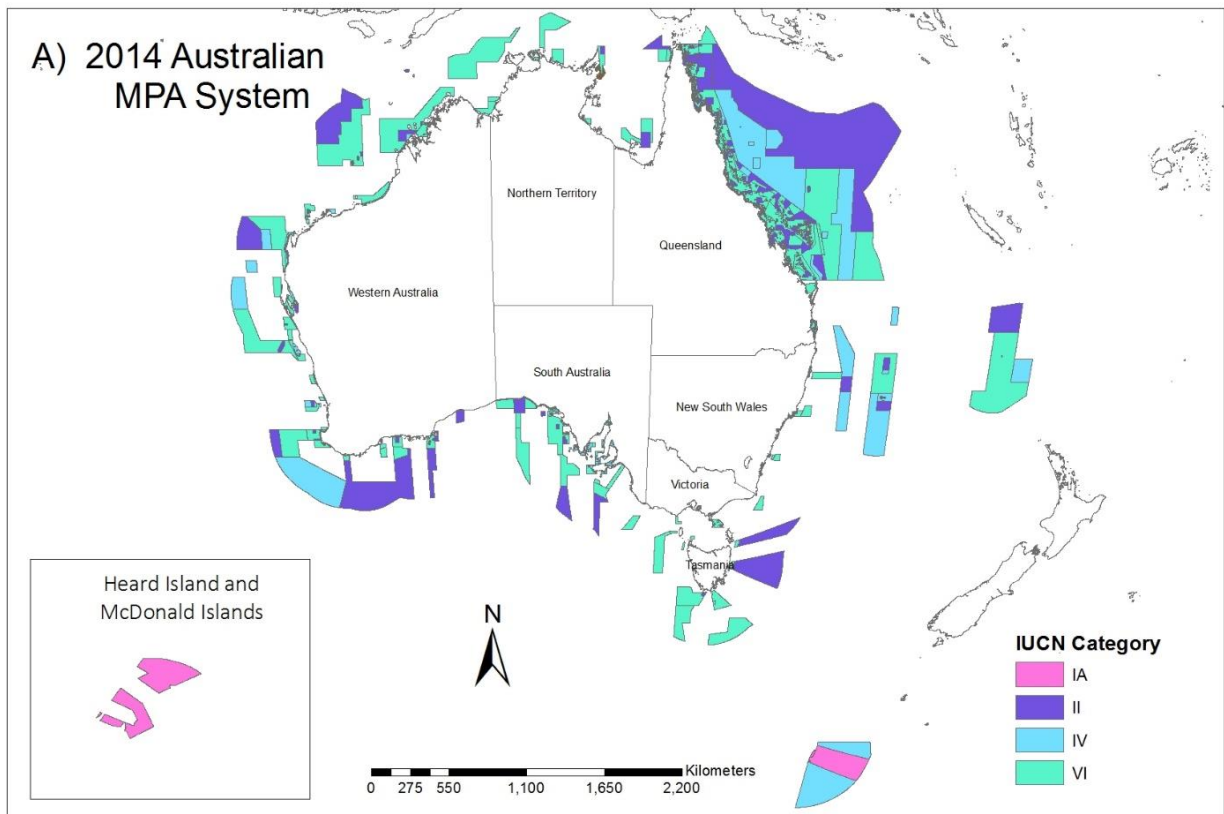


Figure 1. Map of zoning arrangements for the Australian MPA system in A) 2014 and B) 2018.

RESULTS

A total of 360 surveys were received, 308 from Australian residents and 52 from international respondents. The overwhelming majority of Australian respondents (71%) lived in Victoria, with 11% from New South Wales, 6% each from Queensland and South Australia, 2.5% each from Western Australia and Tasmania, and 1% from the Northern Territory. The majority of respondents were female (67%) and well educated, 55% with a Bachelor's degree or postgraduate degree.

Public perceptions of Australia's MPAs

Most Australian respondents indicated that they believed recreational activities to be permitted in MPAs, including scientific collection and research (97%), scuba diving or snorkelling (86%), boating (59%) and to a lesser degree anchoring (39%; Figure 2). Less than a third of Australian respondents (27%) believe recreational fishing to be permitted in MPA boundaries and very few (11%) believe commercial fishing to be permitted. Less than 5% of respondents indicated that trawling or mining are allowed in MPAs in Australia (Figure 2). When compared with international visitors to the Aquarium, Australians were more likely to believe extractive uses were prohibited by MPAs (Appendix D; Figure D1), assuming a lower level of protection overall (Figure 3).

When each survey was assigned an IUCN category based on the activities respondents believed were permitted within MPAs, the majority of Australian respondents (63%) indicated that they believed the MPA system to be 'no-take' (IUCN Category IA or II; Figure 3), offering strict protection to marine biodiversity from extractive activities. When compared with the actual area of no-take protection across the entire MPA system, this result suggests a significant mismatch between public perceptions and actual protection, with only 37% of the area protected within no-take MPAs in 2014 (Figure 4a). This mismatch has increased over time, with the area of no-take protection reduced to 25% by 2018 (Figure 4b). Only 30% of surveys indicated that some extractive uses are permitted, translating to a Category IV interpretation of the MPA system. This category excludes the most destructive activities, such as trawling and mining, where an even smaller percentage of Australian respondents (7%) indicated that these activities were permitted within MPAs, equating to IUCN Category VI.

Multi-use protection (IUCN Category IV & VI) in reality is far more extensive at 75% across all 324 MPAs in 2018, up from 63% in 2014 (Figure 1).

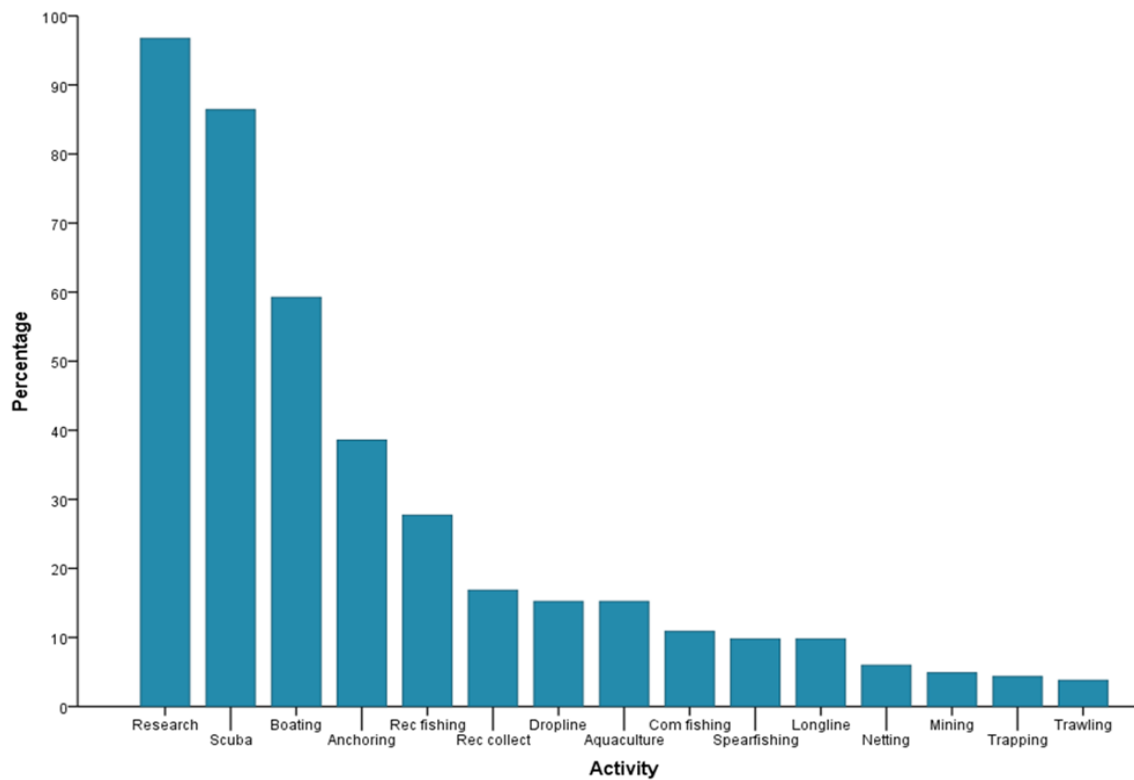


Figure 2. Proportion of Australian respondents that indicated permitted for each activity.

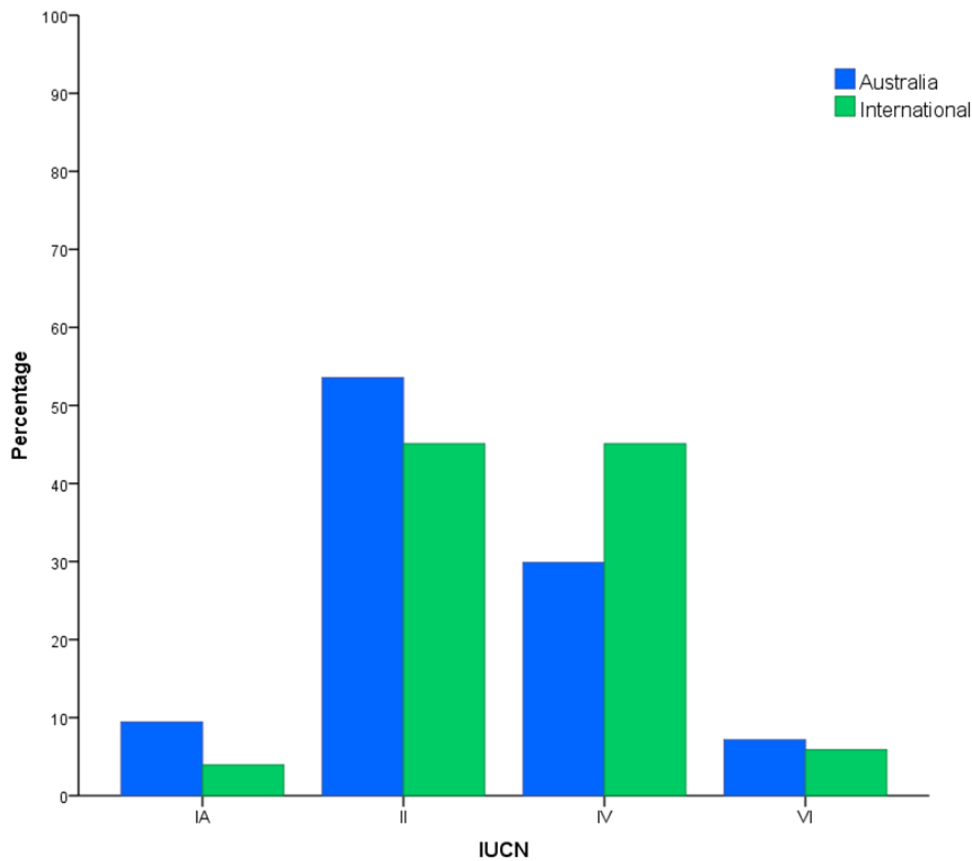


Figure 3. Proportion of Australian and international surveys that translated to each IUCN Category.

Breakdown across states/territory

Victoria

Based on the activities permitted, approximately 65% of Victorians surveyed perceived MPAs to be no-take reserves (IUCN Category IA & II; Figure 5). No-take protection in Victoria equates to approximately 45% of the area protected. The majority of Victorians assumed that research (98%) and scuba diving/snorkelling (86%) are permitted within MPAs (Appendix D, Figure D2). However, 42% of respondents indicated that boating and 65% indicated that anchoring was prohibited in marine parks. Less than 15% of respondents believed longline fishing, dropline fishing or trawling were permitted in MPAs and less than 10% believed that MPAs allow mining (Appendix D, Figure D2).

The reality of marine protection in Victoria is quite different to public perception. Victoria has 30 MPAs distributed across the coastline that range from marine sanctuary (IUCN Category IA) to marine national park zones (MNP; IUCN Category II) to marine and coastal park (MCP; IUCN Category VI). Twenty-four of the MPAs in Victoria are no-take, which equates to 45% of

the total area protected, while the remaining 6 are IUCN Category IV and VI (Figure 5). Due to some variability within MNP and MCP zones as to which activities are restricted, visitors are directed to consult available signage to determine what is permitted in each MPA. Sanctuary and marine national park zones allow for scuba diving, snorkelling, boating, and anchoring. Recreational line fishing as well as trapping, netting, and spearfishing are specified forms of fishing permitted in marine and coastal park zones. Trawling and mining are strictly prohibited in all Victorian MPAs (Table 1).

New South Wales

All respondents from New South Wales (NSW) specified that research, boating and scuba diving/snorkelling are permitted within MPAs (Appendix D, Figure D3). Less than 20% of respondents indicated recreational fishing to be a permitted activity, with no respondents selecting netting, collecting, trapping, or trawling as allowed activities. Only one respondent indicated mining as an activity permitted in MPAs (Appendix D, Figure D3). Similar to Victoria, 68% of residents from NSW believe their current MPAs are predominantly no-take, while in reality no-take protection equates to approximately 20% (Figure 5).

MPAs in NSW MPAs are broken up into multiple zones to indicate permitted activities in a specified area of the MPA. Sanctuary zones are designated no-take, while habitat protection and general use zones allow for fishing and other extractive activities. Habitat protection zones, which make up the majority of multi-use MPAs in NSW (48% of total area), are predominantly dedicated for low impact forms of fishing, such as line fishing and spearfishing. General use zones (32% of total area) allow most forms of commercial and recreational fishing, excluding trawling and mining (Table 1 and Figure 5).
















Queensland

A total of 20 surveys were completed by Queensland residents. Few (26%) Queensland respondents selected recreational fishing as permitted in MPAs and only 11% of respondents indicated commercial fishing to be allowed (Appendix D, Figure D4). No respondents believe that trapping or trawling are allowed in MPAs and only one respondent indicated mining to be permitted, similar to NSW (Appendix D, Figure D4). Approximately 52% of respondents from Queensland view the MPA system as no-take, fewer than those from Victoria and NSW

(Figure 5). No-take protection in Queensland currently encompasses 19% of the total area protected.

Queensland MPAs are zoned similar to those in NSW, excluding the Great Barrier Reef Marine Park which is under federal jurisdiction. Queensland specifies that general use zones (71% of total area protected) permit trawling in addition to other forms of recreational or commercial fishing. Habitat protection zones permit the same forms of fishing as general use zones but prohibit trawling (Table 1).

Table 1. Breakdown across Australian states, territory and federal jurisdiction of permitted or excluded activities in MPAs. ✓ indicates activity is permitted in at least one MPA in that jurisdiction. X indicates activity is strictly prohibited in all MPAs. ✓* indicates activity is permitted under a license or permit. Silhouettes of certain activities correspond to Figure 4.

ACTIVITY	Victoria	New South Wales	Queensland	Western Australia	South Australia	Tasmania	Northern Territory	Commonwealth
Research 	✓	✓	✓	✓	✓	✓	✓	✓
Scuba diving/snorkelling 	✓	✓	✓	✓	✓	✓	✓	✓
Boating 	✓	✓	✓	✓	✓	✓	✓	✓
Anchoring a boat 	✓	✓	✓	✓	✓	✓	✓	✓
Recreational fishing 	✓	✓	✓	✓	✓	✓	✓	✓
Recreational collecting 	✓	✓	✓	✓	✓	✓	✓	✓
Commercial fishing 	X	✓	✓	✓	✓	✓	X	✓
Aquaculture 	X	✓*	✓*	✓*	✓*	X	✓*	✓*
Dropline fishing 	X	✓	✓	✓	✓	✓	X	✓
Spearfishing 	✓	✓	✓	✓	✓	✓	✓	✓
Longline fishing 	X	X	✓	✓	✓	✓	X	✓
Netting 	X	✓	✓	✓	✓	✓	✓	✓
Trapping 	✓	✓	✓	✓	✓	✓	✓	✓
Mining 	X	X	X	✓*	✓*	X	X	✓*
Trawling 	X	X	✓*	✓*	✓*	X	X	✓*

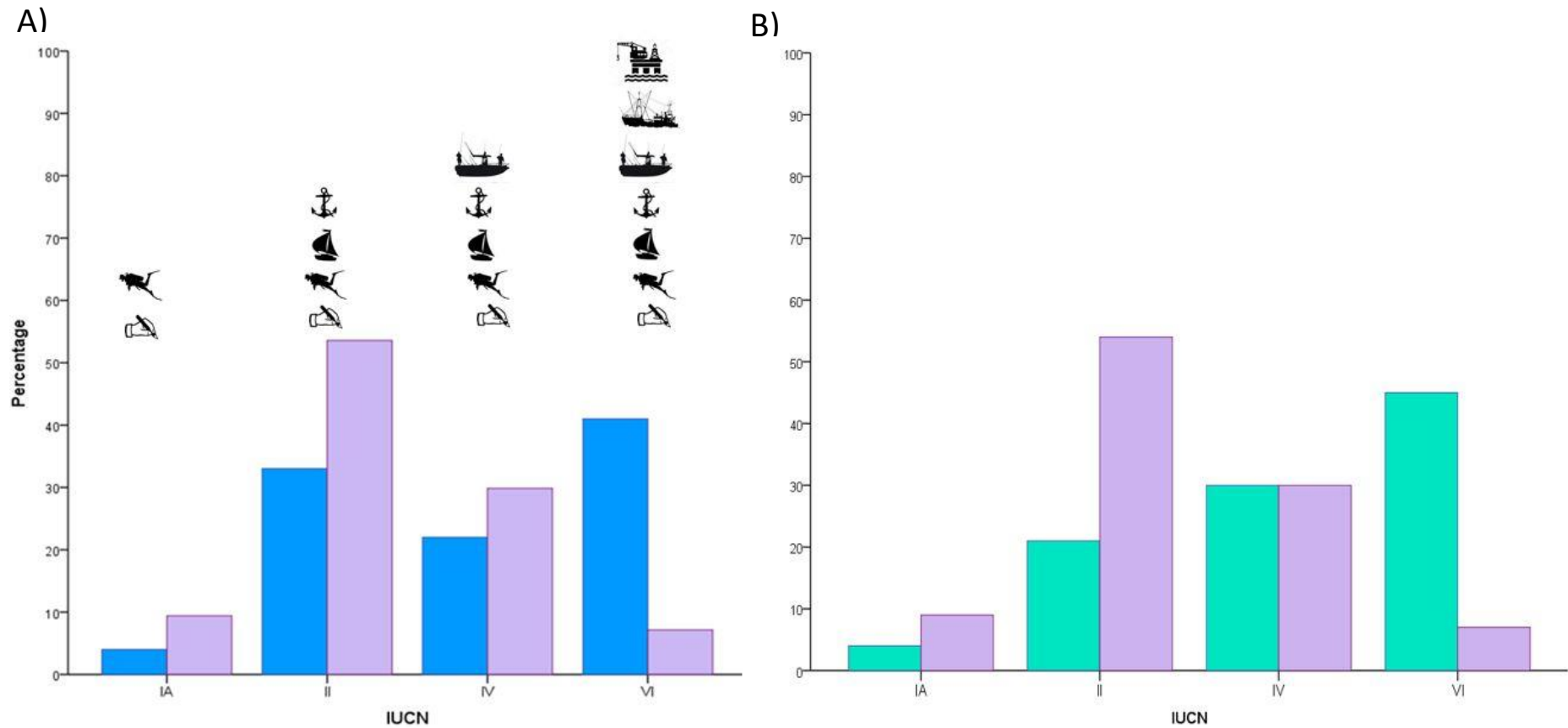


Figure 4. Proportion of A) 2014 actual (blue bars) vs. perceived (purple bars) protection and B) 2018 actual (green bars) vs. perceived (purple bars) marine protection for each IUCN Category. Silhouetted activities above each IUCN category correspond with Table 1.

South Australia

A total of 20 surveys were completed by residents of South Australia. All respondents from South Australia (SA) believe that MPAs permit scientific research and the majority believe boating and scuba diving is allowed (67% and 89%, respectively; Appendix D, Figure D5). Approximately 20% of respondents indicated that all forms of fishing, except trawling (recreational, commercial, spearfishing, etc.) are permitted in MPAs (Appendix D, Figure D5). Additionally, 39% of respondents indicated that recreational collecting is permitted in MPAs, which is the largest proportion from any state for that particular activity. South Australian respondents were also the highest proportions of any state to believe trawling and mining were permitted in MPAs, at 11% for both (Appendix D, Figure D5). Consequently, this led to a lower proportion of surveys that were categorized as no-take (44%) relative to other states (Figure 5). The proportion of no-take protection offered by MPAs in SA is also lower at 16%, compared to 45% for Victoria and 20% for NSW. Trawling and mining are permitted with authorization in SA general use zones, while all other forms of fishing, including commercial, longline and dropline fishing are permitted in habitat protection zones (Table 1).

Other Australian jurisdictions

The small number of surveys (less than 10 per state/territory) completed by residents of Western Australia (WA), Tasmania, and the Northern Territory (NT) makes it difficult to accurately discern patterns in the knowledge of marine protection within each jurisdiction. Of the surveys received, the majority of respondents believe the current MPA system is offering strict no-take protection to the marine environment (75% WA; 62% Tasmania; 50% NT; Figure 5). In WA, only 11% of the area protected by MPAs is designated no-take. Recreational use zones in Western Australia (WA), similar to habitat protection zones in other states, permit commercial and recreational line fishing as well as spearfishing. General use zones allow for most fishing types and mining operations under permit (Table 1). No-take protection in Tasmania is extensive at approximately 80% of the total area. MPAs in Tasmania are not broken up into zones. Instead, activities specified in the management plans as either prohibited or allowed apply to the entire reserve (Table 1). In the NT, only 7% of the MPA area is designated no-take. Most forms of fishing is permitted in general use zones, excluding trawling. The only form of fishing permitted in habitat protection zones is recreational fishing (Table 1).

International surveys

Of the 360 surveys received, 52 were completed by international visitors to SEALIFE Melbourne Aquarium. The respondents represented 20 different countries, with most surveys completed by residents of the United Kingdom (20%) and New Zealand (15%). The majority of international respondents were male (53%) with Bachelor’s degree or higher (71%). Less than 10% of international respondents believe that commercial fishing, trapping, netting, trawling or mining are activities that are permitted in Australian MPAs (Appendix D, Figure D1). However, 30% of respondents indicated that recreational fishing is permitted in MPAs alongside activities such as boating (63%) and scuba diving (90%). International respondents were more likely to believe that the Australian MPA system allows for a wide variety of recreational and commercial activities (Appendix D, Figure D1). Only 49% of international respondents believed MPAs were no-take (IUCN Category IA & II) compared to 63% for Australian residents (Figure 2).

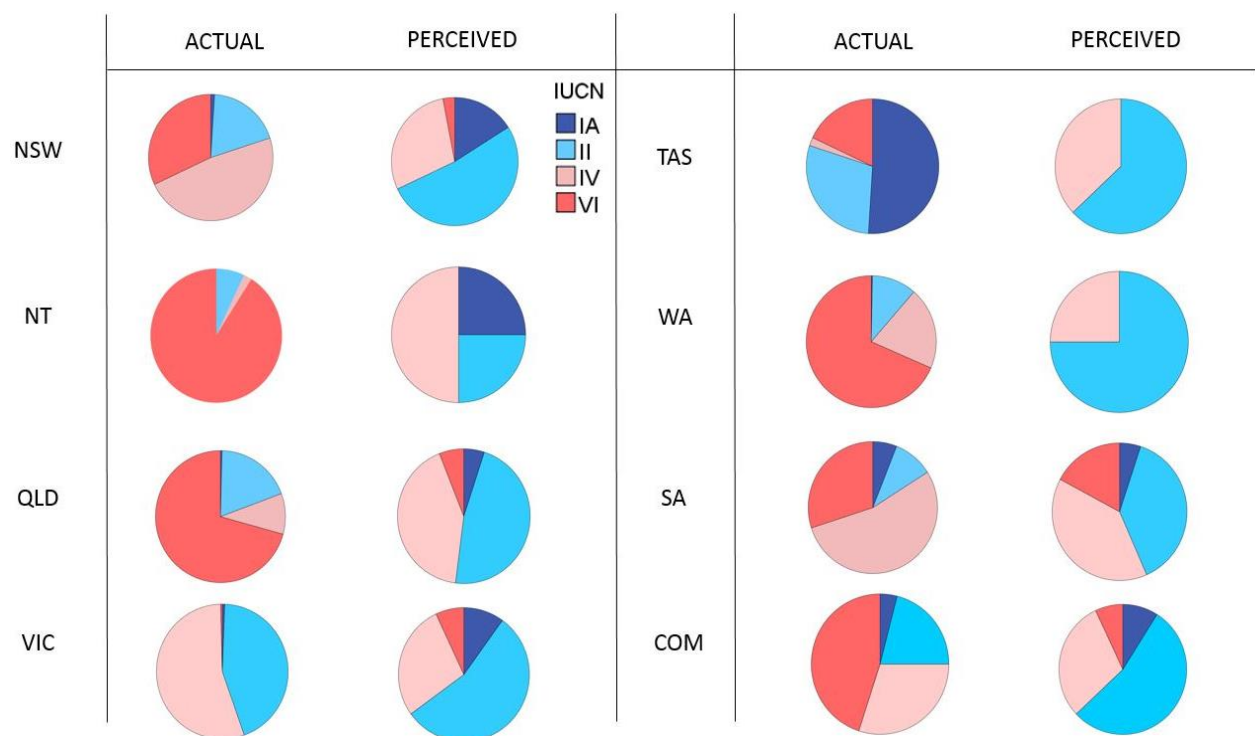


Figure 5. Proportion of actual versus perceived protection broken down by state, territory and federal jurisdiction. Actual protection for Commonwealth MPAs was compared with all Australian respondents

DISCUSSION

Despite having one of the largest MPA systems in the world, there appears to be a significant disconnect between public perception of marine protection in Australia and the actual levels of protection offered within MPAs. The majority of Australians believe their MPAs offer strict no-take protection, although only 25% of the overall MPA system in Australia excludes extractive activities (Figure 4). This suggests that the recent policy changes to MPAs enacted in 2018 that downgraded the protection for 402,360 km² of ocean from no-take to multi-use zones (Appendix D) have actually moved the MPA system further away from the Australian public's understanding of marine protection. This mismatch in public perceptions of protection and the actual protection offered by MPAs has also been observed in studies conducted in New Zealand, the United States, and the United Kingdom (Eddy 2014; Fletcher et al. 2009; Steel et al. 2005), suggesting a widespread gap in the public understanding of marine conservation policy. The general public has the potential to drive positive change and conservation efforts if they are well-informed and aware of environmental issues (Charles and Wilson 2009). However, this study suggests that there is a clear need to improve education and outreach efforts to ensure the public are informed about protection for marine biodiversity, and therefore have the opportunity to contribute to debates about the adequacy of the Australian MPA system.

The Australian perspective

This study demonstrated a misunderstanding about the activities permitted within MPA boundaries, which suggests confusion about the objectives for MPAs. While nearly every marine ecosystem in the world is impacted by some form of human activity, the extent and severity of these impacts vary significantly depending on which activity is being considered. Previous research has quantified the differences among anthropogenic impacts and developed a ranking system to better inform prioritization models for marine biodiversity conservation (Halpern et al. 2007). Trawling, for example, has consistently been identified as having a severe impact on benthic communities, and can produce irreparable damage to habitat structure and community composition (Malecha and Heifetz 2017; Pusceddu et al. 2014). While the overwhelming majority of survey respondents (96%; Figure 2) indicated that trawling was prohibited within MPAs, trawling is permitted in general use zones (IUCN

Category VI) for four jurisdictions (WA, SA, QLD and the Commonwealth; Table 1). The difference in perceived Category VI protection at 7% compared to the actual amount of 45% is the largest misconception of marine protection in Australia revealed by this study.

In addition to documented increases in species density, size and biomass (Edgar and Stuart-Smith 2009, Edgar et al. 2014), MPAs, especially no-take MPAs, can contribute significantly to the replenishment of fished sites through larvae export and adult spillover (Halpern et al. 2009; Harrison et al. 2012). Therefore, it is possible that the large proportion of respondents who do not associate any form of fishing with MPAs do so because of the strong evidence that no-take protection produces better outcomes for biodiversity (Costello and Ballantine 2015). Nevertheless, the perceptions of Australian MPAs as predominantly no-take is representative of only 25% of the total system (Figure 3b). All Australian states, the Commonwealth government, and the Northern Territory have zoning arrangements that permit a variety of recreational and/or commercial fishing methods, from minimal to destructive impact (Table 1). While not as destructive as trawling, fishing practices, such as longline and dropline fishing, can have negative consequences for the marine environment, such as bycatch or the destruction of benthic habitats (Davies et al. 2009). Longline fisheries, for example, frequently result in the death of larger, often endangered animals like sea birds, sea turtles, whales, and sharks (Watson et al. 2005). Policymakers are often making compromises with the recreational and commercial fishing industry when it comes to MPA planning. This often results in the establishment of habitat protection zones, which provide protection to the sea floor but allow for exploitation of biodiversity within the water column (Day et al. 2012). Fishing is now allowed, in some form, throughout 75% of the Australian MPA system. Downgrading of protection for Australian waters has been enacted despite the evidence that no-take MPAs have the greatest benefit for biodiversity and fisheries (Costello and Ballantine 2015; Edgar et al. 2014).

MPAs are often a contentious subject among commercial and recreational fishers (Martin et al. 2016). The opposition to MPAs likely stems from a lack of understanding or scepticism of the long-term benefits associated with MPA zones (Martin et al. 2016). Previous research on perceptions and attitudes of recreational fishers found that, where support for MPAs was lacking, fishers indicated that the rules and regulations associated with different zones were either unclear or not accessible (Martin et al. 2016). Furthermore, most stated that they did

not believe the cumulative impact of recreational fishing had a detrimental impact on the ability of a MPA to achieve biodiversity conservation due to other management actions such as catch limits (Martin et al. 2016). Evidence suggests that, for some fish species, recreational fishing can greatly impact the overall stock status and even surpass that of catch by commercial fishing (Coleman et al. 2004; Lloret and Font 2013). This emphasizes the need for a transparent and effective method of communication to ensure the general public is aware of the science underpinning marine protection as well as the regulatory reality (Voyer et al. 2012). A clear method to improve compliance with MPA regulations and ultimately improve MPA performance in achieving biodiversity objectives is to ensure boundaries are clearly marked with self-explanatory signage and zone rules are effectively communicated prior to implementation.

The international perspective

Surprisingly, the international survey respondents assumed an overall lower level of protection for the MPA system than Australian residents. A total of 20 different countries were represented in survey sample, all with maritime jurisdictions of their own and therefore likely dependent on fishing in some capacity to support livelihoods. Not all countries represented have implemented MPAs. However, the two most well represented countries in the survey, New Zealand (n=8) and the United Kingdom (n=10), have long histories with marine conservation and have both declared MPAs (Eddy 2014; Fletcher et al. 2009). Previous research in both countries presents similar knowledge gaps on ocean conservation issues and, in New Zealand, a similar assumption of a more protected system of MPAs than that currently offered under the legislation (Eddy 2014; Fletcher et al. 2009; Hawkins et al. 2016). Currently, New Zealand's 35 marine reserves are entirely no-take, providing a long-term perspective on the benefits of such legislation to conservation, tourism and fisheries, as well as providing justification for an expanded network of MPAs and increased protection for marine biodiversity (Ballantine 2014). In the UK, previous research has revealed that public opinion of general ocean health is pessimistic yet the majority of respondents support the concept of MPAs and an increase in protection to >40% of the UK's ocean territory (Hawkins et al. 2016). Currently, about 16% of coastal waters under UK jurisdiction is designated by a MPA or other form of conservation management (Hawkins et al. 2016). These insights into

public perceptions from two countries suggest that there is a general need to improve ocean conservation literacy through education and outreach efforts.

Study limitations

This study provides a snapshot of the understanding of the general public about marine protection, sampled from one location in Australia. While the survey was only administered in one location, the consistently high volume of visitors to the Melbourne Aquarium from across Australia, as well as internationally, resulted in almost half of surveys completed by non-Victorian residents (29% from other Australian states; 14% from other countries). Nevertheless, the views of Victorian residents dominate the sample, providing greater confidence in the understanding of the views of Victorian marine protection. Ideally, a survey that is more widely distributed could provide a more comprehensive picture on how perceptions of marine protection varies with local context. Administering the survey at an aquarium may also have biased the sample towards visitors with an interest in the marine environment, who may be more conservation-minded and therefore may not fully represent the broader public. However, if this was the case, then it might be expected that the broader public may have a poor understanding of marine protection than the sample in this study.

The survey was designed to capture perceptions of MPAs Australia-wide, rather than ask respondents about MPA protection in their state/territory of residence. This is due to the assumption that, while there are significant regional differences in MPA zoning across Australia, it is likely that respondents are more familiar with Australia's larger, iconic parks such as the Great Barrier Reef. Respondents were asked to indicate the activities permitted or excluded within an MPA with the lowest level of protection in Australia. To account for the fact that respondents may be more familiar with the protection context in their own jurisdiction, the survey results were then analysed and interpreted according to the protection within their state of residency. Future research could revise the survey question to ensure respondents are asked to answer specifically about their knowledge of MPAs in their region. This will allow for a more comprehensive analysis of the public perception of current marine protection by jurisdiction and could potentially provide valuable insight for future marine spatial planning in a given area.

In an effort to keep the survey concise (1 page limit), respondents were only asked about current allowances/exclusions by MPAs, not what they believed marine protection should be in Australia. Therefore, it was not possible to determine whether marine protection meets the expectations of respondents based on the purpose or objectives of MPAs, nor whether they believe protection should be increased or decreased. Given the mismatch in perceived versus actual protection revealed by this study, there is a clear need for additional research into whether increased awareness of marine policy may lead to pressure for policy change in Australia, especially regarding the adequacy of marine protection.

Conclusions

The global progress of MPAs shows conclusively that there are many long term benefits associated with effective protection, such as the replenishment of fish stocks and increases in species size and biomass (Edgar et al. 2014; Halpern et al. 2009). Global studies also show that public involvement in the MPA planning process is crucial to ensure that protection meets public expectation (Charles and Wilson 2009; Voyer et al. 2012). There is no lack of support for MPAs from the general public when they are made aware of important issues related to ocean health (Steel et al. 2005). This study supports previous research into public perceptions of marine protection (Eddy 2014; Fletcher et al. 2009; Hawkins et al. 2016) and provides further justification for increased protection to Australia's system of MPAs. Given the recent downgrades to many MPAs under Australia's federal jurisdiction, this research is very timely and suggests that the current policy is out of step with public understanding of marine protection. The 38% difference in perceived (63%) versus actual (25%) no-take protection in Australia's waters provides evidence that the opinions of the Australian general public are more in line with scientific understanding of the impacts of destructive fishing methods on the continued preservation of marine biodiversity. This study highlights a clear need for an explicit conversation between policymakers, scientists, and the general public about whether marine protection needs to be increased to better align with public expectations.

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Chapter 6

DISCUSSION

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Scientific research from around the world supports the theory that MPAs are a realistic and effective tool to safeguard vulnerable marine biodiversity (Halpern 2003; Edgar et al. 2014). Protected areas, both marine and terrestrial, are expanding globally yet rarely is the time taken to consider what these protected areas are supposed to achieve or what features lead to an effective MPA. The main driver for increasing the total area protected, as quickly and efficiently as possible, is the international Convention on Biological Diversity (CBD) Aichi Target 11 (Devillers et al. 2015). The Aichi Target, however, encompasses much more detail than just total area protected, recognising that measuring conservation progress with a sole focus on total area is very limiting. This target also calls for MPAs that are ecologically representative of species or habitats, well-connected and effectively and equitably managed. It also emphasizes the need for a participatory and science-based planning process to incorporate clear and quantifiable biodiversity objectives as well as the engagement of local and/or indigenous communities (Secretariat of the CBD 2011).

The value of a MPA system to biodiversity conservation depends on a range of attributes of both individual MPAs and portfolios of MPAs, which are not captured by simple area-based targets. Some of the attributes of effective individual MPAs include no-take protection, well enforced, isolated, large, and at least 10 years old (Edgar et al. 2014). The MPA features required to produce favourable biodiversity outcomes should be incorporated as a crucial element of the planning process. The design of the Australian MPA system has frequently been criticised as deliberately avoiding the areas important for fisheries or oil and gas interests (Devillers et al. 2015) and many have questioned the system's capacity to buffer biodiversity against destructive impacts. An objective evaluation of progress was required in order to determine if the massive growth in the total area in MPAs since 2002 equaled significant strides towards biodiversity conservation. To fill this knowledge gap, the overall objective of this research was to evaluate the design of the Australian MPA system and to assess its capacity to enable biodiversity to persist long term. This study is the first to take an Australia-wide, longitudinal perspective on key aspects of Aichi Target 11 for the world's second largest MPA system and provide insights into the current state of marine protection in Australia.

Leaning heavily on previous research into global MPA effectiveness (Costello and Ballantine 2015; Edgar et al. 2014; Halpern 2003), this study assessed change in the structure of the

MPA system (Chapter 2), representation of biodiversity (Chapter 3), connectivity (Chapter 4), and public perception of protection (Chapter 5). Spatially explicit data were acquired from every jurisdiction in Australia to produce three, high resolution models of the Australian marine environment for Chapters Two through Four. These models enabled the identification of the strengths and possible weaknesses in the MPA system that would impact effective biodiversity conservation. The fifth chapter utilized a survey of the general public to understand public perceptions of marine protection in Australia.

The Good News

The legal procedure for declaring a MPA is often challenging, with limited resources and financial investment available for MPA planning (Christie et al. 2017), and complex relationships with diverse stakeholders and interest groups to navigate. Additionally, the ecological and/or economic benefits associated with MPA establishment could take years or even decades to achieve. Given the challenges, successfully navigating the process of drawing boundaries for MPAs and organizing management plans is intrinsically significant progress for marine conservation. Results of Chapter 2 lend strong support to the claim that Australia has built a well-structured system in terms of broadly representing ecologically distinct bioregions (Roberts et al. 2018). Iconic MPAs, such as the Great Barrier Reef and Ningaloo, are both representative (Chapter 3) and well-connected (Chapter 4). Additionally, despite documenting some of the first marine PADD events in the world (Roberts et al. 2018), this study revealed that the Australian MPA system is much more stable than the terrestrial PA system, where PADD events have impacted over a third of the total area protected (Cook et al. 2017). Now that the boundaries of the MPAs are in place, this assessment has provided clear insight into where to build on these strengths to fill existing gaps in representation, improve connectivity and increase the levels of protection for important biodiversity.

Areas for Improvement

A consistent outcome of across all chapters was a recommendation regarding the level of protection. Previous research has found that MPAs globally produce more favorable biodiversity outcomes when they are designated no-take (Edgar et al. 2014). While MPAs are designated for a variety of reasons other than biodiversity conservation, the amount of no-take protection (IUCN Category IA & II) in Australia is only 25% of the total area. Furthermore,

the bulk of this protection is primarily located in the offshore islands of Heard, McDonald and Macquarie. These island territories are thousands of kilometers from the Australian mainland and are therefore under less pressure from destructive fishing practices or mining interests. The most vulnerable and commercially valuable biodiversity are often concentrated in nearshore environments (Klein et al. 2015), where no-take protection is currently severely lacking (Roberts et al. 2018). These coastal MPAs are also smaller in size than offshore MPAs and thus more fragmented (Roberts et al. 2018). The placement and level of protection within MPAs are important factors to consider in the planning process, but the size of the MPA is also a significant predictor of biodiversity outcomes (Edgar et al. 2014).

Representation

As Aichi Target 11 calls for a representative system of MPAs, the aim of Chapter 3 was to use available data on the distribution of marine environments to evaluate how well the Australian MPA system captures marine biodiversity. While recent studies have examined MPA representation relative to broad bioregions (Barr and Possingham 2013), Chapter 3 developed a finer-scale bio-physical model that incorporated attributes known to be associated with species distributions (e.g. sea surface temperature, primary productivity, and substrate) to model MPA coverage of marine environments and equality of protection. Results from the model reveal that Australian MPAs over-represent tropical, offshore environments such as the Coral Sea on the border of the Great Barrier Reef. Temperate reefs, which are known in Australia for high levels of endemism (Butler et al. 2010), are heavily under-represented by the MPA system.

The bio-physical model demonstrated that the distribution of protection across the marine environment is also considerably uneven. Inshore reefs, which have been identified by previous research to be under sustained pressure from anthropogenic impacts (Klein et al. 2015), have major gaps in no-take protection. The bulk of Category II no-take protection is covering the offshore, tropical areas, again leaving temperate and inshore reefs under-represented. Most importantly, this research highlighted a critical need for comprehensive habitat mapping at large spatial scales. The lack of knowledge regarding the distribution of marine biodiversity makes it difficult to accurately evaluate the representativeness and adequacy of the MPA system as a whole. In the interim, spatial models that use bio-physical attributes, such as those produced in Chapter 3, can provide useful proxies for the

distribution of biodiversity, as well as a valuable tool for policymakers to identify priority areas for new MPAs or areas where protection should be increased to benefit vulnerable biodiversity.

Connectivity

While much more complex and often difficult to measure, the Aichi Target 11 stresses the need for designated MPAs to be well-connected. This requires that MPAs are placed in a configuration that allows them to function as an ecological network, rather than a collection of isolated parks (Santini et al. 2016). Despite being of crucial importance for population persistence, connectivity is often neglected or oversimplified in MPA planning (Magris et al. 2016). The lack of quantitative data to obtain a true measure of connectivity, as well as the limited knowledge of the distribution of biodiversity, explains the growing concern that MPAs are being implemented opportunistically and therefore unlikely to be as effective at supporting the persistence of marine populations.

Despite the Australian MPA network being touted as one of the best examples of ocean conservation in the world (Devillers et al. 2015), the extent to which the MPA system was functioning as a network for important biodiversity had not been quantified. The aim of Chapter 4 was to develop a multi-species dispersal model of connectivity to quantify the degree to which the MPA system in Australia functions as an ecological network, allowing for larval exchange and rescue among MPAs. The results from this study suggest that Australia's MPAs cannot be referred to as a single network, but rather a collection of numerous smaller networks delineated by natural breaks in the connectivity of reef habitat (Chapter 4). While iconic parks such as the Great Barrier Reef and Ningaloo are well designed to safeguard ecological connectivity, the rest of the EEZ is built up of between 25 and 47 individual networks, depending on which focal species is being considered. This result further highlights the inadequacies of measuring success by only reporting total area protected. While this research identified several areas around Australia that could benefit from additional MPAs to facilitate connectivity, managers need to be very strategic regarding the placement of any new MPAs. Planning for effective MPAs requires the consideration of many different components, among them should be the underlying natural connectivity of a study area so MPAs can leverage functional connections to ensure they make the greatest contribution to connectivity and therefore the persistence to marine species.

Public perceptions of marine protection

The level of protection within MPAs, specifically the small amount of strict, no-take protection, has been a central theme throughout this research. The level of protection within MPAs is important for understanding the effectiveness of protection for biodiversity but also for local communities and users of the marine realm to understand what recreational or commercial activities are permitted within MPA boundaries. Due to the fact that not all MPAs offer equal protection to the marine environment, there is often confusion surrounding what is permitted or prevented within a MPA (Voyer et al. 2012). To reveal some of this confusion and inform the debate between science and public policy, Chapter 5 aimed to understand how the general public perceives the protection offered to the marine environment by Australian MPAs.

The timing of the survey used to assess the public understanding of marine protection coincided with a large policy change enacted by the Australian government, which downgraded over 400,000 km² of ocean from no-take to multi-use zones. Results from Chapter 5 reveal that the majority of Australians (63%) perceive the Australian MPA system to be predominantly no-take, when the actual amount of no-take protection is currently only 25%. This suggests that the existing policy is out of touch with public opinion when it comes to marine protection. Given the lack of knowledge surrounding what is allowed in MPAs, it is evident that an explicit conversation between policymakers and the general public regarding these changes needs to occur. Constructive public engagement should be considered a vital component of the MPA planning and management process to ensure all established MPAs can achieve their goal of biodiversity conservation without negative reactions from the general public (Watson et al. 2014).

Future work

This research highlighted the importance of multiple elements of effective MPAs that are important to consider in the MPA planning and implementation process. However, some crucial parameters to evaluate MPA effectiveness could not be addressed due to a lack of data at the required spatial scale. These analyses, however, are important next steps to achieve a comprehensive evaluation of progress in marine protection and add immeasurable value for future MPA planning. MPAs will be of minimal value for biodiversity conservation if

they are placed in areas under little or no pressure from anthropogenic impacts. Therefore, it is crucial to incorporate a spatially explicit threat analysis as part of the MPA planning process. If MPAs are to be designed on the basis of systematic conservation planning, the knowledge of threats to species and ecosystems is central to assigning appropriate levels of protection. Previous approaches to localised threat analyses involve combining all available spatial layers into a threat index to obtain a cumulative effect score (Ban et al. 2010). Areas of highest impacts can therefore be identified and integrated into management strategies to reduce or manage impacts from existing human activities.

A key challenge for MPA planning is accounting for the impact of climate change on the marine environment. While there is increasingly good information about how these changes will impact marine environments, such as ocean acidification, more frequent bleaching of coral reefs and sea level rise changing coastal environments (Hoegh-Guldberg et al. 2007), there is a major gap in understanding how, or if, MPA planning can help mitigate these impacts. Recent advances, however, suggest that well managed, no-take MPAs have the potential to increase an ecosystem's resilience to some climate change impacts and slow adaptation in others (Roberts et al. 2017). Additionally, as changes in salinity and ocean currents continue to influence the redistribution of species, networks of no-take MPAs could serve as crucial stepping stones for dispersal and ultimately improve population connectivity (Roberts et al. 2017).

Aichi Target 11 specifies that MPAs must be effectively and equitably managed in order to achieve observable conservation benefits. Previous studies have clearly stated that MPAs cannot reach their full potential without active enforcement (Edgar et al. 2014) and sufficient resources for management (e.g. staff, equipment; Gill et al. 2017). MPAs globally are known for having a variable suite of management approaches, often varying to the point of mass public confusion and resulting difficulties for enforcement. Future research into Australian MPA performance should include a focus on broad scale management effectiveness to better interpret trends in the structure of the MPA system.

The aim for Chapter 5 was to determine the degree to which the general public understand the activities permitted within MPAs, and how this accords with the actual protection these areas offer to biodiversity. It was, however, not possible to determine the beliefs or expectations of the general public in regards to marine protection. Future work into MPAs

should aim to understand whether the general public support current levels of protection for MPAs or what levels of protection they believe MPAs should be providing to marine biodiversity. As Chapter 5 revealed substantial confusion regarding the objectives for MPAs, future research should also evaluate the possible impact that confusion among the general public has on effective enforcement of MPAs, as well as the perceived value of marine conservation.

As international targets continue to drive the rapid establishment of MPAs, these new insights, alongside the attributes examined in Chapters 2 through 5, would help design a more robust system of MPAs and ensure more favourable biodiversity outcomes. What is clear from this research is that total area is an inadequate measure of MPA effectiveness. Yet evaluating all parameters that reflect what is known about well-designed and effective MPAs was beyond the scope of this study. There is a clear need for a renewed focus on systematic conservation planning that incorporates all components of effective protected areas in general as countries continue to build towards targets.

Conclusions

Aichi Target 11 is meant to be interpreted in a much broader context than just total area. As the 2020 deadline looms closer, the challenge becomes achieving a holistic implementation of this target (Spalding et al. 2016). Chapters 3 through 5 were able to evaluate several features listed in Aichi Target 11 using the Australian MPA system as a case study: total area, representation and connectivity. Although Australia has surpassed the 10% target of area protected and should be acknowledged for the effort, a more nuanced picture of progress reveals that this target was met through the establishment of large, multi-use MPAs, often far offshore, with insufficient attention paid to the implications for biodiversity. Although the cost and complexities associated with no-take MPA establishment in coastal, high-use areas are limiting, the outcomes of this effort can be immensely positive for biodiversity conservation in the long run and often have trickle down impacts for fisheries, tourism, and economic investments.

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APPENDICES

Appendix A. Supplementary material for chapter 2: 'Measuring progress in marine protection: A new set of metrics to evaluate the strength of marine protected area networks'

Appendix B. Supplementary material for chapter 3: 'Bio-physical models of marine environments reveal biases in the representation of protected areas'

Appendix C. Supplementary material for chapter 4: 'Marine protected areas often fail at safeguarding ecological connectivity: An Australian case study'

Appendix D. Supplementary material for chapter 5: 'Evaluating perceptions of marine protection in Australia: does policy match public expectation?'

APPENDIX A

Marine components of terrestrial PAs

Terrestrial PAs also protect the marine environment by extending across the intertidal zone. These marine components can account for up to 70% of the terrestrial PA (e.g., island reserves). Therefore, we included these areas in our evaluation of the MPA network. To capture the marine components of terrestrial PAs, the terrestrial CAPAD spatial layers for each corresponding time step were clipped to the Global Self-consistent, Hierarchical, High-resolution Shoreline (GSHHS) spatial layer for Australia to identify all areas of intertidal and marine protection. All the identified polygons were scrutinized to ensure they were not the result of corrections to the mapping of the coastline by removing those areas that were less than 1km distance from the shoreline. All remaining areas verified as marine components, by reviewing management plans and contacting the relevant state agencies, were included in subsequent analyses.

Spatial Analyses

To quantify any changes made to the NRSMPA during the study period (2002-2014), we conducted a series of spatial analyses of the, corrected shapefiles were sourced directly from relevant agencies for several large MPAs (the Great Barrier Reef Marine Park, the Great Barrier Reef Coast Marine Park, Moreton Bay and all the MPAs in Victoria's state waters). Other overlapping polygons reflected overlays of protection (e.g., aquatic reserves within MPAs) or overlap between jurisdictions (e.g., Commonwealth managed Great Barrier Reef Marine Park and state managed Great Barrier Reef Coast Marine Park). In these cases, a topology correction in ArcGIS was applied to assign overlapping polygons to the highest level of protection based on the IUCN category, or to the larger polygon in the case of equal protection. These corrections enabled the area in each protection category to be accurately calculated.

Spatial calculations

Spatial unions were performed between the corrected layers for each time step to identify any change in area or protection across the network. For verified area changes, the calculate

geometry function was used to obtain the area (km²) of the affected polygon. Changes to the level of protection were identified using a unique condition statement to determine where an upgrade or downgrade occurred between two time steps and then confirmed with relevant state agencies.

To calculate the proportion of each IMCRA provincial bioregion within the NRSMPA, we used the IMCRA spatial layer (Version 4.0). Geoprocessing tools in ArcGIS were used to calculate the proportion of each bioregion within MPAs, and the proportion of each bioregion within each protection category, to quantify the changes in the representation and levels of protection for different bioregions over time.

Changes to the level of protection for marine biodiversity

In total, 8 MPAs (affecting 110,388 km²) had their protection upgraded as a result of rezoning or when small reserves were encompassed within the boundaries of new MPAs. In two cases, these upgrades were the result of strategic rezoning exercises for specific MPAs. In Moreton Bay Marine Park in Queensland, no-take areas increased from 0.5% to 16% coverage (DNPSR 2008), while the Great Barrier Reef Marine Park (GBRMP) rezoning increased no-take zones from 5% to 33% of the MPA. The other six upgrades occurred when aquatic reserves were encompassed within the boundaries of new MPAs, increasing their level of protection from Category IV to Category II (DEWNR 2012).

A total of 33 MPAs (affecting 2,839 km²) had their protection downgraded during the study period. Three downgrades were the result of formal rezoning in two MPAs in NSW (Solitary Islands and Jervis Bay Marine Parks) that initially increased protection but were later repealed via legislative amendment following a change in government in 2011 (NSW Government 2011). A further 30 sanctuary zones within 5 MPAs were the subject of a policy downgrade, whereby in 2013 the NSW Government changed the law to permit recreational shore fishing, reducing the protection for sanctuary zones from Category IA to Category IV. In December 2014, after another change in government, this change was repealed for 20 of the 30 affected sanctuary zones (NSW Government 2014).

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Table A1. Full breakdown of permitted activities within Australian jurisdictions and assigned IUCN category

ZONE ID	ABBREVIATION	ASSIGNED IUCN
Sanctuary Zone (Permit only)/Restricted Access Zone	SZP	IA
Sanctuary Zone	SZ	II
Marine National Park Zone (VIC & COM)	MNP	II
Habitat Protection (NT)	HP_PE	IV
Recreational Use Zone (WA & COM)	RUZ	IV
Habitat Protection Zone	HPZ	IV
Marine Park Zone (VIC)	MPZ	IV
General Use Zone	GUZ	VI
Special Purpose Zone	SPZ	VI
Not specified	NS	X

ACTIVITY	NSW	VIC	TAS	SA	WA	NT (1 MPA)	QLD	COM
Commercial line fishing	HPZ GUZ SPZ	MPZ (park basis)	reserve restricted	HPZ GUZ	RUZ (not SBMP) GUZ SPZ	GUZ	HPZ GUZ	HPZ GUZ
Trapping	HPZ GUZ SPZ	NS	reserve restricted	HPZ GUZ	NS	NS	NS	GUZ
Longline fishing	X	NS	reserve restricted	HPZ GUZ	NS	GUZ	NS	HPZ GUZ
Dropline fishing	GUZ SPZ	NS	reserve restricted	HPZ GUZ	NS	NS	NS	HPZ GUZ
Netting	GUZ	NS	reserve restricted	HPZ GUZ	GUZ	GUZ	HPZ GUZ	HPZ GUZ
Trawling	GUZ	NS	X	GUZ	NS	GUZ	GUZ	X
Spearfishing	HPZ GUZ SPZ	MPZ (park basis)	reserve restricted	HPZ GUZ	RUZ GUZ SPZ	GUZ	HPZ GUZ	RUZ HPZ GUZ
Recreational fishing	HPZ GUZ SPZ	MPZ (park basis)	reserve restricted	HPZ GUZ	RUZ GUZ SPZ	HP_PE GUZ	HPZ GUZ	RUZ HPZ GUZ
Anchoring	SZ HPZ GUZ SPZ	SZ MNP MPZ	all reserves	SZ HPZ GUZ	SZ RUZ GUZ SPZ	SZ HP_PE GUZ	NS	NS
Boating	SZ HPZ GUZ SPZ	SZ MNP MPZ	all reserves	SZ HPZ GUZ	SZ RUZ GUZ SPZ	SZ HP_PE GUZ	SZ HPZ GUZ	MNP RUZ HPZ GUZ
Scuba/snorkel	SZ HPZ GUZ SPZ	SZ MNP MPZ	all reserves	SZ HPZ GUZ	SZ RUZ GUZ SPZ	SZ HP_PE GUZ	SZ HPZ GUZ	MNP RUZ HPZ GUZ
Scientific collecting/research	SZP HPZ GUZ SPZ	NS	all reserves	SZP HPZ GUZ	SZ RUZ GUZ SPZ (permit)	SZ HP_PE GUZ	SZP SZ HPZ GUZ	SZ MNP RUZ HPZ GUZ
Recreational collecting	HPZ GUZ SPZ	MPZ (park basis)	reserve restricted	GUZ	X	NS	HPZ GUZ	NS
Mining operations	NS	NS	X	GUZ (limits)	GUZ (limits)	NS	NS	GUZ
Aquaculture and pearling	NS	NS	NS	HPZ GUZ	GUZ (permit)	GUZ (permit)	NS	HPZ GUZ

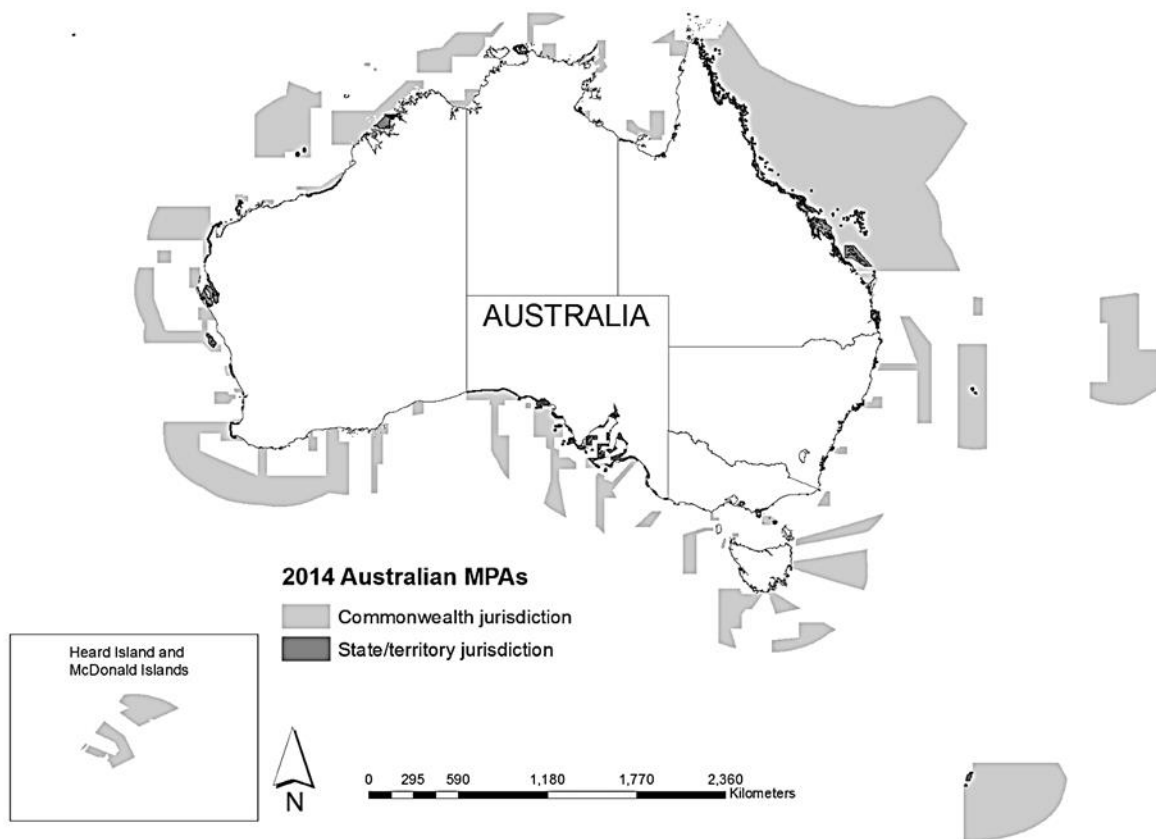


Figure A1. The distribution of Commonwealth versus state and territory managed marine protected areas as at 2014.

Table A2. Changes in the level of protection for marine protected areas across the Australian National Reserve System Marine Protected Area Network

<i>STATE/JURISDICTION</i>	# OF MPAS AFFECTED	APPROXIMATE AREA AFFECTED (km²)	ORIGINAL IUCN CATEGORY	FINAL IUCN CATEGORY
	UPGRADES			
<i>South Australia</i>	6 (aquatic reserves)	480	VI	II
<i>Queensland</i>	1	160	IV	II
<i>Commonwealth</i>	1	109,748	IV & VI	II
	DOWNGRADES			
<i>Northern Territory</i>	1	2,233	II	VI
<i>New South Wales</i>	32*	606	IA & II	IV & VI

*20 of the 30 downgraded sanctuary zones within 5 MPAs later had protection re-established to IA

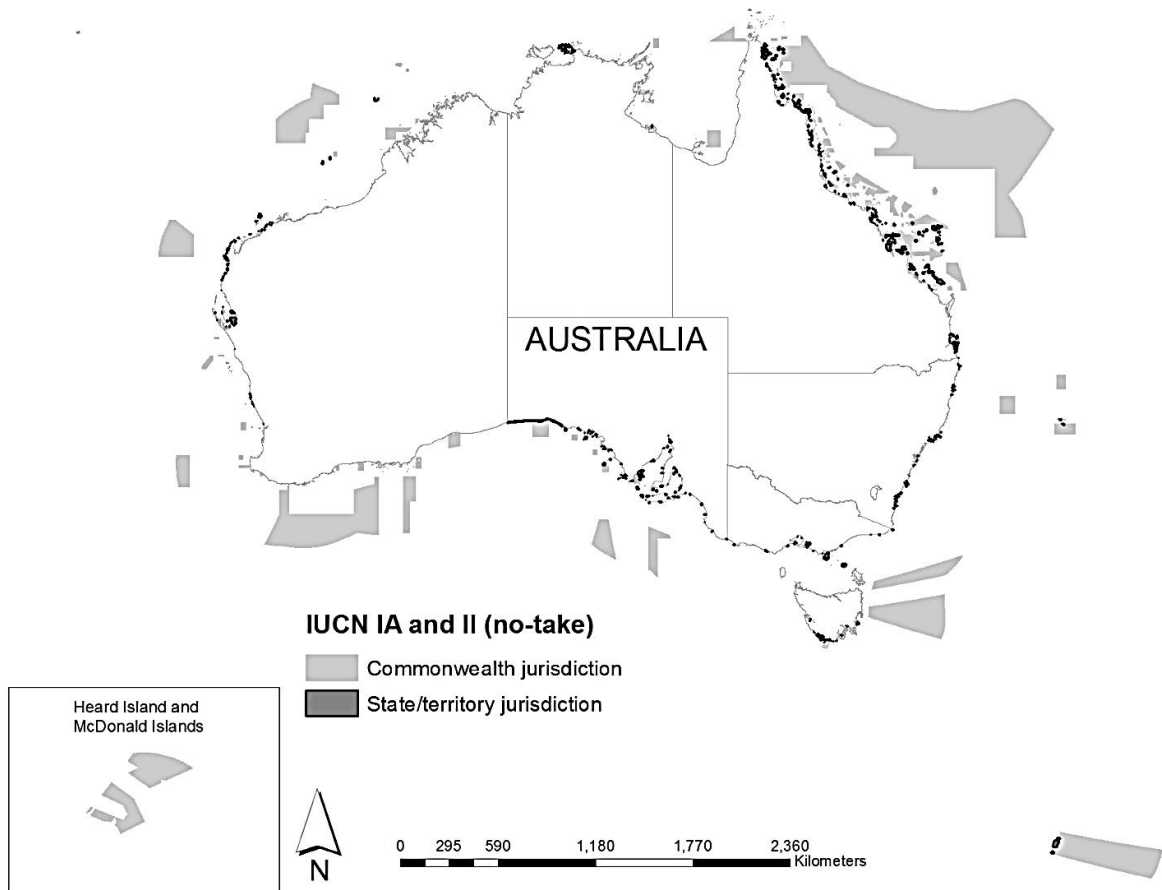


Figure A2. The distribution of no-take marine protected areas across Australia waters as at 2014.

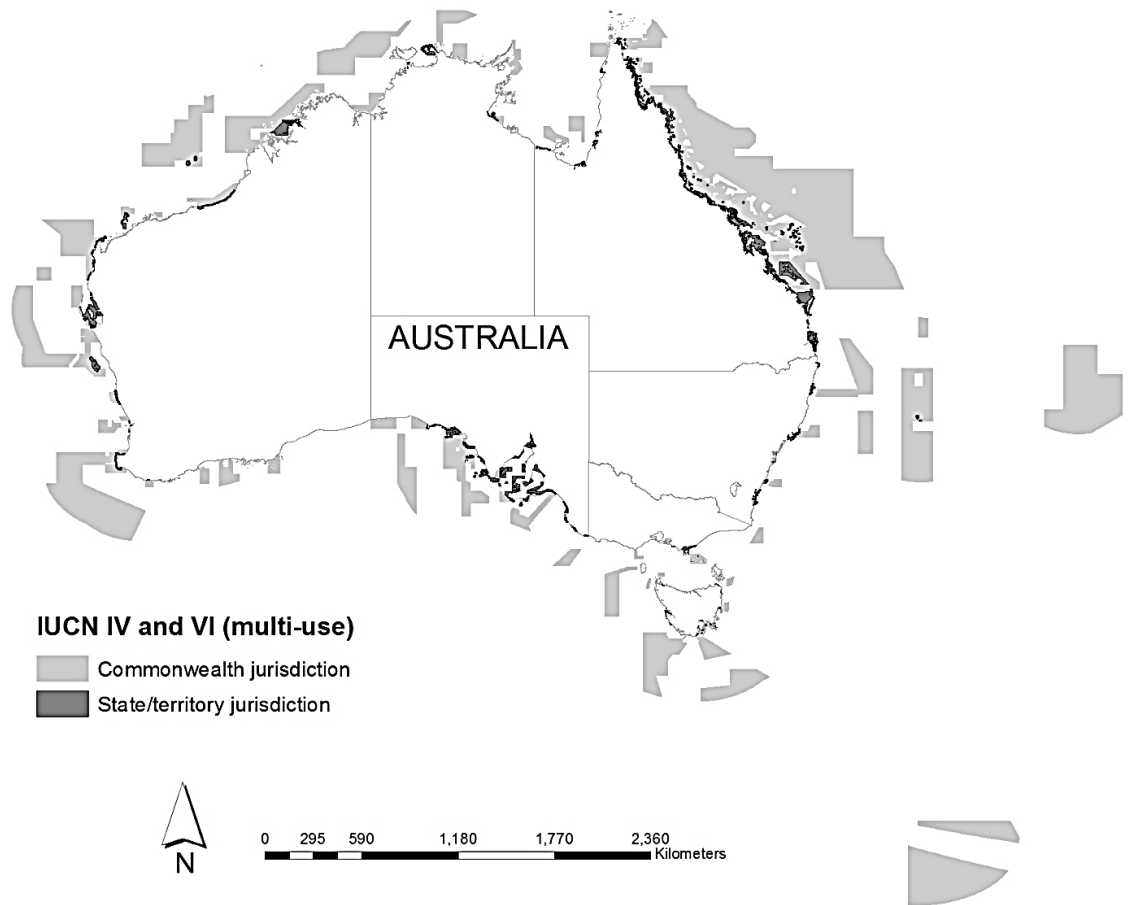


Figure A3. The distribution of multi-use marine protected areas across Australia waters as at 2014.

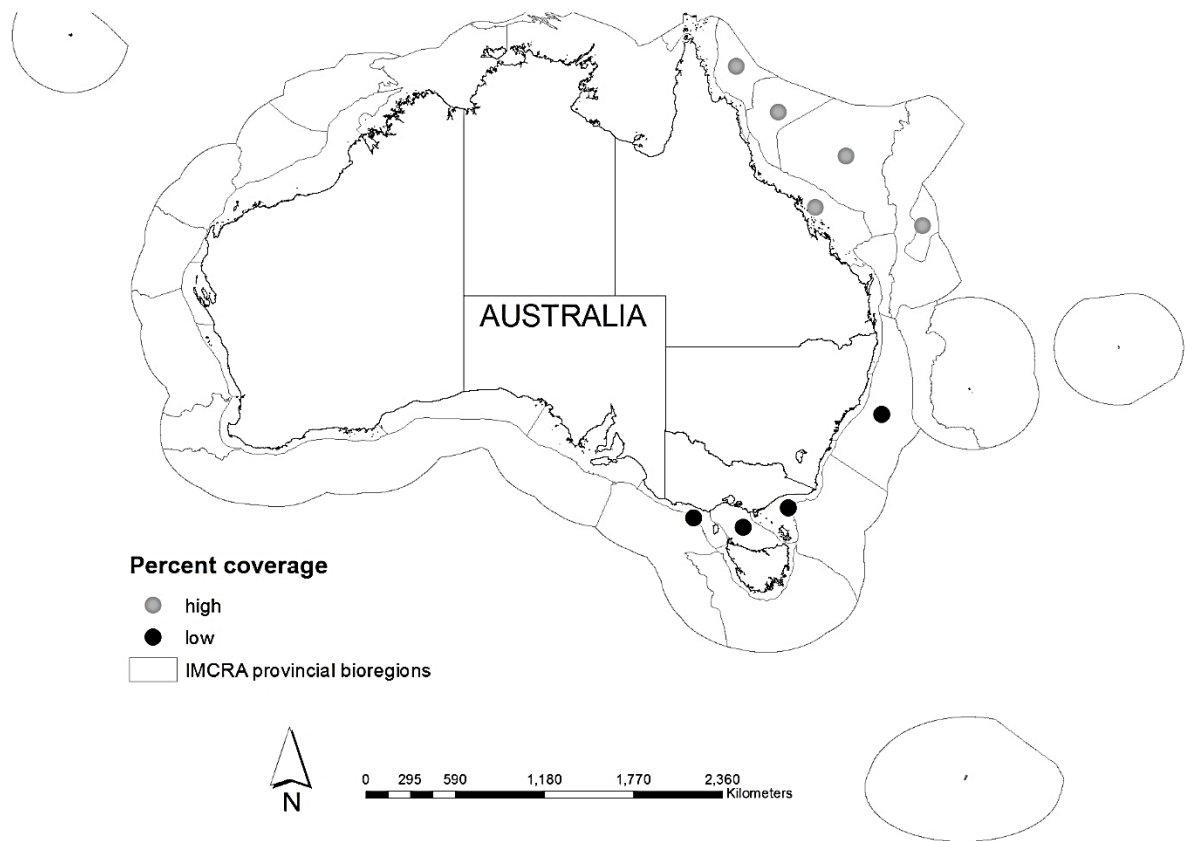


Figure A4. Current distribution of well represented (>75% of the bioregion protected) and poorly represented (<10% of the bioregion protected) IMCRA Provincial Bioregions within the Australian MPA network.

APPENDIX B

Incorporating spatial data for benthic substrate into the biophysical model

To standardise spatial data for substrate, we generated a substrate index (1-9) to classify benthic substrate from softest (i.e., mud, silt, sand) to hardest (i.e., rock, reef). As data were retrieved from numerous sources across Australia, there were many inconsistencies with how boundary lines were drawn in ArcGIS. When an overlap occurred between the state and federal data, condition statements in raster calculator allowed for priority to be assigned to the higher resolution state mapping data.

Table B1. Substrate index incorporated into the biophysical model

ORDINAL SCALE	SUBSTRATE TYPE
1	Calcareous ooze
2	Silt or mud
3	Mud and clay mixture
4	Muddy sand
5	Hard or consolidated sand
6	Mixture of sand and rubble
7	Cobble, gravel, or rubble
8	Rock
9	Rocky or coral reef

Table B2. Spatial data incorporated into IMCRA 4.0 versus biophysical model

IMCRA DATA	BIOPHYSICAL MODEL DATA
	BENTHIC SUBSTRATE
SEA SURFACE TEMPERATURE (SST)	SEA SURFACE TEMPERATURE (SST)
SALINITY	SALINITY
CHLOROPHYLL_A	CHLOROPHYLL_A
BATHYMETRY	BATHYMETRY
SEA SURFACE HEIGHT	
NITRATE	
PHOSPHATE	
SILICATE	
DISSOLVED OXYGEN	
WAVE HEIGHT	
CURRENTS	
TIDES	
BIOGEOGRAPHY OF DEMERSAL FISH	

Table B3. Correlation matrix of the biophysical variables chosen for the PCA.

	SST	SUB	CHL_A	SSS	DEPTH
SST	1.000	.286	.173	-.899	.345
SUB	.286	1.000	.409	-.278	.539
CHL_A	.173	.409	1.000	-.200	.344
SSS	-.899	-.278	-.200	1.000	-.386
DEPTH	.345	.539	.344	-.386	1.000

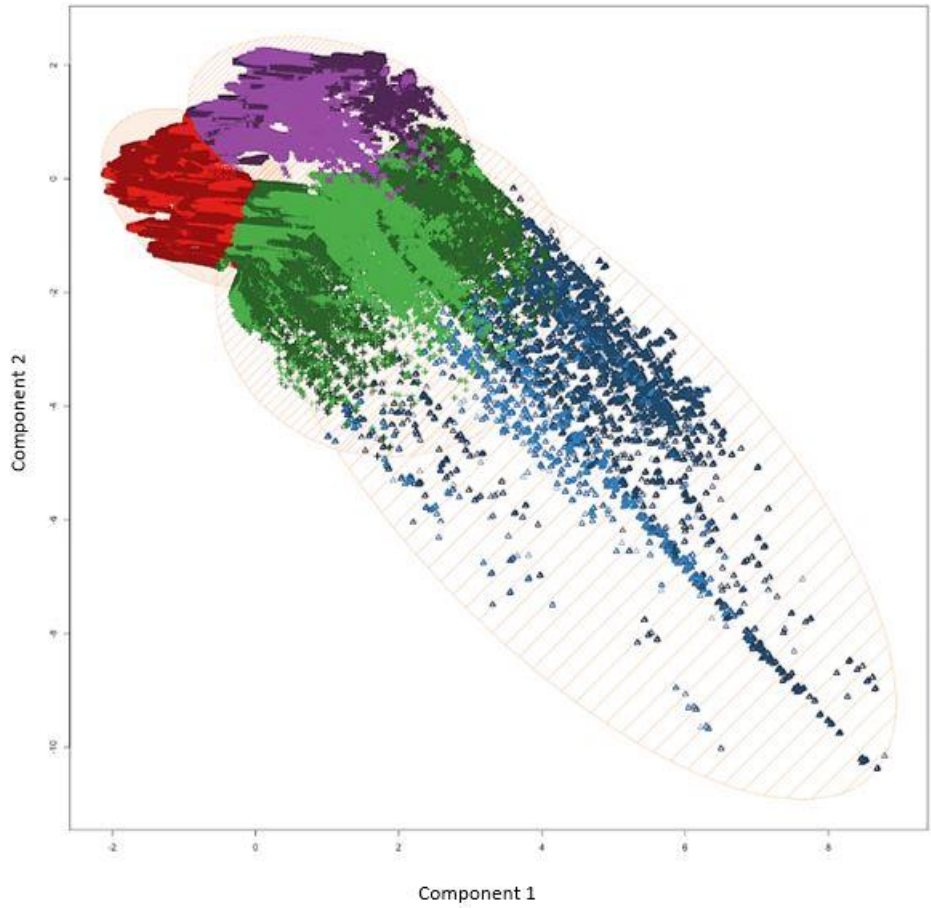


Figure B1. Bi-plot resulting from K-means cluster analysis. MPAs per cluster are represented by the darker equivalents of their respective colors.

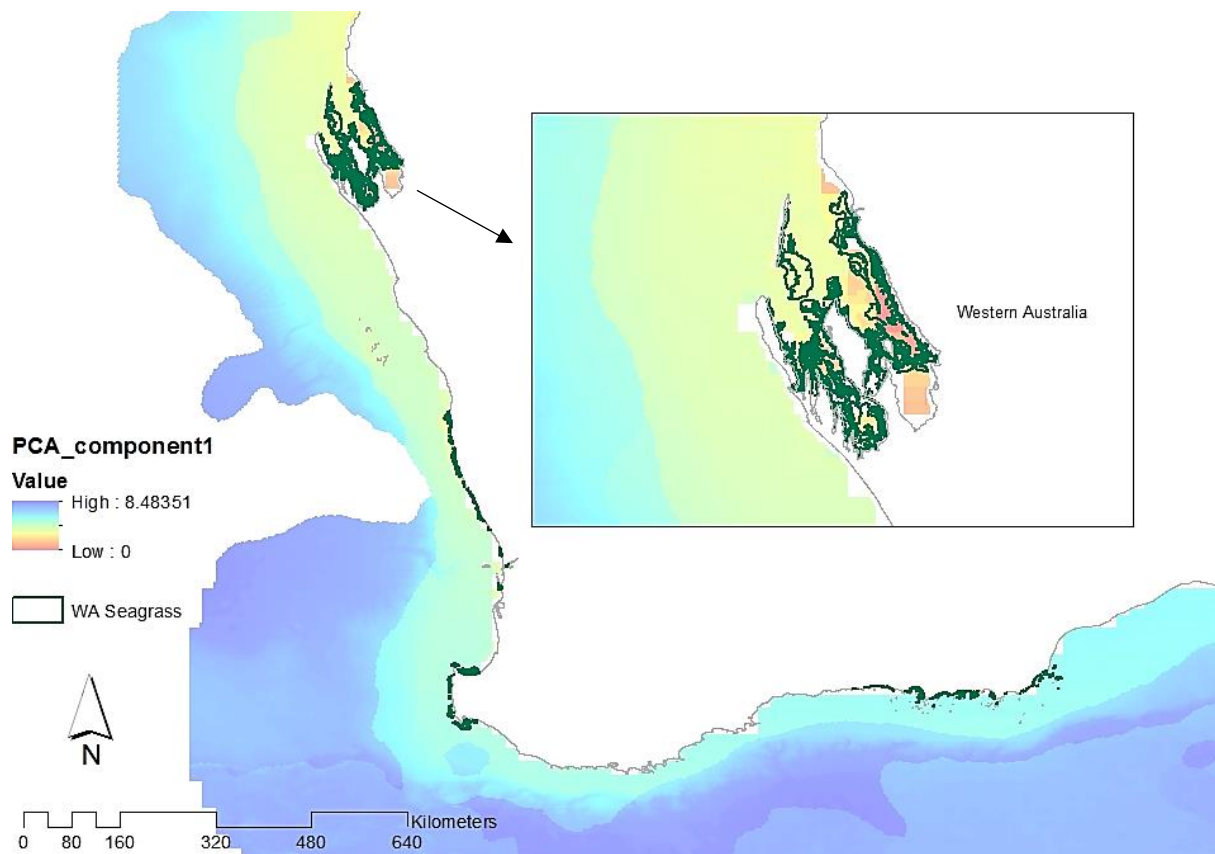


Figure B2. PC1 raster layer with ground-truthed locations of seagrass in Western Australia

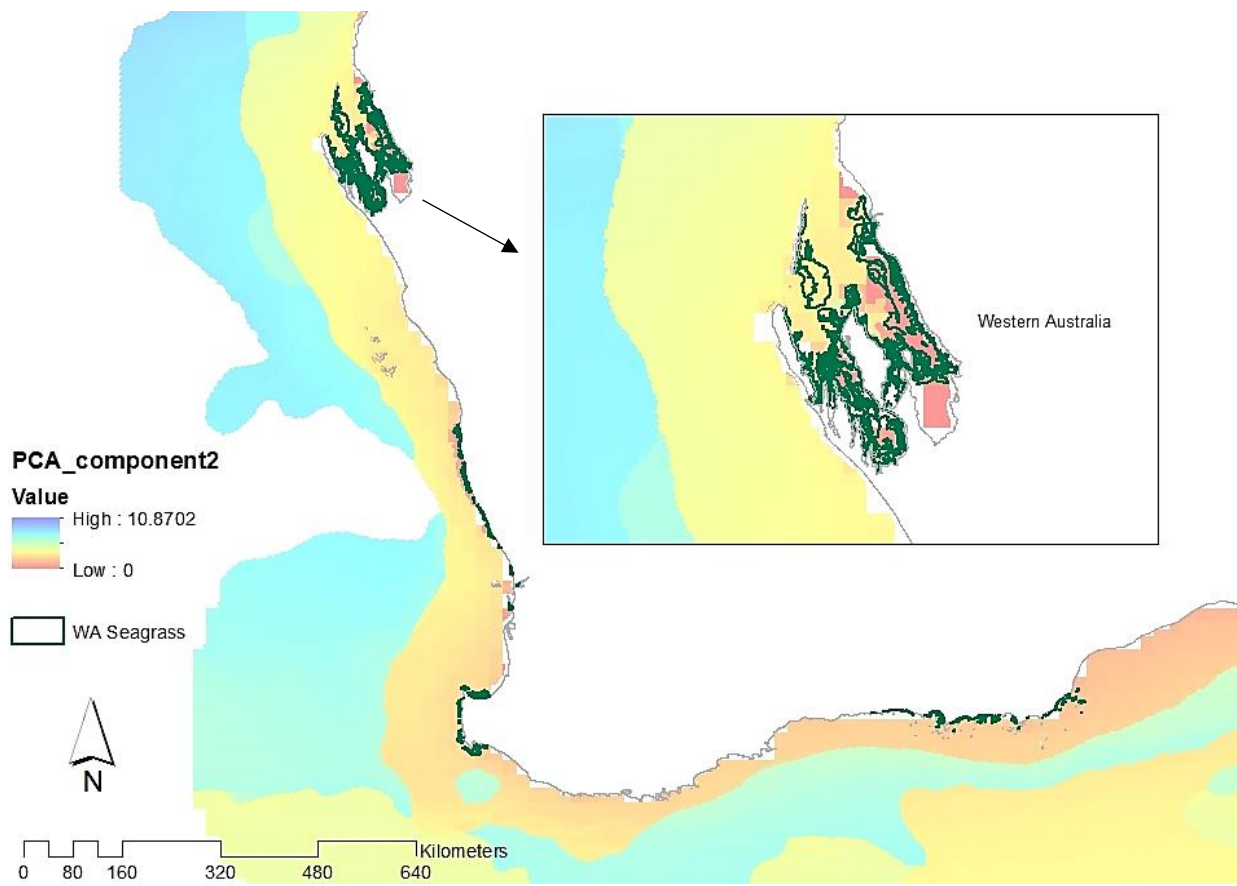


Figure B3. PC2 raster layer with ground-truthed locations of seagrass in Western Australia

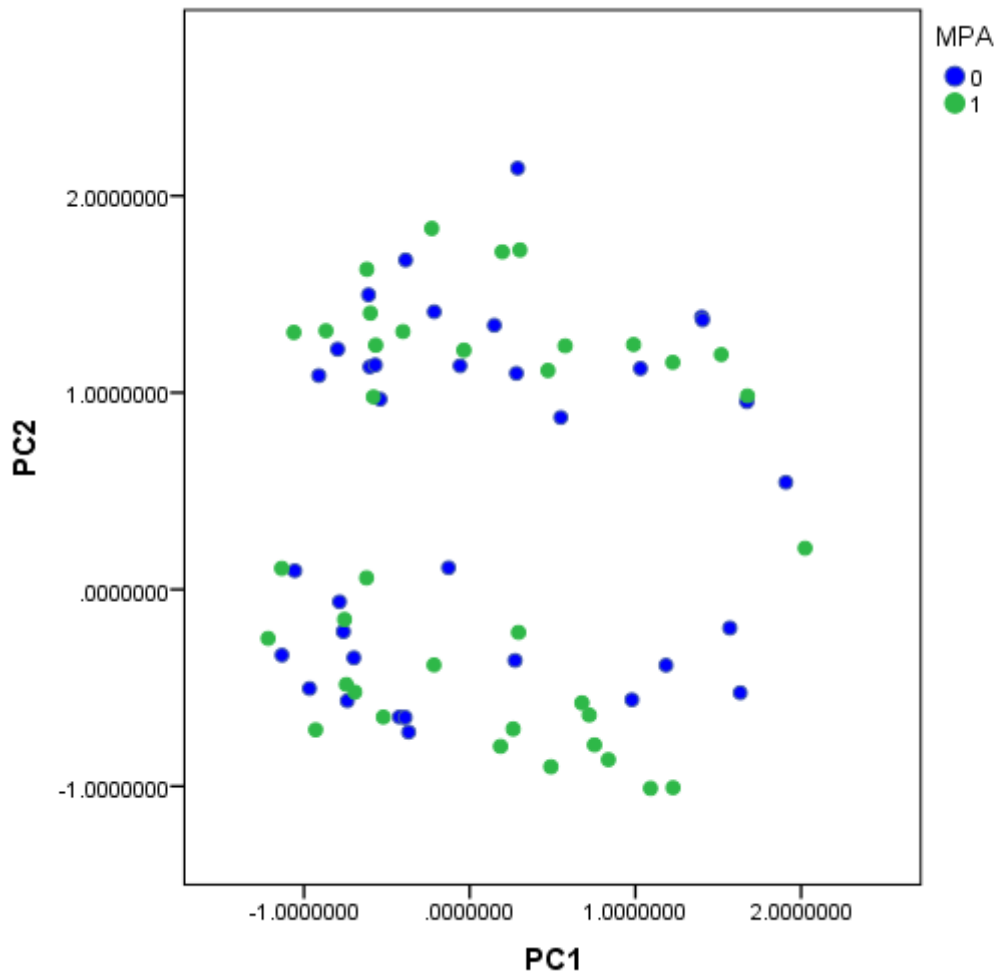


Figure B4. Scatterplot distribution of the centroid (mean value) of all 40 bioregions separated by MPA (1, green dot) and non-MPA (0, blue dot).

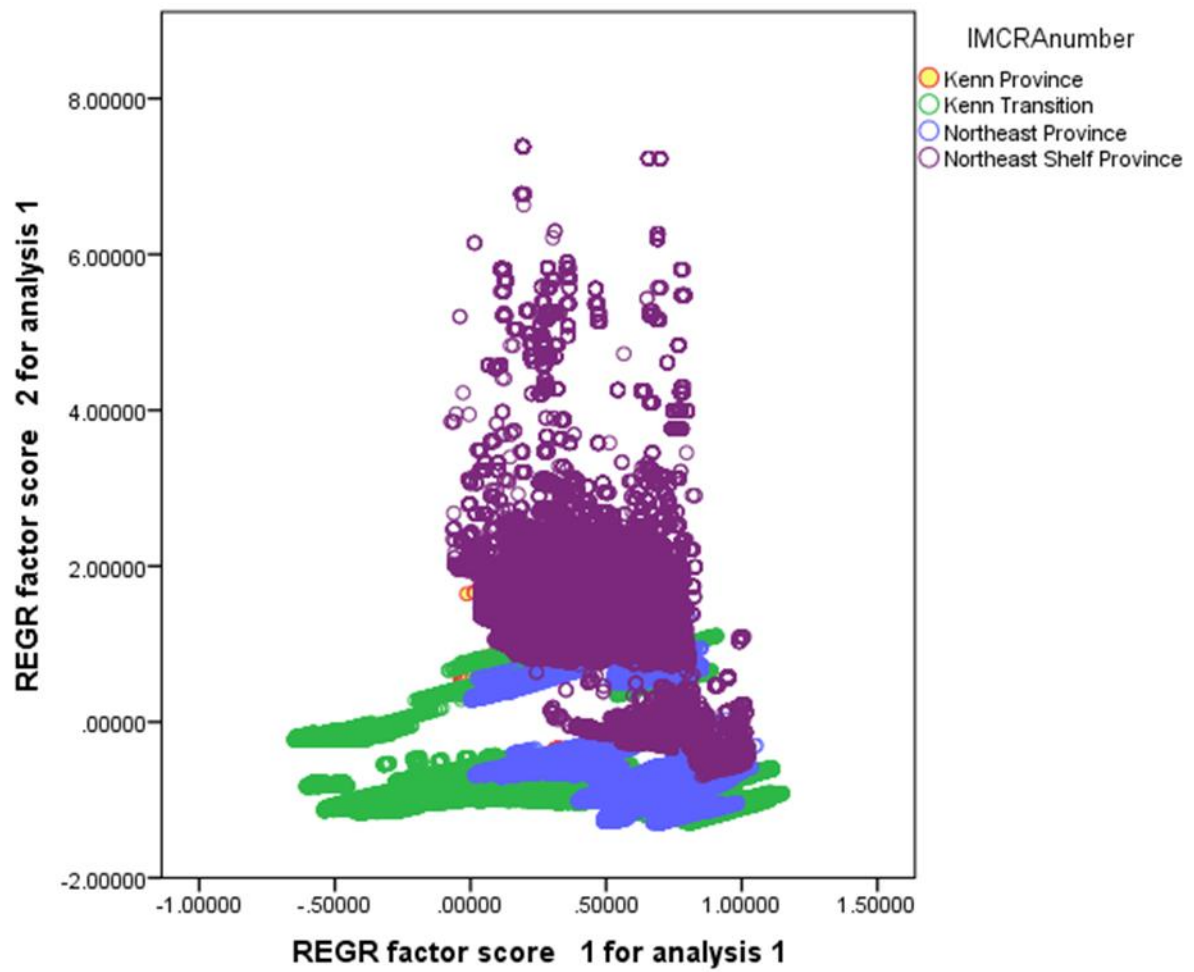


Figure B5. Scatterplot distribution of 4 IMCRA provincial bioregions located from the Great Barrier Reef out to the Coral Sea in North Queensland alongside PC1 and PC2

APPENDIX C

Methods

Analyses at the ecoregion scale

By summarizing the results at the ecoregion scale, we were able to create a normalised scenario around the Australian EEZ where each ecoregion is evaluated individually for functional connections between MPAs. This was necessary due to the extensive Great Barrier Reef Marine Park, which accounts for a significant amount of Australia's reef habitat and heavily influenced some overall results. Additionally, as ecoregions represent ecologically distinct zones in the marine environment, it is more important for connectivity to be protected within rather than between ecoregions. Therefore, results can better represent the degree of connectivity for species with more localised distributions or narrower habitat requirements.

Results

Table C1. Comparison of similar taxa PLD (in days) to modelled species *urchin*

FAMILY	SCIENTIFIC NAME	PLD	SOURCE
Ophiochitonidae	<i>Ophionereis schayeri</i>	7	Falkner & Byrne 2003
Anthozoa	<i>Alcyonium siderium</i>	4.5	Shanks 2009
Cladorhizidae	<i>Stylopsus spp.</i>	2.5	Ayling 1980
Onchidorididae	<i>Adalaria proxima</i>	2	Shanks 2009
Vermetidae	<i>Dendropoma corallinaceum</i>	3.5	Hughes 1978
Acanthasteridae	<i>Acanthaster planci</i>	14	Moran et al. 1992
Echinometridae	<i>Heliocidaris erythrogramma</i>	5	Williams & Anderson 1975
Balanidae	<i>Balanus glandula</i>	14	Schwindt 2007
Muricidae	<i>Rapana venos</i>	14	Shanks 2009

Table C2. Comparison of similar taxa PLD (in days) to modelled species *damselfish*

FAMILY	SCIENTIFIC NAME	PLD	SOURCE
Pomacentridae	<i>Chromis cyanea</i>	29	Wellington & Victor 1989
Pomacentridae	<i>Centropyge tibicen</i>	30	Brothers & Thresher 1985
Pomacentridae	<i>Plectroglyphidodon sindonis</i>	30	Wellington & Victor 1989
Pomacentridae	<i>Chromis alpha</i>	30	Wellington & Victor 1989
Pomacentridae	<i>Chromis hanui</i>	27	Wellington & Victor 1990
Pomacentridae	<i>Chromis insolata</i>	20	Wellington & Victor 1991
Pomacentridae	<i>Dascyllus reticulatus</i>	20	Wellington & Victor 1992
Pomacentridae	<i>Chromis xanthura</i>	26	Wellington & Victor 1993
Pomacentridae	<i>Chromis traceyi</i>	23	Wellington & Victor 1994
Pomacentridae	<i>Stegastes lividus</i>	25	Wellington & Victor 1995

Table C3. Comparison of similar taxa PLD (in days) to modelled species *wrasse*

FAMILY	SCIENTIFIC NAME	PLD	SOURCE
Labridae	<i>Halichoeres semicinctus</i>	29	Victor & Wellington 2000
Labridae	<i>Macropharyngodon meleagris</i>	30	Brothers & Thresher 1985
Labridae	<i>Coris batuensis</i>	30	Froese & Pauly 2006
Labridae	<i>Cirrhilabrus temminki</i>	30	Brothers & Thresher 1985
Labridae	<i>Bodianus mesothorax</i>	30.3	Victor 1986
Labridae	<i>Halichoeres nicholsi</i>	30.4	Victor & Wellington 2000
Labridae	<i>Epibulus insidiator</i>	30.4	Victor 1986
Labridae	<i>Labropsis xanthonota</i>	30.5	Victor 1986
Labridae	<i>Halichoeres maculipinna</i>	30.5	Schultz & Cowen 1994
Acanthuridae	<i>Acanthurus nigrofuscus</i>	31	Wilson & McCormick 1999

Table C4. Comparison of similar taxa PLD (in days) to modelled species *trevally*

FAMILY	SCIENTIFIC NAME	PLD	SOURCE
Carangidae	<i>Caranx melampygus</i>	30	Kim et al. 2001
Polynemidae	<i>Polydactylus sexfilis</i>	30	Kim et al. 2001
Carangidae	<i>Caranx ignobilis</i>	40	Sudekum et al. 1991
Scorpaenidae	<i>Pterois volitans</i>	40	Shanks 2009
Carangidae	<i>Caranx ignobilis</i>	40	Leis et al. 2001
Carangidae	<i>Pseudocaranx dentex</i>	45	Paxton et al. 1989
Mullidae	<i>Mulloidichthys flavolineatus</i>	45	Longenecker et al. 2008
Mullidae	<i>Parupeneus porphyreus</i>	47	Longenecker et al. 2008

APPENDIX D

Methods

Table D1. Breakdown of recreational or commercial activities permitted in each zone

Level of protection	No-take zones		Extractive use zones	
	IA	II	IV	VI
Activity				
Research	✓	✓	✓	✓
Scuba		✓	✓	✓
Boating		✓	✓	✓
Anchoring		✓	✓	✓
Recreational fishing			✓	✓
Recreational collecting*			✓	✓
Commercial fishing*			✓	✓
Aquaculture*			✓	✓
Dropline fishing*			✓	✓
Spearfishing*			✓	✓
Longline fishing*			✓	✓
Netting*			✓	✓
Trapping*			✓	✓
Mining ⁺				✓
Trawling ⁺				✓

*= varies by jurisdiction whether permitted in IUCN Category IV

+ = activity permitted only under license or permit and varies by jurisdiction

Spatial calculations for determining downgrades to Australian MPA system 2014-2018

Data for zoning arrangements were retrieved from the 2014 CAPAD database and from the Parks Australia webpage for 2018 (<https://parksaustralia.gov.au/marine/maps/>). At the time of this study, the CAPAD data for 2018 had not been compiled and reported policy changes were only available on an individual park basis. Therefore, spatial joins were performed in ArcGIS to obtain exact area changes in zoning between the 2014 and 2018 Australian MPA system. These changes were reported as a single value for the downgrades to protection across the whole system.

How protected are the Marine Protected Areas of Australia?

Please note you must be 18 years old or older to complete this survey

Please fill in the following details:

Sex: Male Female Country of residence: _____ Postcode: _____

Level of Education:

- Less than Year 12 or equivalent
- Year 12 or equivalent
- Diploma or certificate
- Bachelor degree (including with honours)
- Postgraduate degree (Masters, Doctorate)
- Other - please specify

A Marine Protected Area is defined by the Department of the Environment as 'an area of the sea especially dedicated to the protection and maintenance of biodiversity, and of natural and associated cultural resources, and managed through legal or other effective means.'

Please indicate which of the following activities you think are permitted or prevented within the above definition of a marine protected area. If unsure about a particular activity, please see definitions below table for further detail.

ACTIVITY	PERMITTED	PREVENTED
Scientific collection/research	<input type="checkbox"/>	<input type="checkbox"/>
Anchoring a boat	<input type="checkbox"/>	<input type="checkbox"/>
Boating	<input type="checkbox"/>	<input type="checkbox"/>
Scuba diving/snorkelling	<input type="checkbox"/>	<input type="checkbox"/>
Commercial line fishing	<input type="checkbox"/>	<input type="checkbox"/>
Trapping	<input type="checkbox"/>	<input type="checkbox"/>
Longline fishing	<input type="checkbox"/>	<input type="checkbox"/>
Dropline fishing	<input type="checkbox"/>	<input type="checkbox"/>
Spearfishing	<input type="checkbox"/>	<input type="checkbox"/>
Recreational fishing	<input type="checkbox"/>	<input type="checkbox"/>
Recreational collecting	<input type="checkbox"/>	<input type="checkbox"/>
Aquaculture and pearling	<input type="checkbox"/>	<input type="checkbox"/>
Netting	<input type="checkbox"/>	<input type="checkbox"/>
Trawling	<input type="checkbox"/>	<input type="checkbox"/>
Mining operations	<input type="checkbox"/>	<input type="checkbox"/>

Longline fishing: baited hooks are attached to the longline by short lines called snoods that hang from the mainline; targets large pelagic species such as tuna or billfish

Dropline fishing: mainline with an anchor at one end and a float at the other and short baited hooks; targets species that live on or near the seafloor

Trawling: nets shaped like a cone or funnel are towed by a boat through the water column or along the sea floor; collects all species in its path

Results

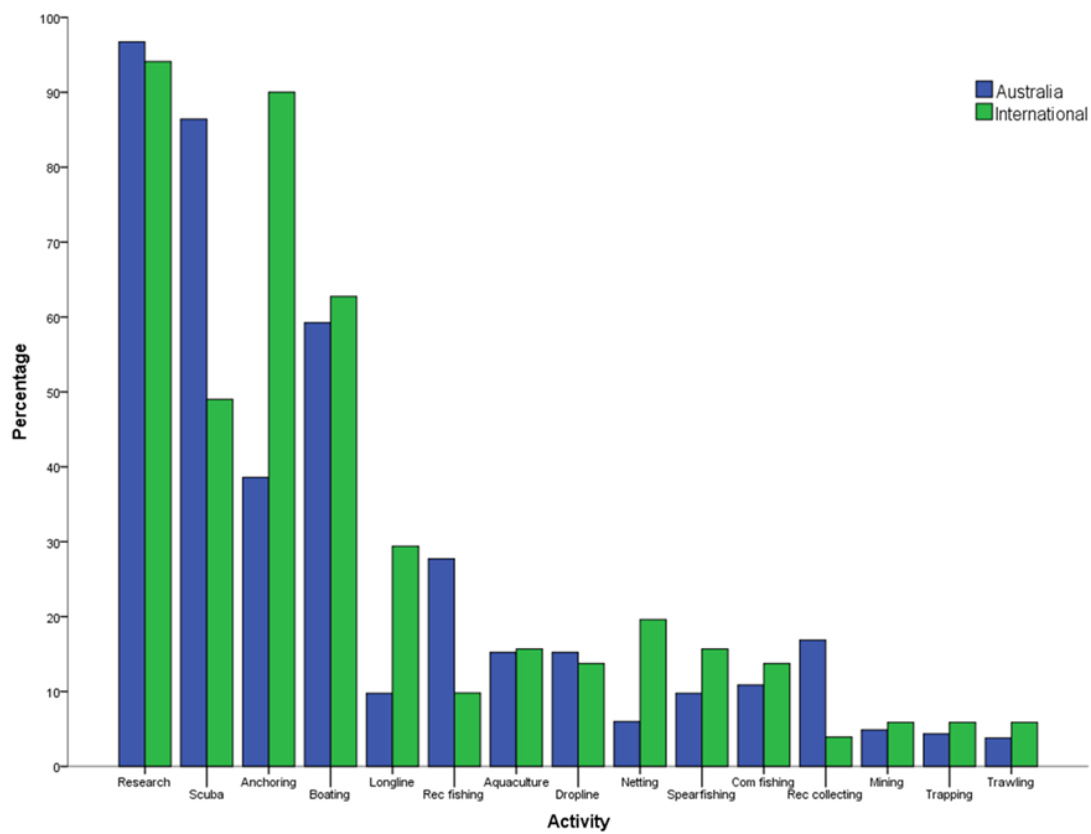


Figure D1. The proportion of Australian and international respondents that indicated permitted for each activity.

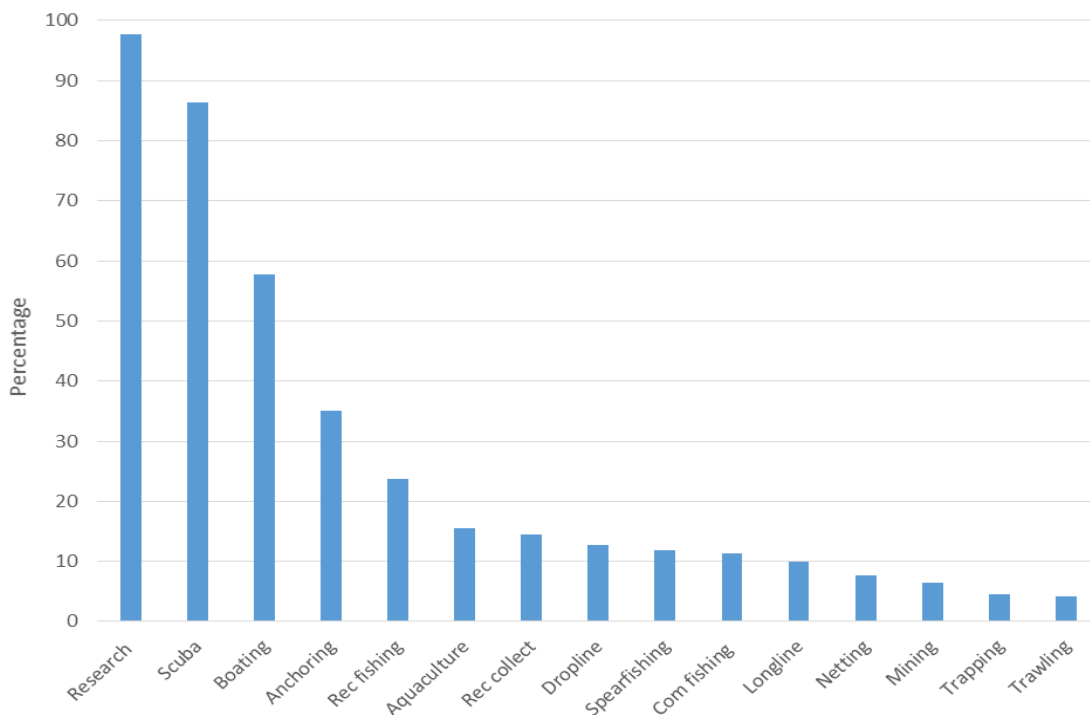


Figure D2. The proportion of respondents from VICTORIA that indicated permitted for each activity

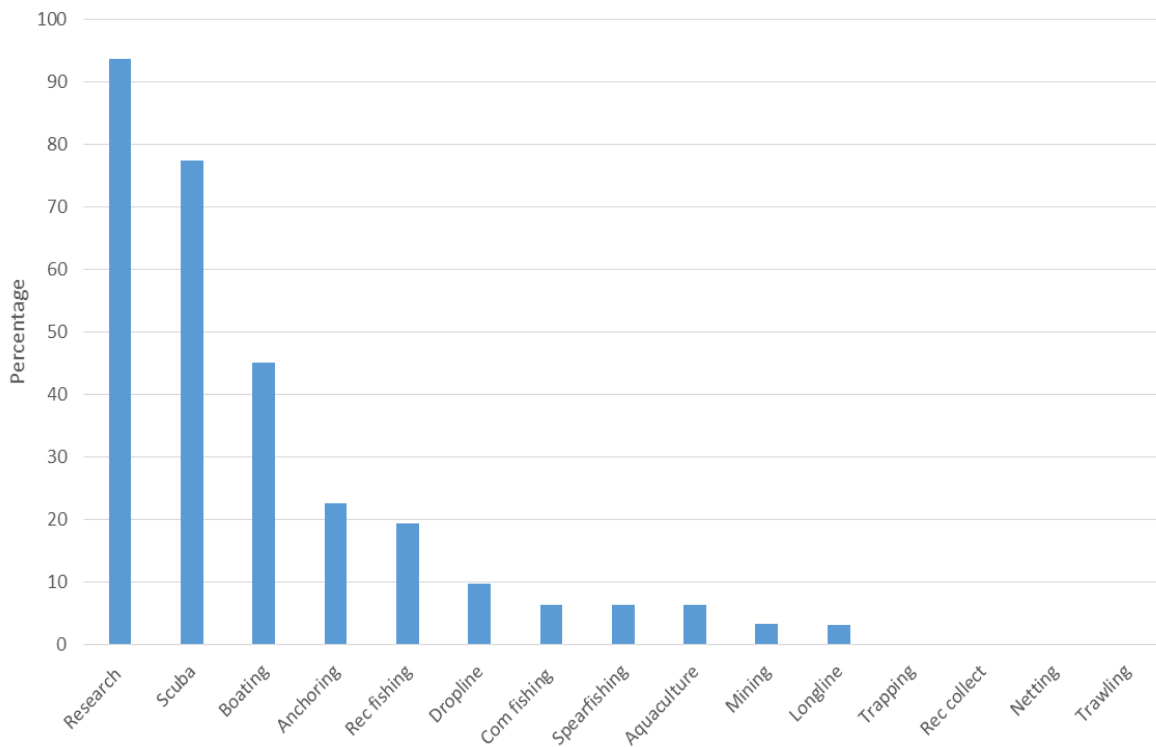


Figure D3. The Proportion of respondents from NEW SOUTH WALES that indicated permitted for each activity

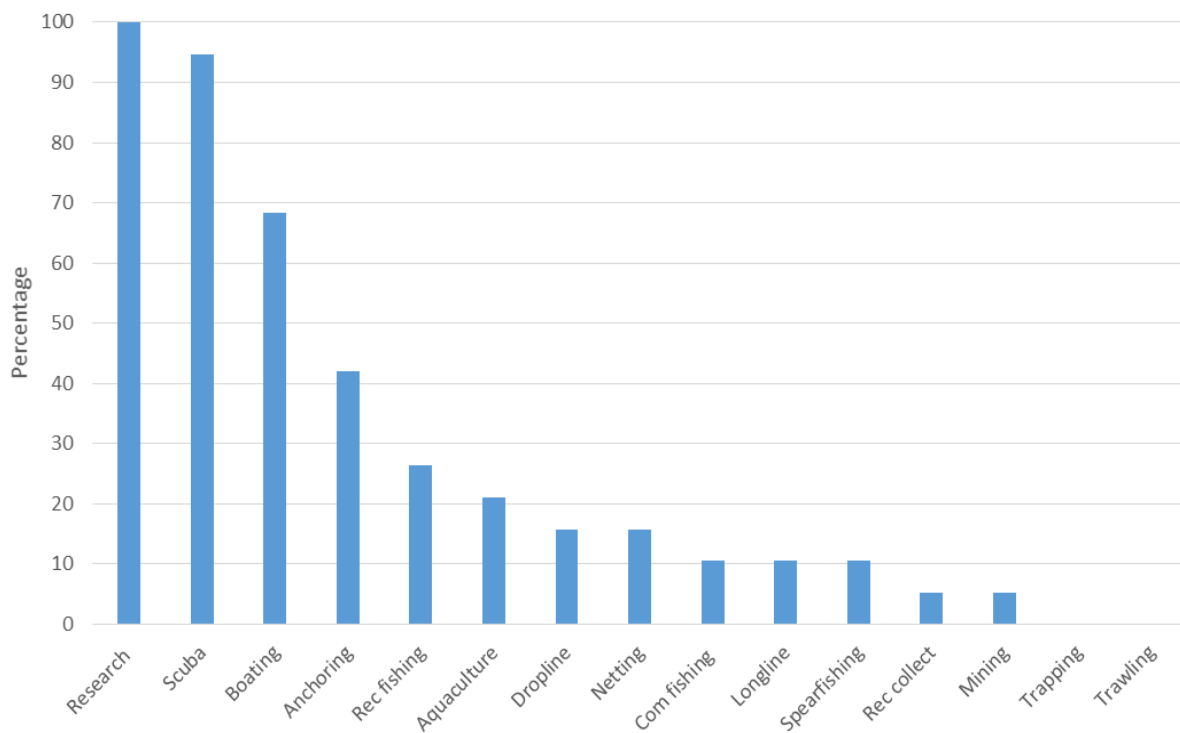


Figure D4. The Proportion of respondents from QUEENSLAND that indicated permitted for each activity

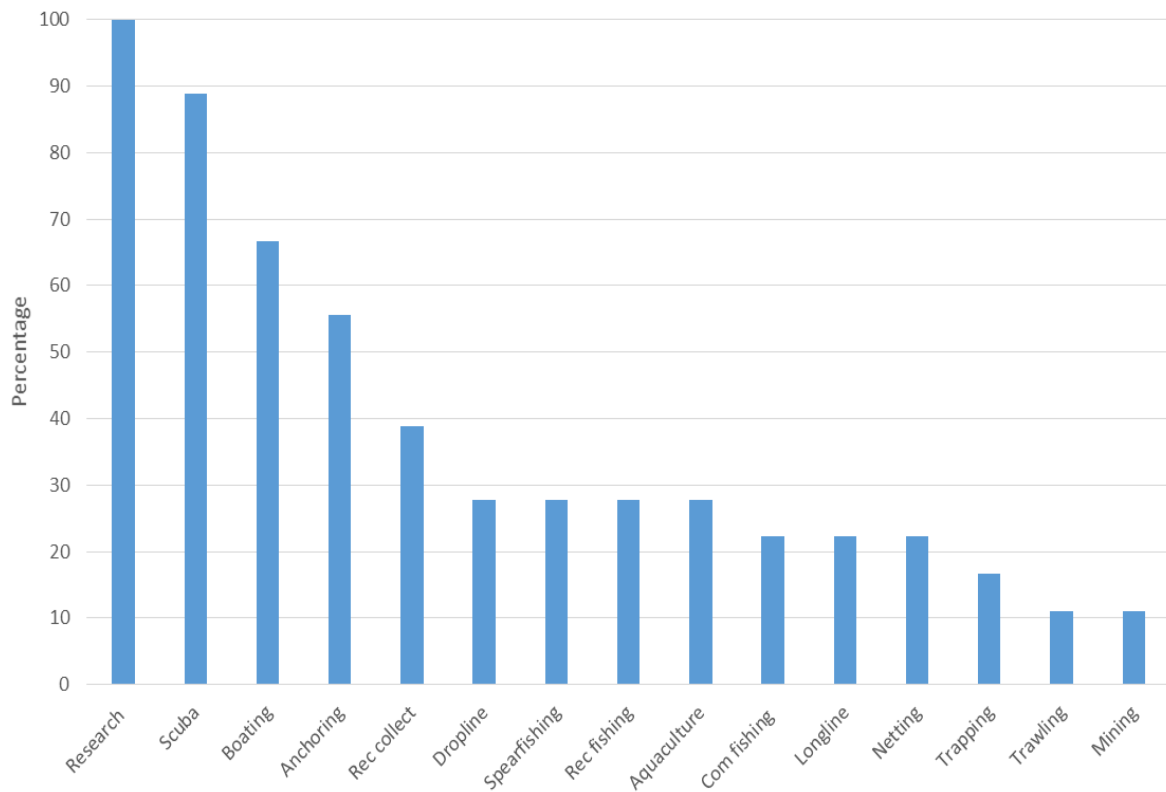


Figure D5. The proportion of respondents from SOUTH AUSTRALIA that indicated permitted for each activity

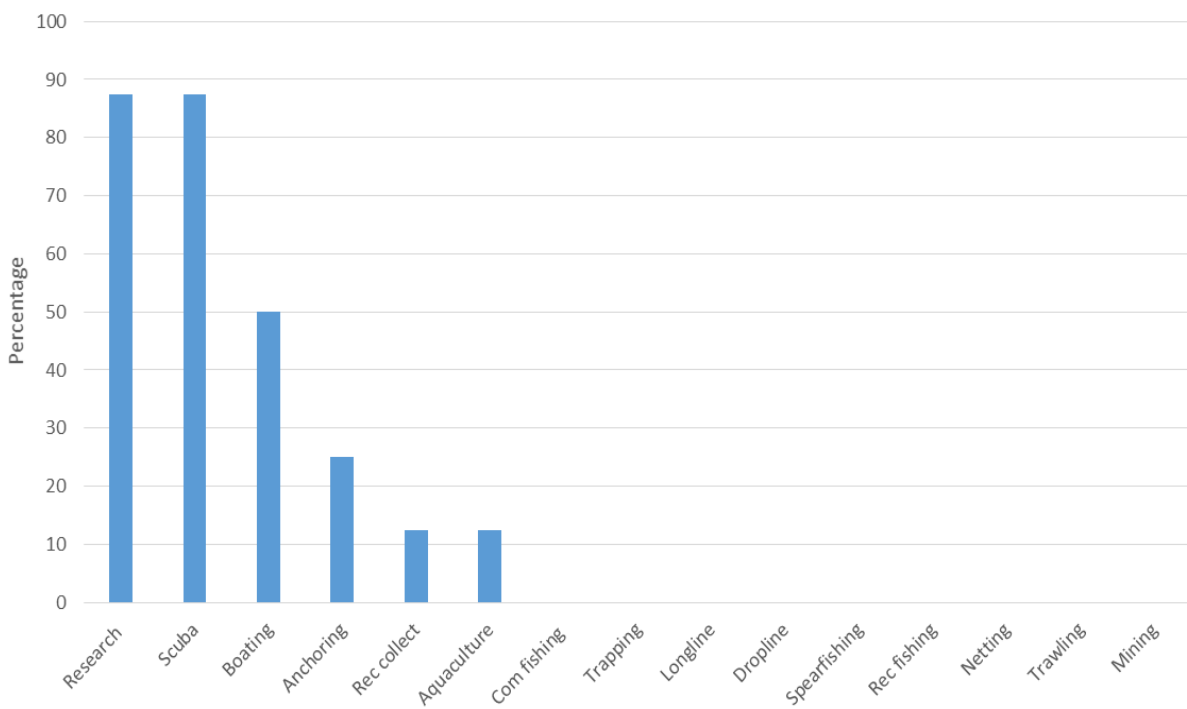


Figure D6. The proportion of respondents from WESTERN AUSTRALIA that indicated permitted for each activity

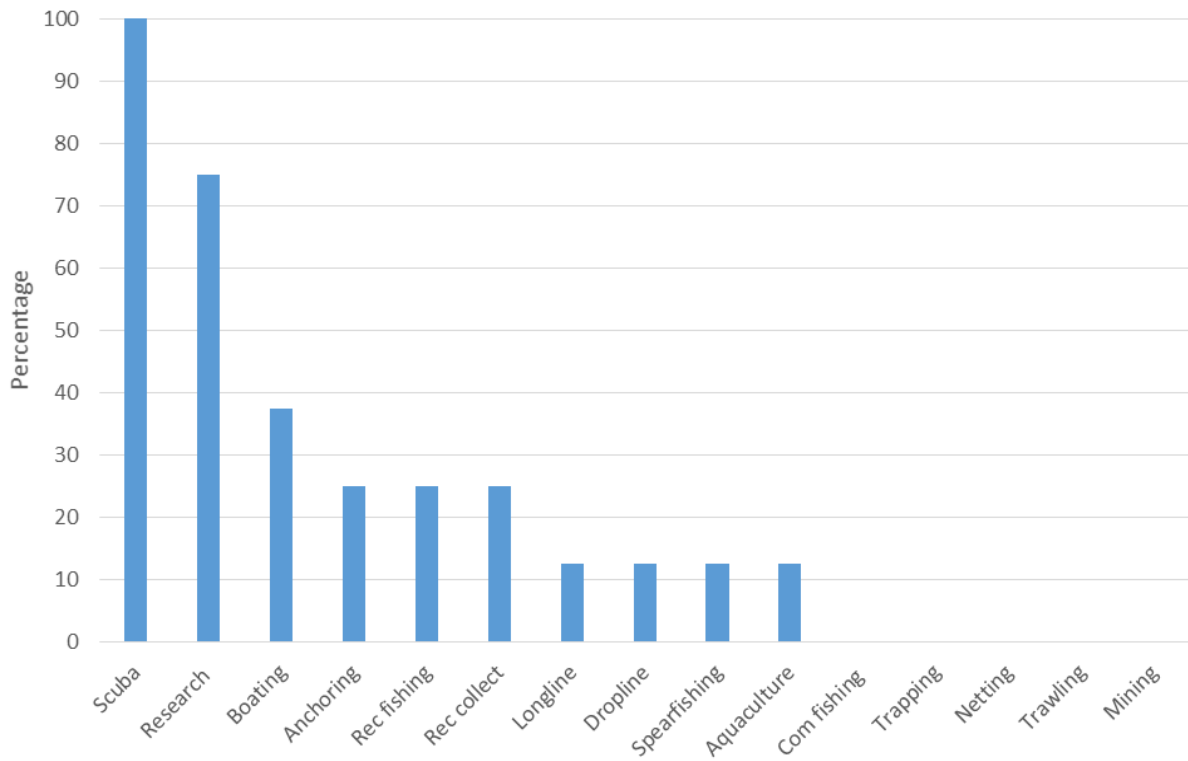


Figure D7. The proportion of respondents from TASMANIA that indicated permitted for each activity

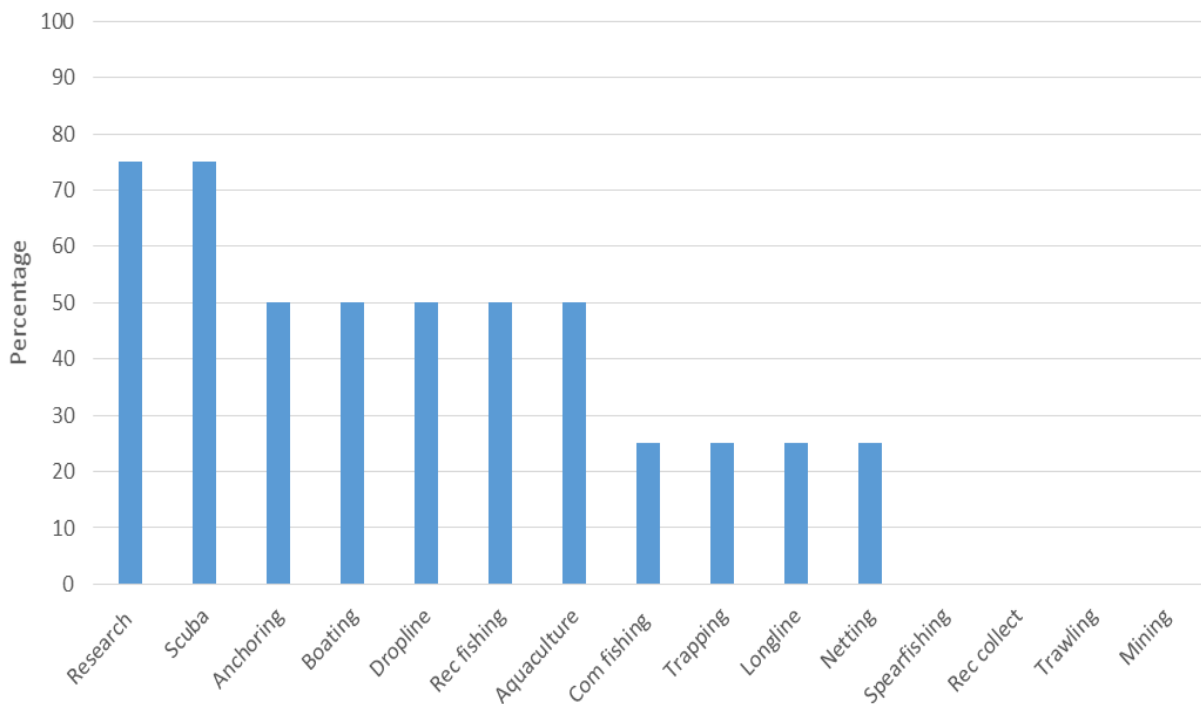


Figure D8. The proportion of respondents from the NORTHERN TERRITORY that indicated permitted for each activity

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