INTEGRATING SUSTAINABLE NATURAL RESOURCE MANAGEMENT AND CONSERVATION OBJECTIVES: APPROACHES TO ECOSYSTEM BASED MANAGEMENT

By

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ABSTRACT OF THE DISSERTATION

Integrating sustainable natural resource management and conservation objectives: approaches to ecosystem based management

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Ecosystem based management (EBM) is a wholistic approach to natural resource management whereby interactions with an ecosystem, including humans are considered in management rather than considering single issues, species or ecosystem services in isolation. Historically, natural resources were managed under a single management paradigm. However, when ecosystems are managed by their individual components important complexities and nuances in socio-ecological dynamics are lost. Given the recent shift to an ecosystem based approach, much research is needed on the integration of social and ecological components of ecosystems. My research contributes to this literature by (1) investigating the definition and implementation of ecosystem based fisheries management (EBFM) in the northeast and mid-Atlantic united states, (2) exploring biocultural homogenization of coral reef fish assemblages off the coast of Florida and (3) conducting a systematic review of interactions between fisheries and marine mammals at a global scale.

Acknowledgements

Firstly, I would like to thank my Ph.D. advisors Dr. Julie Lockwood and John Wiedenmann without whom this dissertation would not be possible. I will forever be grateful for their mentorship and guidance. Secondly, I would like to thank my committee members; Olaf P. Jensen and Ayana Elizabeth Johnson. I would also like to thank all of my co-authors. I would also like to thank my grandparents, William Jordan and Ruby Jordan without their love and support none of this would have been possible. I would like to thank my mother, Lori Dawn Jordan, who knew I had the capacities to achieve this before I knew it myself. I would like to thank my analyst, Isak De Vries, without whom this journey would have been impossible

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knowledge of biological baselines, and claiming cultural identity in the fish themselves or dive site. Given this connection, I posit that experienced divers will respond with increased conservation vigilance when realizing that fish assemblages they frequently encounter are homogenizing. The increase in conservation action will then tend to decrease homogenization levels (panel A). In contrast, novice divers may not notice that assemblages have homogenized, and thus will do nothing to prevent further homogenization, or even act in ways that further increase homogenization (panel B).

Atlantic/Caribbean; 5, northeast Atlantic; 6, southwest Atlantic; 7, eastern Atlantic; 8, Mediterranean; 9, Western Indian Ocean; 10, eastern Indian Ocean; 11, northwest Pacific; 12, southwest Pacific; and 13, southern Ocean/Antarctica. Grey represents areas where < 5 studies occurred, blue represents areas where more than 5-9 studies occurred, yellow represents areas where 10-19 studies occurred and red represents areas where ≥ 20 studies occurred. The stars represent areas of high marine mammal by-catch from Lewison et al (2013).

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Introduction

While conservation of natural resources and biodiversity may both aim for sustainability, the short-term goals of each can be at odds (de Groot et al. 2010). For example, the reduction of catch quotas for an economically important fish species to preserve a fraction for an endangered species that also feeds on the same fish species can be seen by the fishery as a temporary economic loss. For effective natural resource management and conservation, there must occur an evaluation and quantification of the tradeoffs between stakeholders (Daily et al. 2009). This dissertation explores how to integrate marine conservation ecology and natural resource management by (1) assessing the implementation of ecosystem-based fisheries management, (2) analyzing how natural resource user behavior may affect community composition, (3) and conducting a global systematic review of marine mammal and fishery interactions.

Natural resources form the foundation of all economies; they are the raw materials for production. Globally, the demand for and the extraction of natural resources continue to grow, as does the population. However, natural resources are valued by cultures and nations for more than their economic value. Natural resources are often closely linked to culture and possess an intrinsic value. Therefore, managing these resources is of critical importance both economically and socially. Although continued sustainability of natural resources would ultimately be mutually beneficial for all stakeholders, more immediately and over the short-term there are often major economic, social, and ecological tradeoffs between conservation and natural resource management. Before new management can be implemented and existing policies changed, these tradeoffs should be examined from

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multiple angles to ensure acknowledgement of duplicities of investments from stakeholders.

Chapter 1: Ecosystem Based Fisheries Management

This chapter was formatted for publication in Fisheries and will be submitted there.

(Safiq AD, Free CM, Caracappa JC, Valenti JL, and Jensen OP)

Within the context of marine natural resources, fish stocks and endangered species have historically been managed as single species. The single-species framework considers the biological requirements of a target species and may include some abiotic interactions on a local scale. However, single-species management ignores the complex reality of socio-economic influences on markets and ultimately on ecosystems. More importantly, single-species management ignores the fact that species are parts of complex ecosystems, and interdependent on other species.

Ecosystem-based fisheries management (EBFM) is a holistic approach to management that takes into account the environmental and biological interactions of multi-species in an ecosystem as well as the human dimensions within the ecosystem. Broadly, the primary components of EBFM are involvement of stakeholders in the management process, consideration of all-important ecological interactions through the best available science, and the consideration of abiotic interactions on the system.

Through the Magnuson-Stevens Conservation Fisheries Act, Marine Mammal Protection Act, and Endangered Species Act, require an ecosystem-based approach to managing fish stocks. However, there is no formal protocol for states to implement to transition from single-species management approach to EBFM. States must adhere to federal mandates, however, due to the biological and geographical heterogeneity between systems among states, the demands on natural resources and their management differs significantly. EBFM is complex, because ecosystems are complex, so there are often extensive data requirements, and some experts debate the feasibility of its implementation (Link 2015).

The first chapter investigates implementation of EBFM in the Mid-Atlantic and North East U.S. to (1) understand the working definition of EBFM at the state level and (2) identify barriers to implementing EBFM. Data was gathered via phone interviews with state fishery personnel regarding the implementation of and importance of specific components of EBFM. Our eight-part component of EBFM is defined in Table 1. Overall, our study found that there exists a common core working definition of EBFM at the state level, and that stakeholder involvement created barriers to the overall implementation of EBFM. This paper is currently in preparation for the journal Fisheries.

Chapter 2: Bio-cultural homogenization

This chapter was formatted for publication in Springer and was accepted there. (Safiq AD, Brown JA, and Lockwood J.L.)

Biological homogenization is the increase of genetic, taxonomic, or functional similarity of communities (McKinney 2006). This process is occurring across terrestrial, aquatic, and marine systems as a result of globalization, global climate change, and extinction events (McKinney and Lockwood 1999). Negative effects of biological homogenization include but are not limited to reduction or alteration of ecosystem function, and reduced genetic diversity (Hobbs et al. 2006). Although the ecological effects of biological homogenization have been well studied, the societal and social affects have not been studied to the same extent.

Bio-cultural homogenization adds a human dimension to biological homogenization by considering the potential to influences of natural resource users and their changing socio-economics associated with particular ecosystems (Safiq et al. in press). My second chapter investigates the occurrence of bio-cultural homogenization on coral reefs off the coast of Florida. These coral reefs provide one of the largest ecotourism attractions in the U.S. (Failler et al. 2015). Peoples' choices can have direct and indirect effects on ecosystem health and sustainability. In order to design and implement effective conservation, science and policy must begin to integrate the dynamics between the choices of resource users and the biological interactions between and among systems. The aim of this chapter is two-fold: (1) to investigate whether biological homogenization is happening on tropical reefs and (2) to understand how social values could affect the degree of biological homogenization through the actions of resource users. I found evidence of homogenization of some coral reef communities, but not all. I also postulate theoretical interactions between societal values, diver- practices, diver experience, and severity of homogenization. Overall, I demonstrate that tropical reefs off the coast of Florida are socio-ecological systems that are vulnerable to biological homogenization. This work has been submitted as a book chapter and it is currently under review by the editors of Springer.

Chapter 3: Systematic Review

This chapter is formatted for submission to Trends in Ecology and will be submitted there by the co-authors (A.D. Safiq, J. Wiedenmann, and J. Lockwood).

In my third chapter, I write a systematic review of global fishery and marine mammal interactions. In recent years, methods for systematic reviews in ecology have been extensively published (Koricheva et al. 2013). Commercial fisheries exist the world over, as do marine mammals. The occurrence of competition between fishermen and marine mammals is well documented internationally (Hardwood and Croxall 1988). By conducting a systematic review of these interactions, I have provided a global overview of the frequency and types of these interactions, and which species of fish and marine mammals are involved. To my knowledge, there is not yet a global review of marine mammal and fishery interactions. This information will be useful to natural resource managers, conservation scientists, and marine ecologists.

References

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B. E. Chasco, I.C. Kaplan, A.C. Thomas, A. Acevedo-Gutierez, D.P. Noren, M.J. Ford, M.B. Hanson, J.J. Scordino, S.J. Jeffries, K.N. Marshall, A.O. Shelton, C. Matkin, B.J. Burke, and E.J. Ward. 2017. Competing tradeoffs between increasing marine mammal predation and fisheries harvest of Chinook salmon. Scientific Reports 7, 14, 1-12.

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P. D. Christensen. 1997. Competition Between Fisheries and Marine Mammals for Prey and Primary Production in the Pacific Ocean. Journal of Northwest Atlantic Fishery Science. 22, 1- 12.

V. Christensen, C. J. Walters. 2004. Ecopath with Ecosim: methods, capabilities and limitations. Ecological Modelling. 172, 109-139.

G. C. Daily, S. Polasky, J. Goldstein, P.M. Kareiva, H.A. Mooney, L. Pejchar, T.H. Ricketts, J. Salzman, and R. Shallenberger. 2009. Ecosystem services in decision making: time to deliver. Frontiers in Ecology and the Environment. 7, 21-28.

R. S. de Groot, R. Alkemade, L. Braat, L. Hein, L. Willemen. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological Complexity. 7, 260-272.

P. Failler, E. Petre, T. Binet, J. P. Marechal. 2015. Valuation of marine and coastal ecosystem services as a tool for conservation: The case of Martinique in the Caribbean. Ecosystem Service 11, 67-75.

E. A. Fulton, J.S. Link, I.C. Kaplan, M. Savina-Rolland, P. Johnson, C. Aimsworth, P. Horne, R. Gorton, R.J. Gamble, A.D.M Smith, and D.C. Smith. 2011. Lessons in modelling and management of marine ecosystems: the Atlantis experience. Fisheries. 12, 171-188.

J. Harwood, J. P. Croxall. 1988. The assessment of competition between seals and commercial fisheries in the north-sea and the Antarctic. Marine Mammal Science. 4, 13-33.

R. J. Hobbs, S. Arico, J. Aronson, J.S. Baron, P. Bridgewater, V.A. Cramer, P.R. Epstein, J.J. Ewel, C.A. Klink, A.E. Lugo, D. Norton, D. Oijima, D.M. Richardson, E.W. Sanderson, F. Valladares, M. Vila, R. Zamor, and M. Zobel. 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global Ecology and Biogeography 15, 1-7.

J. Matthiopoulos., S. Smout, A.J. Winship, D. Thompson, I.L. Boyd, and J. Hardwood. 2008. Getting beneath the surface of marine mammal - fisheries competition. Mammal Review. 38, 167-188.

M. L. McKinney. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127, 247-260.

M. L. McKinney, J. L. Lockwood. 1999. Biotic homogenization: a few winners replacing many losers in the next mass extinction. Trends in Ecology and Evolution. 14, 450-453.

D. P. Mitchell. 2009. Land Administration Trends and their Implications for Australian Natural Resource Management. Journal of Spatial Science. 54, 35-47.

W. S. Patrick, J. S. Link. 2015. Myths that continue to impede progress in ecosystem-based fisheries management. Fisheries 40, 155-160.

E. K. Pikitch, C. Santora, E.A. Babcock, A. Bakun, R. Bonfil, D.O. Convoer, P. Dayton, P. Doukakis, D. Fluharty, B. Heneman, E.D. Houde, J. Link, P.A. Livingston, M, Mangel, MK. McAllister, J. Pope, and K.J. Sainsbury. 2004. Ecosystem-based fishery management. Science 305, 346-347.

E. E. Plaganyi, D. S. Butterworth,W. F. Perrin, B. Wursig, J. G. M. Thewissen. 2009. Competition with Fisheries. Encyclopedia of Marine Mammals, 2nd Edition (Elsevier Academic Press Inc, San Diego.

C. Roche, C. Guinet, N. Gasco, G. Duhamel. 2007.Marine mammals and demersal longline fishery interactions in Crozet and Kerguelen Exclusive Economic Zones: An assessment of depredation levels. CCAMLR Science. 14, 67-82.

J. F. Herman E. Daly, Ecological Economics, Second Edition: Principles and Applications. (Island Press, 2004).

J. G. Julia Koricheva, Kerrie Mengersen, Handbook of Meta-analysis in Ecology and Evolution. (Princeton University Press, Princeton and Oxord).

Perceptions of ecosystem-based fisheries management among state natural resource agency scientists in the Northeastern United States

Abstract

Ecosystem-based fisheries management (EBFM) is an application of ecosystem-based management in which abiotic, biotic, and socio-economic interactions are considered when managing fisheries. The primary objectives of this study were: (1) to understand how state fishery scientists define EBFM; (2) to rank the importance of inclusion and implementation of our eight-part component definition of EBFM; and (3) to identify potential barriers in implementing EBFM at the state level. The uniformity across inclusion responses indicated that there was a shared definition of EBFM among state fishery scientists. The most frequently implemented component was *engaging stakeholders* and *accounting for uncertainty* in ecosystems was least frequently implemented. Overall, the most frequently cited barrier in the New England region was *stakeholder engagement* while in the Mid-Atlantic region *regulatory* barriers was most frequently mentioned. These findings can help identify where potential human and fiscal resources should be allocated for successful implementation of EBFM at the state level.

Keywords: ecosystem-based fisheries management, fishery management plan, human dimensions of fisheries, natural resource management

Introduction

Ecosystem-based fisheries management (EBFM) is a holistic approach to fisheries management that recognizes the physical, biological, economic, and social complexities of managing living marine resources and considers the broader interactions between and among species that interact with target management species (Link 2010). Historically, policymakers and managers have relied on a single-species approach to natural resource management, but recent acknowledgements of the importance of complex interactions among species (such as trophic cascades, indirect competition, etc.) has led to a shift from the single-species approach towards an ecosystem-wide approach in the science supporting management (Pikitch et al. 2004; Hilborn 2011).

In recent years, emphasis has been placed on implementing ecosystem-based approaches to management within the context of marine systems, notably following the implementation of the U.S. Magnuson-Stevens Fishery Conservation and Management Act (MSA; NOAA 2006; Essington et al. 2016). The MSA mandates the preservation of sustainable stocks by using the best available science, fair resource access among stakeholders, and an expectation to capitalize on the economic potential of stocks without degrading their ability to provide benefits in the future (i.e., optimum yield). In essence, the MSA calls upon science to inform sustainable management, efficient economic strategies, and stakeholder engagement, which are all components of EBFM (Service 2018). Although EBFM is not explicitly mentioned in the MSA, U.S. federal government agencies still point to the MSA as a legislative mandate for implementing EBFM (DOC 2016). In addition to the governmental call for the implementation of EBFM, it has been noted by the scientific community to be one of the preferred frameworks for analyzing trade-offs between and among social, economic, and ecological systems (Walters and Martell 2004).

Biedron and Knuth (2016) found that at the federal level most stakeholders agreed on definitions, practices, and possible outcomes for EBFM. The study also found that the Mid-Atlantic Fishery Management Council and New England Fishery Management Council believed that state level fisheries management should gradually transition from single-species management to ecosystem-based management. Because their study focused exclusively on federally managed fisheries, it remains unknown if similar perceptual trends exist within state fishery management systems.

Despite the acknowledgement among scientists and policymakers at the federal level of the need to move toward EBFM, it remains unclear how state natural resource agencies should go about transitioning from single-species management to EBFM (Arkema et al. 2006). In a study by Patrick and Link 2015, the authors identify six myths that are noted to inhibit the implementation of EBFM. These myths are: linguistic uncertainty surrounding the definition of EBFM; lack of governance structure and mandates to implement EBFM; limitations surrounding quality and quantity of data; EBFM results in too restrictive management suggestions; and the lack of resources to fully implement EBFM in an already complicated socioeconomic predicament. It remains uncertain which if any of these prohibit the implementation of EBFM at the state level. State agencies are not unified under a single legislative mandate and are more heterogeneous in their approach to management. They often have broader management missions and a wider set of responsibilities compared to federal agencies. To advance and improve the implementation of EBFM, we need to have a baseline understanding of how state fishery scientists perceive and implement EBFM across regions. State fishery scientists are defined as those individuals who work directly with the biology and interactions of stock species although all scientists contacted in this study may not identify as strictly biologists. The goals of this study were to: (1) assess the level of agreement between state fishery scientists on the definition of EBFM; (2) identify how components of EBFM are incorporated in management plans; and (3) identify any barriers to the implementation of EBFM at the state level. We focused on the eleven states represented in the Mid-Atlantic and New England Fishery Management Councils. State fisheries agency scientists were the targets of our survey efforts, as they are both knowledgeable about local fisheries management practices and were deemed likely to discuss perceptions on management and policy candidly.

Methods

Respondent demographics

We interviewed 40 state fishery scientists working for state natural resources agencies in the coastal states of the Mid-Atlantic and New England fisheries management regions (Figure 1) about their agencies' interpretation and implementation of EBFM. We identified state fishery scientists to interview through a combination of agency websites and referrals from colleagues and interviewed scientists (i.e., "snowball sampling"; Ellard-Gray et al. 2015). The response rate of our study is 35.4% (40 of 112 scientists contacted). Respondents were first contacted through e-mail. If respondents replied to the e-mail, they were then contacted via phone. If respondents did not respond to the initial e-mail, a follow-up e-mail was sent. Respondents worked in a variety of systems (e.g., marine vs. freshwater, finfish vs. shellfish) and had medians of 15 and 19 years of experience at their agencies and in fisheries management, respectively (Figure 1).

Interview design

We conducted standardized interviews with respondents over the phone using a survey (Appendix A) designed to collect five categories of information: (1) respondent demographics (e.g., primary habitat work area such as freshwater or marine, years of experience with their agency and fisheries, and familiarity with EBFM); (2) definitions of EBFM; (3) implementation and effectiveness of EBFM; and (4) barriers to the implementation of EBFM. After asking respondents for basic demographic information, we asked for their open-ended definition of EBFM, prior to prompting them with our definition. Next, we asked respondents whether each of the eight components of EBFM listed in Table 1 are included in their agencies' interpretation of EBFM and whether their agencies are implementing each of these components. These components represent a simplification of the thirteen components of EBFM identified by Biedron and Knuth (2016). Responses to component inclusion and implementation were measured on a Likert scale of 1-5 and "do not know" where 1 represents "strongly disagree", 3 represents "neutral", and 5 represents "strongly agree". Finally, we asked respondents to provide an open-ended commentary on the effectiveness of and barriers to each component. While our survey was not identical in design or scope to that of Biedron and Knuth (2016), it was used as a guide to investigate perceptions at the state level.

Data analysis

Kruskal-Wallace (K-W) tests were performed, with pairwise comparisons, to determine whether components differed in inclusion or implementation. Non-parametric tests were necessary due to the non-normal distribution of scores. K-W tests were also used to compare scores between regions (Mid- Atlantic vs. New England) and systems (freshwater vs. marine). Finally, we identified twelve unique barrier categories in a post-hoc analysis of respondents' open-ended discussion of barriers to EBFM implementation and examined the frequency with which these barriers were mentioned by respondents for each component.

Results

Implementation and inclusion of EBFM components

Variation existed in the scoring of the eight components with respect to their implementation and inclusion. Some components were scored highly for inclusion but scored lower for implementation. An example of such is the component *accounting for uncertainty*. Other components were ranked highly for implementation and low for inclusion. Such an example was *considering socio-economics*. Components that were ranked highly on both inclusion and implementation were *stakeholder engagement* and *habitat protection*. *Habitat protection* was received the highest inclusion scores. The degree to which the eight EBFM components were implemented was lower compared to inclusion for all eight components (Paired T-Tests, all $p<0.05$; Figure 2). Stakeholder engagement ranked the highest for implementation while *accounting for uncertainty*, though a critical component of EBFM, showed the lowest implementation score
The components *geographically specific* (K-W: $\chi^2 = 4.8$, df = 1, p =0.028) and *complex interactions* (K-W: χ^2=4.3, df = 1, p = 0.037) differed significantly by region for implementation with New England ranking higher for implementation of these two components (Figure 3). When inclusion and implementation scores were compared

within systems (e.g., freshwater and marine), we only saw differences between regions for *complex interactions* (K-W: $\chi^2 = 8.11$, df = 1, p = 0.017). However, when asked how well participants thought EBFM was being implemented overall, those working in marine systems ranked their agencies more highly than those working in freshwater systems (K-W: $\chi^2 = 7.96$, df = 1, p = 0.019).

[B]Barriers to implementing EBFM

We identified twelve commonly cited barriers to EBFM in respondents open-ended discussion of the effectiveness and challenges of implementing EBFM (Supp. Table 1). *Stakeholders* and *regulatory* barriers were cited most frequently (36% and 31% mean response rate across components, respectively) across a variety of EBFM components (Figure 4). One respondent said "fisheries are political. Fisheries [management] is not about managing fish, it is about managing people". Following *stakeholders* and *regulatory* in frequency were *data* and *funding* (20% and 19% mean response rate, respectively). Comparatively, *politics* (8%), *staff* (7%), and *resources* (3%) represented less significant barriers to the implementation of the EBFM components (Figure 4). Data was a frequently cited barrier to *accounting for uncertainty* and funding is a frequently cited barrier to *habitat protection* (Figure 4). One respondent mentioned that "data quality, amount of data, funding to get data, and expertise" were barriers and another echoed those concerns when he/she said, "[it is] hard to plan for any unexpected events that might happen and in that regard, it is almost impossible to get funding for something like that".

Discussion

Defining EBFM

Previous studies have suggested that one of several challenges in implementing EBFM has been a lack of consensus on its definition (Patrick and Link 2015). Since the means of all component scores lie above 4, corresponding to ‘agree’, there appears to be agreement on the inclusion of our eight components of EBFM, suggesting that a shared core definition of EBFM exists among state fishery scientists in the Northeast. However, a recent study published by Trochta *et al*. found that, globally, there does not appear to be consensus on the definition of EBFM between different fisheries (Trochta et al. 2018). Our results do not contradict the findings of Trochta *et al*., instead our results offer greater insight into how perceptions of EBFM may change given different geographical scales. Since disagreement in the definition of EBFM disappear on the smaller regional scale of our study, our findings suggest that geographical scale may influence the degree of agreement on the definition of EBFM. Biedron and Knuth (2016) also found that there exists a common definition of EBFM among Mid-Atlantic Fishery Management Council and New England Fishery Management Council members and stakeholders which also focused on the inclusion of trophic interactions between species, stakeholder engagement, and the inclusion of habitat as a focal point for maintaining sustainable ecosystems (Biedron and Knuth 2016).

Implementing EBFM

All components of EBFM were reported to be under implemented given their importance to the definition of EBFM. This may be due to respondents being self-critical in their assessment of whether their agency was doing enough to implement EBFM. It could also be that there is genuinely more that could be done by state agencies to implement these components of EBFM.

While state fishery scientists agree that *accounting for uncertainty* is an important component of EBFM, the respondents note that implementing this component is particularly challenging. Several respondents mentioned that acquiring funding for the measures needed in order to implement *accounting for uncertainty* was a barrier. Although *stakeholder engagement* was reported to be the most highly implemented component of EBFM, *stakeholders* were also one of the most frequently reported barriers to implementation of EBFM components. Several respondents mentioned that failure to reach compromise between competing objectives inhibited or prolonged implementation of management objectives. This suggests that future work should aim to investigate the relationship between stakeholders and regulatory agencies to identify specific operational, political, and economic dynamics that could be improved upon. One potential solution to this problem would be to have a moderator at sessions who is specifically trained at moving discussion along an 'argument' based framework as opposed to a consensus based framework. Stakeholders who believe in public participation process to create consensus enter the process with inflated expectations, only to be disenchanted by contrary behavior of those with opposing views (Peterson et al. 2002). The resolution to this problem lies in the ability to hold opposing ideas in sustained debate based in an ecological framework without the expectation of consensus by majority. Consensus by majority often only reinforces the current socio-political imbalance of power which only further inhibits the implementation of multiple stakeholders (Peterson *et al.* 2004).

It is unsurprising that *stakeholder engagement* and *habitat protection* ranked highly in both categories since these components are often incorporated into the mission statements of state agencies. For example, North Carolina's Division of Marine Fisheries vision statement says, "North Carolina Division of Marine Fisheries ensures healthy, sustainable marine and estuarine fisheries and habitats through management decisions based on sound data and objective analysis" (NC Environmental Quality 2018). New York state DEC Division of Marine Resources is another example of an agency explicitly mentioning stakeholder engagement and habitat protection as management objectives.

Their mission statement is as follows: "to manage and maintain the state's living marine, estuarine and anadromous resources, and to protect and enhance the habitat upon which these resources depend… to achieve optimum benefit by providing for the broadest range of uses including commercial and recreational harvest, human consumption, natural forage and observation and appreciation (Conservation 2018). Although *accounting for uncertainty* was ranked highly for inclusion, it is also one of the most difficult to implement (Figure 3). For example, Wiedenmann and Jensen (2018) found that although stock assessment models account for uncertainty, they still resulted in biased projections of stock status, resulting in inaccurate catch limits and ultimately, overfishing of target species. Making progress on the implementation of EBFM will likely require developing new methods for accounting for uncertainty and making them more accessible (e.g., training scientists to implement these methods, etc.).

Survey participants in freshwater systems indicated that they perceived that EBFM was being implemented more than those working in marine systems. While this difference was not present when asked about individual components of EBFM, the perception that EBFM was more implemented in freshwater systems may reflect the comparatively longer history of stock enhancement and habitat management in freshwater systems (Welcomme et al. 2010). For example, fish stocking, the transfer or introduction of fish into a new environment, began in the U.S. in the mid-1800s and was central to the strategy of the U.S. Fish Commission, established in 1871 (Pister 2001). Though controversial (Cowx 1994), stocking has remained an important component of U.S. freshwater fisheries management, where it is implemented largely by state agencies (Anders Halverson 2008) and is used to subsidize recreational fisheries and rehabilitate endangered or threatened fish populations. By contrast, marine stock enhancement has been rare and more controversial (Grimes 1998; Hilborn 1998): red drum is the only fully marine species with a stock enhancement program in the U.S. (implemented by state agencies in Texas, Florida, and South Carolina). Similarly, habitat restoration is a central component of freshwater fisheries management (Cooke *et al*. 2014) and has been more easily implemented in freshwater systems than in marine systems due to the higher accessibility, reduced spatial size, and lower dispersal and connectivity of freshwater systems (Arlinghaus et al. 2016; Geist and Hawkins 2016).

Limitations of the study

A standard for what is considered an acceptable survey response rate has yet to be clearly defined (Rogelberg and Stanton 2007). The reported response rate (35.4%) for this study was comparable to other published studies in the field of fisheries science; for example, a study by Biedron and Knuth 2016 had a reported response rate between 57%

and 14% among components and a study by Brinson and Wallmo 2017 had a reported response rate between 24% and 38% among states. However, response representativeness is arguably more important for evaluating the quality of collected survey data as it directly addresses the potential for nonresponse bias (Cook et al. 2000; Rogelberg and Stanton 2007). The state fishery scientists interviewed in this study were fairly evenly distributed among survey regions (i.e. Mid Atlantic and New England) and systems of interest (i.e. freshwater and marine), and represented the gamut from early career to seasoned professionals. Additionally, at least two state fishery agency scientists from each of the 11 states comprising the Mid-Atlantic and New England regions was interviewed, demonstrating an apparent lack of non-response bias in the data set. Given the relatively small number of respondents from each state, comparisons of EBFM perceptions between individual states were not performed in order to keep the identities of the survey respondents anonymous. This precluded analyses of the influence of state specific frameworks and management requirements on perceptions of EBFM among state fishery scientists.

This study specifically targeted the perceptions of EBFM implementation and barriers among state fishery agency scientists in the northeastern U.S. A quantitative assessment of how well the scientists' perceptions corresponded with the true status of EBFM in their state agencies was not performed, yet would be useful given the potential for disconnect between perception and reality (Dunning et al. 2003). State fishery scientists interact with managers, council members, and stakeholders, and therefore offer a unique perspective on the implementation of and barriers to EBFM within their respective state agencies. However, state fishery scientists only represent one constituent within the agency and likely hold different perspectives about EBFM than state social scientists, economists, and managers (Clay and McGoodwin 1995; Trochta et al. 2018). Perceptions of EBFM also differ across larger geographic scales (Pitcher et al. 2009) and thus may differ regionally within the U.S. Future studies of EBFM perceptions in the U.S. should consider additional regional perspectives (e.g. Pacific, South Atlantic) and the perceptions of state social scientists, economists, and managers, council members, and stakeholders in order to attain a comprehensive assessment of the perceptions of EBFM at the state level (Jacobson and McDuff 1998; Kaplan and McCay 2004).

Overall, our study provides evidence that the perceived definition and implementation of EBFM of state fisheries scientists is similar across New England and Mid-Atlantic states. Participants also indicated that components of EBFM were not being implemented to the same degree as their perceived importance. Our study also highlights the specific barriers that state fisheries scientists face in implemented components of EBFM. In all, we provide a useful framework for future studies to assess perceptions of EBFM and barrier towards implementation in other geographic regions.

Acknowledgements

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References

Commented [JW4]: These references are not formatted correctly.

Arkema, K. K., Abramson, S.C., and B.M., Dewwsbury. 2006. Marine ecosystem-based management: from characterization to implementation. *Frontiers in Ecology and the Environment* **4**: 525-532.

Arlinghaus, R., Lorenzen, K., Johnson, B. M., Cooke, S. J., and I.G. Cowx. 2016. Management of freshwater fisheries: addressing habitat, people and fishes. *Freshwater Fisheries Ecology* **1**:557-579.

Biedron, I. S. and B. A. Knuth. 2016. Toward shared understandings of ecosystem-based fisheries management among fishery management councils and stakeholders in the US Mid-Atlantic and New England regions. *Marine Policy* **70**: 40-48.

Brinson, A. A., and K. Wallmo. 2017. Determinants of saltwater anglers' satisfaction with fisheries management: Regional perspectives in the United States. *North American Journal of Fisheries Management* **37**: 225-234.

Clay, P. M., and J. R. McGoodwin. 1995. Utilizing social sciences in fisheries management. *Aquatic Living Resources* **8**: 203-207.

Cooke, S.J., Arlinghaus, R., Bartley, D.M., Beard, T.D., Cowx, I.G., Essington, T.E., Jensen, O.P. Lynch, A., Taylor, W.W., and R. Watson. 2014. Where the waters meet: sharing ideas and experiences between inland and marine realms to promote sustainable fisheries management. *Canadian Journal of Fisheries and Aquatic Sciences* **71**: 1 – 9.

Cook, C., F. Heath, and R. L. Thompson. 2000. A meta-analysis of response rates in web- or internet-based surveys. *Educational and Psychological Measurement* **60**: 821-836.

Conservation of N.D. Division of Marine Resources. Available: https://www.dec.ny.gov/about/796.html#Mission. (March 2018)

Cowx, I. G. 1994. Stocking strategies. *Fisheries Management and Ecology* **1**: 15-30.

Dill, L. M., M. R. Heithaus, and C. J. Walters. 2003. Behaviorally mediated indirect interactions in marine communities and their conservation implications. *Ecology* **84**:1151 – 1157.

Dunning, D., K. Johnson, J. Ehrlinger, and J. Kruger. 2003. Why people fail to recognize their own incompetence. *Current Directions in Psychological Science* **12**: 83-87.

Ellard-Gray, A., N. K. Jeffrey, M. Choubak, and S. E. Crann. 2015. Finding the hidden participant: solutions for recruiting hidden, hard-to-reach, and vulnerable populations. *International Journal of Qualitative Methods* **1**:1-10.

Essington, T. E., P. S. Levin, L. G. Anderson, A. Bundy, C. Carothers, F. C. Coleman, J. H. Grabowski, L. R. Gerber, E. D. Houde, O. P. Jensen, C. Möllmann, K. Rose, J. N. Sanchirico, and A. D. M. Smith. 2016. Building Effective Fishery Ecosystem Plans: A Report from the Lenfest Fishery Ecosystem Task Force. *Lenfest Ocean Program*.

Geist, J., and S.J., Hawkins. 2016. Habitat recovery and restoration in aquatic ecosystems: current progress and future challenges. *Aquatic Conservation: Marine and Freshwater Ecosystems* **26**: 942-962.

Grimes, C.B. 1998. Marine stock enhancement: sound management or techno-arrogance? *Fisheries* **23**: 18-23.

Halverson, M. A. 2008. Stocking trends: a quantitative review of governmental fish stocking in the United States, 1931 to 2004. *Fisheries* **33**: 69-75.

Hilborn, R. 1998**.** The economic performance of marine stock enhancement projects. *Bulletin of Marine Science* **62**: 661-674.

Hilborn, R. 2011. Future directions in ecosystem based fisheries management: A personal perspective. *Fisheries Research* **108**:235 -239.

Jacobson, S. K., and M. D. McDuff. 1998. Training idiot savants: The lack of human dimensions in conservation biology. *Conservation Biology* **12**: 263-267.

Kaplan, I. M., and B. J. McCay. 2004. Cooperative research, co-management and the social dimension of fisheries science and management. *Marine Policy* **28**: 257-258.

Leslie, H. M. 2009. Ecosystem-Based Management for the Oceans. Island Press.

Link, J. S. 2010. Ecosystem-Based Fisheries Management Confronting Tradeoffs, Cambridge University Press.

Marshak, A. R., and S. K. Brown. 2017. Habitat Science Is an Essential Element of Ecosystem-Based Fisheries Management. *Fisheries* **42**:300.

National Marine Fisheries Service. Fishery Management Council Committees. Available: http://www.mafmc.org/committees/. (March 2018)

NC Environmental Quality. Vision Statement. Available: http://portal.ncdenr.org/web/mf/vision-statement. (March 2018)

NOAA National Oceanic and Atmospheric Administration. 2006. Magnuson-Stevens.

Fishery Conservation and Management Act : MSA. U.S. Government. 2007. Public Law 94-265.

Patrick, W. S. and J. S. Link. 2015. Myths that continue to impede progress in ecosystem-based fisheries management. *Fisheries* **40**: 155-160.

Peterson, M. N., Peterson, M.J., and T.R. Peterson. Conservation and the myth of consensus. 2004. *Conservation Biology* **19**:762-67.

Peterson, M. N., T.R. Peterson, M.J. Peterson, R.R. Lopez, and N.J. Silvy. 2002. Cultural conflict and the endangered Florida Key deer. *Journal of Wildlife Management* **66**:947-68.

Pikitch, E. K., C. Santora, E. A. Babcock, A. Bakun, R. Bonfil, D. O. Conover, P. Dayton, P. Doukakis, D. Fluharty, B. Heneman, E. D. Houde, J. Link, P. A. Livingston, M. Mangel, M. K. McAllister, J. Pope, and K. J. Sainsbury. 2004. Ecosystem-based fishery management. *Science* **305**:346-347.

Pikitch, E. K., C. Santora, E.K., and E.A. Babcock. 2004. Ecosystem-based fishery management. *Science* **305**: 346-347.

Pitcher, T. J., D. Kalikoski, K. Short, D. Varkey, and G. Pramod. 2009. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. *Marine Policy* **33**: 223-232.

Pister, E. P. 2001. Wilderness fish stocking: history and perspective. *Ecosystems* **4**: 279-286.

Rogelberg, S. G., and J. M. Stanton. 2007. Introduction: Understanding and dealing with organizational survey nonresponse. *Organizational Research Methods* **10**: 195-209.

Service, N. M. F. 2018. About Ecosystem-Based Fisheries Management. Available: https://www.st.nmfs.noaa.gov/ecosystems/ebfm/about. (January 2018).

Trochta, J. T., Pons, M., Rudd, M.B., Krigbaum, M, Tanz, A., R. Hillborn. 2018. Ecosystem-based fisheries management: Perception on definitions, implementations, and aspirations. *PLOS One* **13**(1).

Walters CJ, Martell SJD. 2004. Fisheries Ecology and Management (Princeton Univ Press, Princeton).

Welcomme, R. L., Cowx, I. G., Coates, D., Béné, C., Funge-Smith, S., Halls, and A., Lorenzen. 2010. Inland capture fisheries. *Philosophical Transactions of the Royal Society B: Biological Sciences* **365**:2881-2896.

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Wiedenmann, J., and O. Jensen. 2018. Uncertainty in stock assessment estimates for New England groundfish and its impact on achieving target harvest rates. *Canadian Journal of Fisheries and Aquatic Science* **75**:342–356.

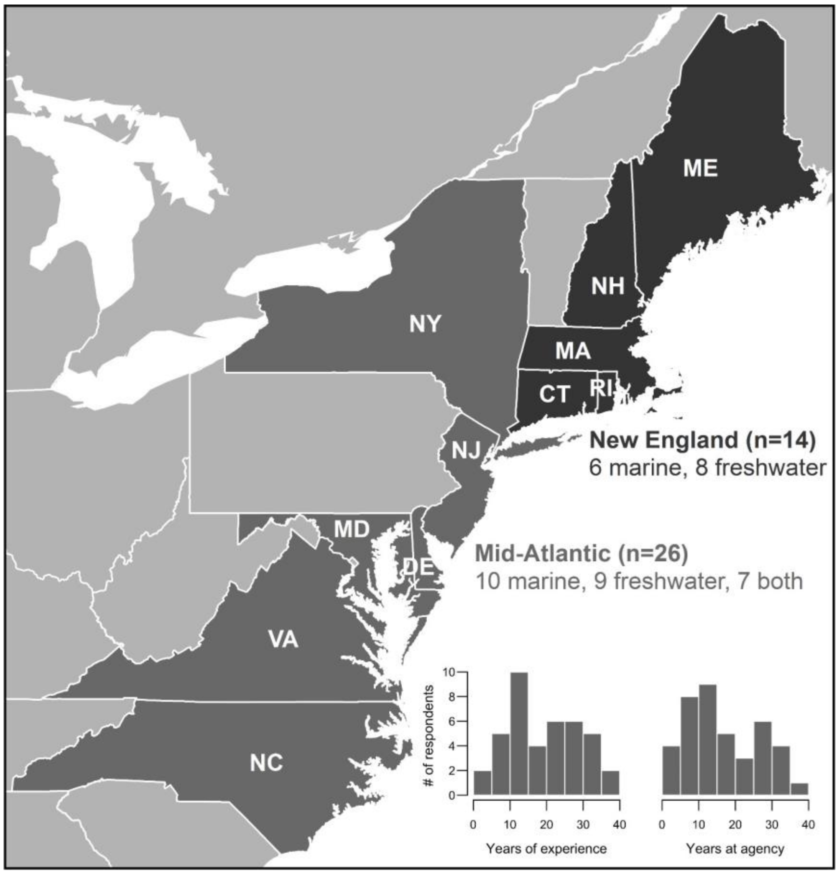

Figure 1. Map of the study region and respondent demographics.

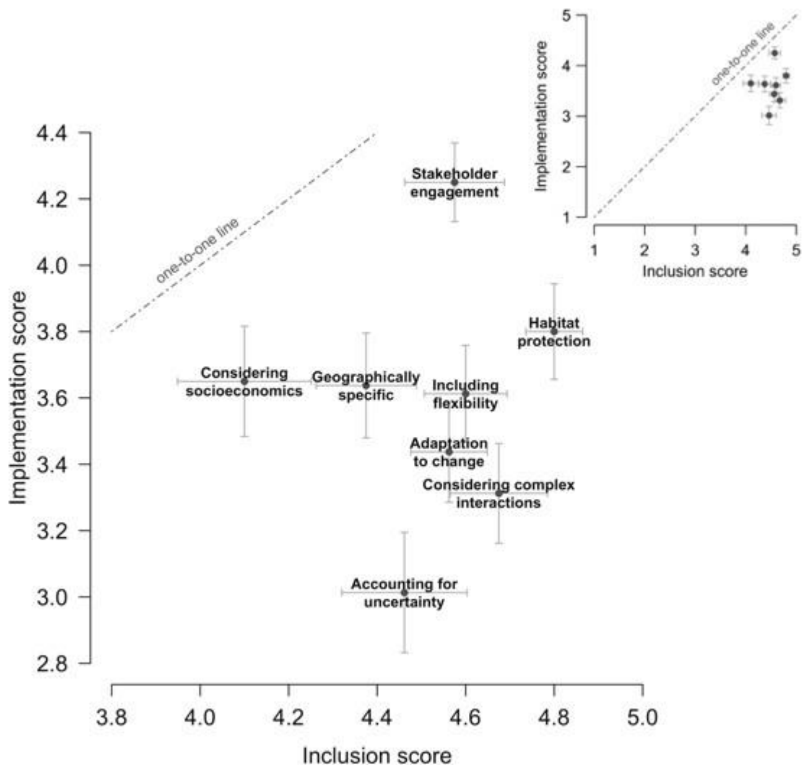

Figure 2. Mean inclusion and implementation scores for the eight EBFM components. The one-to-one line represents equal inclusion and implementation. Error bars represent ±1 standard error of the mean.

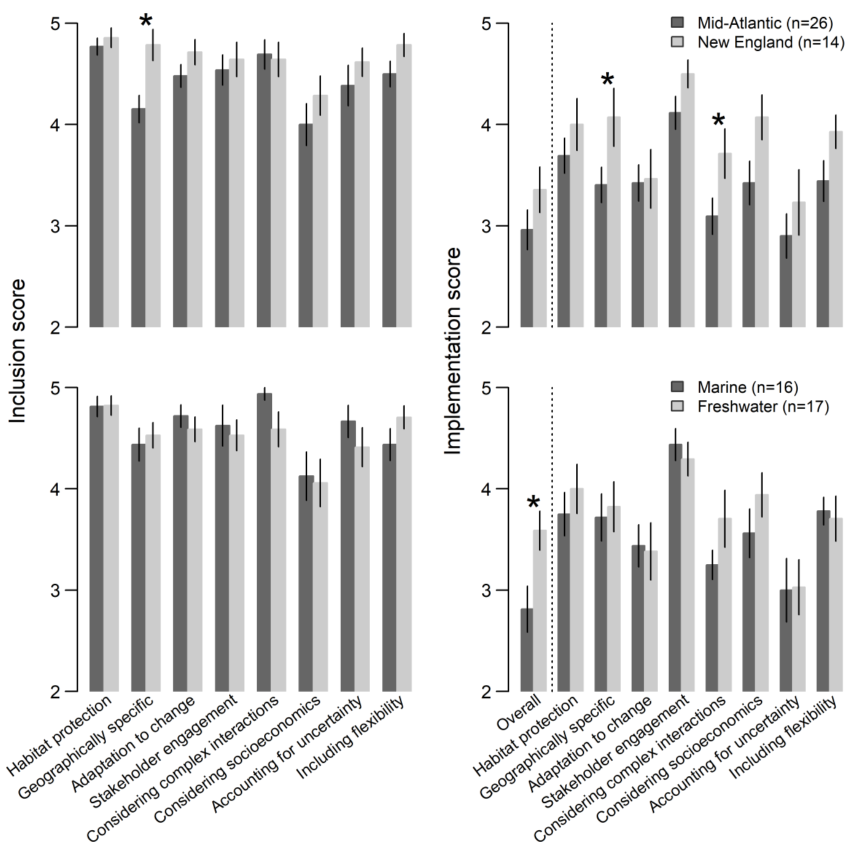

Figure 3. Mean inclusion and implementation scores for the eight EBFM components between Mid-Atlantic, New England, marine and freshwater respondents. Respondents working in both systems were not included. Error bars represent ±1 standard error of the mean. Asterisks represent statistically significant means.

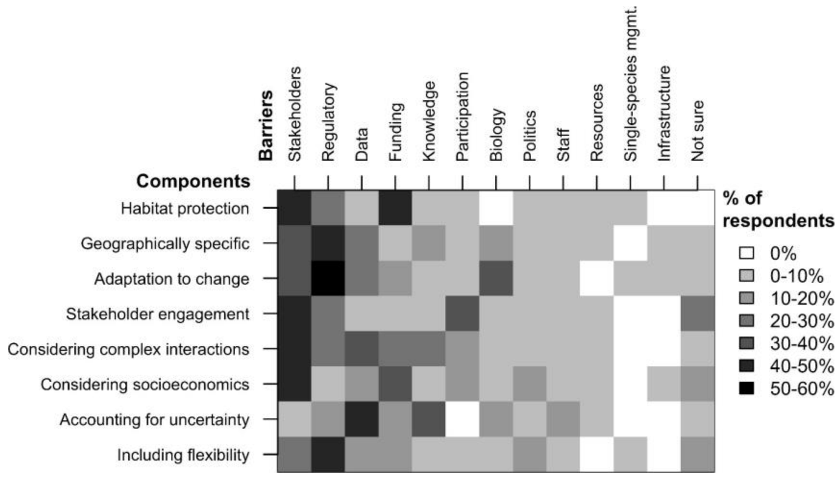

Figure 4. – Barriers to the eight EBFM components. Gray shading illustrates the percent of respondents who reported a specific barrier to a specific component.

Table 1. EBFM components (adapted from Biedron and Knuth 2016).

	Component	Definition
1	Habitat protection	Protecting/enhancing habitat
2	Geographically specific	Incorporating geographically specific management needs
3	Adaptation to change	Adapting to changing biological and social conditions
4	Stakeholder engagement	Engaging stakeholders
5	Considering complex interactions	Considering the interactions between the physical, biological, and human factors that affect the health of fisheries
6	Considering socioeconomics	Considering the social, economic, and cultural impacts on industries and communities that depend on fisheries
7	Accounting for uncertainty	Accounting for uncertainty in ecosystems
8	Including flexibility	Including flexibility in management strategies

Supp. Table 2. Keywords of EBFM component barriers.

	Barrier	Definition
1	Stakeholders	Stakeholder buy-in
2	Regulatory	Conflicts arising from vagueness, conflicting objectives, and/or scope of previously existing rules and laws
3	Data	Data limitations
4	Funding	Funding limitations
5	Knowledge	Staff experience/knowledge
6	Participation	Engagement of stakeholders and/or staff
7	Biology	Challenging species biology and/or community ecology (e.g., food webs are often poorly characterized)
8	Politics	Conflicts arising from lack of consensus, engagement, and/or mismatched objectives
9	Staff	Staff size and/or time
10	Resources	Availability of tools and/or technology
11	Single-species management	Inertia to overcome single-species management policy
12	Infrastructure	Difficulty navigating and/or operating within pre-existing programs and/or departments
13	EBFM	Challenges intrinsic to the idea of EBFM (e.g. accounting for complexities arising from interactions among/between components)

Appendix A

Ecosystem-based Fisheries Management note-taking template

Today's date:

Who is interviewing? Who is taking notes?

What state does this person work for?

Preface

Thank you so much for agreeing to take this survey. There are two of us here; me (name), and (name), who will be taking notes.

We sent you the consent form, and we hope you had time to review it. To summarize: all responses will be anonymous, and you can stop the interview at any time. Do you have any questions about the consent document? And I just need to ask for verbal consent: do you agree to take the survey? Check here if yes:

The survey has about 15 questions, and will take approximately 20-30 minutes to complete. There are some open-ended questions, and others that are answered on a scale of 1 to 4, or 1 to 5.

Do you have any questions before we begin?

Possible questions/answers:

- *Who else have you interviewed?* State fishery scientists from your state and other states, but the specific names are kept anonymous.
- *How will you keep my answers anonymous?* We are taking notes on your responses but the information is not associated with your name.
- *I don't know anything about EBFM.* This project is about your thoughts on EBFM, and all of your responses will be extremely helpful.
- Note: avoid using acronyms unless they do so first.

Questions

1. How long have you been working for your agency? (# of years)

2. How long have you been working in fisheries management in general? (# of years)

3. Which of the following systems do you work in?

System	Answer
Marine	
Freshwater	
Finfish	
Shellfish	
Aquaculture	
Diadromous	

4. In addition to your role as a fisheries biologist, which of the following roles do you identify as?

Option	Answer
Recreational fisherman	
Commercial fisherman	
Management Council member	

5. On a scale of 1-4, how familiar are you with the term "ecosystem-based fisheries management (EBFM)"? [*Read number + word options.*]

Option	Answer
1 - Not familiar at all	
2 - Slightly familiar	
3 - Moderately familiar	
4 - Very familiar	

Part 2 (Definition and implementation of EBFM):

6. How would you define Ecosystem-Based Fisheries Management? (Open-ended)
If they ask whether we're looking for the personal or the institutional definition of EBFM, ask them for both)

7. To what extent do you agree or disagree that your organization is implementing this definition of Ecosystem-Based Fisheries Management? You can answer on a scale of 1

to 5, with 1 being strongly disagree, and 5 being strongly agree. (We sent you the scale in your email – do you want to pull the email up so you can look at it?)

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

Part 3 (EBFM components)

For the next section, we're going to ask you about eight management practices. For each management practice, we have two questions:

8. To what extent do you agree or disagree that this practice is part of ecosystem-based fisheries management,
9. To what extent do you agree or disagree that this practice is being implemented by your agency?

You're going to answer each question on a 1 to 5 scale, with 1 being strongly disagree, and 5 being strong agree. We sent you the scale in your email.

a. Protecting and/or enhancing habitat.

Commented [JW6]: You have 8 and 9 above, so this can't be right.

Same for ALL of the ones below. This needs to be consistent throughout. Also, what is the a. above?

To what extent do you agree or disagree that this practice is part of ecosystem-based fisheries management?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

To what extent do you agree or disagree that this practice is being implemented by your agency?

Option	Answer
1 - Strongly disagree	

2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

If the answer to Q9 is 3, 4, or 5:

What barriers, if any, have you faced in implementing it?

In your opinion, how effective has it been as part of fishery management?

If the answer to Q9 is 1 or 2:

Does your organization plan to start implementing it in the future?

Option	Answer
Yes	
No	

If yes, when is that planned? What barriers, if any, do you anticipate to face in implementing it?

If no, why not?

b. Incorporating geographically specific management needs.

To what extent do you agree or disagree that this practice is part of ecosystem-based fisheries management?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

Commented [JW7]: Why do you start at 8b here?

Your numbering is very confusing

To what extent do you agree or disagree that this practice is being implemented by your agency?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

If the answer to Q9 is 3, 4, or 5:

What barriers, if any, have you faced in implementing it?

In your opinion, how effective has it been as part of fishery management?

If the answer to Q9 is 1 or 2:

Does your organization plan to start implementing it in the future?

Option	Answer
Yes	
No	

If yes, when is that planned? What barriers, if any, do you anticipate to face in implementing it?

If no, why not?

c. Adapting to changing biological and social conditions.

To what extent do you agree or disagree that this practice is part of ecosystem-based fisheries management?

Commented [JW8]: Same as above, why start at 8?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

To what extent do you agree or disagree that this practice is being implemented by your agency?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

If the answer to Q9 is 3, 4, or 5:

What barriers, if any, have you faced in implementing it?

In your opinion, how effective has it been as part of fishery management?

If the answer to Q9 is 1 or 2:

Does your organization plan to start implementing it in the future?

Option	Answer

If yes, when is that planned? What barriers, if any, do you anticipate to face in implementing it?

If no, why not?

d. Engaging stakeholders.

To what extent do you agree or disagree that this practice is part of ecosystem-based fisheries management?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

To what extent do you agree or disagree that this practice is being implemented by your agency?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

If the answer to Q9 is 3, 4, or 5:

What barriers, if any, have you faced in implementing it?

In your opinion, how effective has it been as part of fishery management?

If the answer to Q9 is 1 or 2:

Does your organization plan to start implementing it in the future?

Option	Answer
Yes	
No	

If yes, when is that planned? What barriers, if any, do you anticipate to face in implementing it?

If no, why not?

e. Considering the interactions between the physical, biological, and human factors that affect the health of fisheries.

To what extent do you agree or disagree that this practice is part of ecosystem-based fisheries management?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

To what extent do you agree or disagree that this practice is being implemented by your agency?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

If the answer to Q9 is 3, 4, or 5:

What barriers, if any, have you faced in implementing it?

In your opinion, how effective has it been as part of fishery management?

If the answer to Q9 is 1 or 2:

Does your organization plan to start implementing it in the future?

Option	Answer
Yes	
No	

If yes, when is that planned? What barriers, if any, do you anticipate to face in implementing it?

If no, why not?

f. Considering the social, economic, and cultural impacts on industries and communities that depend on fisheries.

To what extent do you agree or disagree that this practice is part of ecosystem-based fisheries management?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	

To what extent do you agree or disagree that this practice is being implemented by your agency?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

If the answer to Q9 is 3, 4, or 5:

What barriers, if any, have you faced in implementing it?

In your opinion, how effective has it been as part of fishery management?

If the answer to Q9 is 1 or 2:

Does your organization plan to start implementing it in the future?

Option	Answer
Yes	
No	

I have no idea, but I think we would have to in order to evolve as an agency

If yes, when is that planned? What barriers, if any, do you anticipate to face in implementing it?

Data and communication 14f. If no, why not?

g. Accounting for uncertainty in ecosystems.

To what extent do you agree or disagree that this practice is part of ecosystem-based fisheries management?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	

4 - Agree	
5 - Strongly agree	

To what extent do you agree or disagree that this practice is being implemented by your agency?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

If the answer to Q9 is 3, 4, or 5:

What barriers, if any, have you faced in implementing it?

In your opinion, how effective has it been as part of fishery management?

If the answer to Q9 is 1 or 2:

Does your organization plan to start implementing it in the future?

I sure hope so - we definitely acknowledge eat uncertainty, but not enough to do anything about it

Option	Answer
Yes	
No	

If yes, when is that planned? What barriers, if any, do you anticipate to face in implementing it?

If no, why not?

h. Including flexibility in management strategies.

To what extent do you agree or disagree that this practice is part of ecosystem-based fisheries management?

Option	Answer
1 - Strongly disagree	

2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

To what extent do you agree or disagree that this practice is being implemented by your agency?

Option	Answer
1 - Strongly disagree	
2 - Disagree	
3 - Neutral	
4 - Agree	
5 - Strongly agree	

If the answer to Q9 is 3, 4, or 5:
What barriers, if any, have you faced in implementing it?

In your opinion, how effective has it been as part of fishery management?

If the answer to Q9 is 1 or 2:

Does your organization plan to start implementing it in the future?

Option	Answer
Yes	
No	

If yes, when is that planned? What barriers, if any, do you anticipate to face in implementing it?

If no, why not?

Chapter 2: Homogenization of fish assemblages off the coast of Florida

Introduction

Marine fish assemblages have been substantially altered through the effects of over- fishing, habitat loss, and climate change (Levin et al. 2006; Airoldi et al. 2008; Nyitrai et al. 2012; Riegl et al.2009). Such impacts have been measured almost exclusively in the context of loss of key species or fish biomass. Rarely has anyone measured these impacts through the lens of compositional change, which is a perspective that acknowledges that species losses and additions combine to increase or decrease spatial similarity in fish assemblages through time (Olden et al. 2004). The few studies that have adopted this perspective have highlighted the homogenizing influence of habitat simplification and phase shifts, especially among coral reefs (Thrush et al. 2006, Airoldi et al. 2008, Alvarez-Filip et al. 2015), or of heavy influxes of exotic species after the establishment of inter-ocean canals (Edelist et al. 2013). The data necessary to track longer term changes in composition are hard to come by, but especially so within marine ecosystems which are by default harder to track due to limited logistical access. Here we take advantage of a citizen-science initiative to document near-shore fish assemblages to explore the extent to which we see homogenization within the Atlantic coastal waters of Florida (USA). Florida has nearly 12,000 miles of coastline, which includes the string of small islands that lie at the southern extreme of the state called the Keys (Figure 1). The near- shore marine waters of the Florida Atlantic coast and Keys exhibit a diverse array of seabed types including significant expanses of coral and artificial reefs. These reefs, and other habitat types, are home to hundreds of fish species some of which are important sources of commercial seafood or sold in the marine aquarium trade (Johns et al. 2014;

Commented [JW9]: INDENT first line in paragraph throughout – here and

Brucker 2005). These fish assemblages also support a massive SCUBA and snorkel dive industry, which generates on average US$3 billion annually in the Keys alone (World Wildlife Fund Global, Accessed 12 June 2017). These reefs and similar habitats thus represent significant sources of ecosystem services (Lane et al. 2015). Caribbean reefs, of which the Florida group are northern most members, have suffered several major disturbance events over the last decades including bleaching, the loss of top predators, and the emergence of diseases that have reduced reef structural complexity (Manzello et al. 2007; Green and Bruckner 2000). The fish that use these reefs have shown mixed responses to such changes, with some assemblages showing signs of recovery within a decade or less, with others showing strong lags whereby fish assemblages have yet to recover to their pre-disturbance composition and biomass (Alevizon and Porter 2014). There is recent evidence showing Caribbean-wide shifts in fish assemblages toward habitat generalists and away from specialists (Alvarez-Fillip et al. 2015), and overall loss in fish abundance (Paddack et al. 2009). The reefs along the Florida Keys were included in these assessments, however no one has explored whether the spatial similarity of these reefs have increased through time or if these reefs have become more similar to other non-reef habitats along Florida's Atlantic coast.

We posit that changes in fish assemblage spatial structure will influence the attractiveness of Florida marine habitats for dive tourism. One of the main motivations for diving any area is to experience the variety of fish species that use that site, with the more species variety the better (Bhat, 2002). Homogenization of fish assemblages across sites may provide a strong feedback into the dive tourism industry by reducing the attractiveness of Florida's diving locations. From a recreational diver's perspective, homogenization may lead one to conclude that once fish at one location have been seen, there is no reason to travel to other sites that have essentially the same diversity. To our knowledge, the degree to which homogenization (or differentiation) influences the attractiveness of reefs and other habitats for dive tourism have not been explored. Therefore, we see our results as providing a starting point for this conversation by documenting the degree of homogenization realized among Florida's fish assemblages, which parts of Florida contribute most to any observed homogenization, and which species are 'winning' or 'losing'.

Methods

Survey Sites

We queried the Reef Environmental Education Foundation (REEF) online database to obtain presence-absence data from across survey sites along the Atlantic coast of Florida (World Wide Web electronic publication; www.reef.org, date of download: 12 May 2017; Figure 1). REEF uses a hierarchical coding system whereby survey sites are assigned a unique identification number, and these are grouped into larger spatial units. Our data comes from the REEF Florida East Coast and Keys (Zone 3) of the Tropical Western Atlantic region. This Zone covers survey sites from Saint Mary's River at the northern Atlantic coast boundary between Florida and Georgia, down to the Dry Tortugas, which is the western-most island group in the Keys (Figure 1). Within this zone there are 13 sub-zones within which surveys are conducted at various sites. Survey sites include natural coral reefs, artificial reefs, and other seafloor structures that attract relatively large numbers of marine fish (e.g., natural rubble, marine ridges).

In all REEF surveys, fish species presence is recorded by trained volunteer SCUBA divers using the Roving Diver Technique (RDT).This survey protocol requires divers to record all fish species readily identifiable while freely swimming around a survey site (Schmitt et al. 1996). RDT has proven easy to use by trained volunteers (Schmitt et al. 2002), and the resulting data can be reliably used to derive differences in fish community composition between sites (Hassell et al. 2013). Because the REEF program began its volunteer survey program within the Florida Keys, some survey records in this zone date back to the early 1990s and continue to the present day. There are large differences in the total number of sites surveyed across sub- zones (from 14 to 502), with bigger sub-zones have more survey sites. Between 1994 and 2015 REEF volunteers conducted 37,695 surveys at 1903 survey sites in this zone, summing to 34,305 dive hours with the average dive lasting 45-minutes.

Our goal here is to document longer-term trends in fish assemblages, thus we choose a subset of REEF survey sites that had data from two years that were at least 10 years apart. Only 44% of the total survey sites were ever resampled (853), and only 41% of these resampling events occurred over a >10-year time frame. In addition, because the number of species detected is highly dependent on survey effort (in this case measured as dive time), we needed to standardize effort across years within survey sites to ensure relatively unbiased temporal assessments of compositional change. Thus, the years we chose to include in our analyses had to have dive times (effort) that were no more than 20% different in length. Of the survey sites that fit our inclusion criteria, the time between re-sampling events ranged from the minimum of 10 years up to 20 years (mean = 15 years, standard deviation = 3.4 years). Initial surveys were conducted between the years of 1997 to 2004 and the later resampling surveys occurred from 2011 to 2017.

With these criteria in mind we selected 353 survey sites for inclusion in our analyses, which represents 20% of all sites surveyed in this REEF zone. We included between 1 and 139 survey sites per sub-zone, representing between 4 – 30% of all sites surveyed within each sub-zone. The sub-zones that had very few survey sites in our analysis were from the small and remote keys of Marquesas and Dry Tortugas. We had at least one survey site that qualified in terms of our criteria from all sub-zones. The survey sites located in the northern section of this zone are dominated by artificial reefs (e.g., sunken boats or concrete pilings), natural rubble fields, concrete culverts, and some scattered natural reefs and marine ridges. As one moves south into the Keys, the survey sites shift to include mostly individual or aggregated coral reefs and colonized pavement (Florida Fish and Wildlife Conservation Commission-Fish and Wildlife Research Institute accessed June 2017; Walker and Gilliam 2013).

Species Composition

We created two lists of species present for each of the 353 survey sites we analyzed. The first was composed only of species recorded as present in recent surveys ('current' time frame), and the second was only the species present in surveys at least 10 years prior (‘historical’ time frame). We combined the species lists for each site within each REEF sub-zone to create a scaled-up accounting of species composition for each of the 13 sub- zones (Figure 1). So for example, all of the 55 survey sites we included within the Key Largo sub-zone we combined into one historical and one current species list.

We decided to use this larger spatial grain size to analyze our compositional data for two reasons. First, the number of fish species recorded across all of the 353 sites we included totaled 452. Tracking compositional changes in this number of species through time and across all survey sites is computationally challenging and difficult to visualize. We could, however, easily track and visualize temporal compositional changes in this many species across the 13 sub-zones. Second, the scarcity of survey sites that were ever resampled meant that we could include only one year of survey information from each site within each of the time periods (historical and current). The chances that one 45-minute dive missed recording rare or elusive species is relatively high (Rota et al. 2009). By combining survey sites within sub-zones, we can alleviate this issue to some extent. The more survey sites included, the more likely at least one diver recorded the rarer or more elusive species using a sub-zone in a given yearn thus allowing us to include them in our analyses. Note, however, that we could not fully compensate for detection issues with this dataset. We next built two species-location matrices whereby the REEF sub-zones were columns (13 columns) and all species recorded as present within any of these sub-zones within any survey were given a unique row (452 rows). One matrix represented survey information from the historical time period, and the other from the current time period.

We entered each matrix into a hierarchical clustering algorithm to generate dendrograms that visually depict the similarities in species composition across sub-zones (R-package hclust). This clustering algorithm proceeds by placing each sub zone in a multidimensional space with the coordinates corresponding to its vector of fish species presence-absence. Using methods from Murtagh and Legendre (2014), the Bray-Curtis dissimilarity score between all sub zones is calculated, which ranges from 0 (exact same composition) to 1 (completely different composition). The two sub zones with the most similar species compositions are then placed into a single cluster. The Lance-Wallace dissimilarity update then calculates the centroid of the newly formed cluster along with the variance around this centroid (Murtagh and Legendre 2014). The process repeats by then measuring the compositional similarity between all sub-zones as well as the newly formed cluster centroid. Once again, the two most similar clusters are combined and then the process repeats until all sub-zones have been assigned a cluster. At the end of this process, the clusters are shown as a dendrogram where clusters joined by short branches have very similar species composition, and longer branch lengths represent increasingly divergent species composition. As a coarse measure of degree of homogenization we calculated the overall branch lengths of each of these two dendrograms (historical and current). If sub-zones have become more similar overall in species composition (homogenized) we expected to see that the current dendrogram has a lower summed branch length than the historical dendrogram; and we would take the opposite pattern (longer summed branch lengths) as evidence of differentiation. We can also more closely track how the cluster affiliation of each sub-zone may have changed between time frames by comparing the higher-order branching patterns between these two dendrograms. We would take as evidence of homogenization a shift toward branch nodes that indicated higher similarity (shift in branch location to the right between time frame). In contrast, substantive differentiation would be characterized by an increase dissimilarity score at higher-order branching nodes (shift in branch location to the left). This more detailed comparison of the two dendrograms allows us to pinpoint sub-zones that are homogenizing and those that are not changing, or are differentiating.

There are several studies that suggest that species composition of a region can change substantially through time while species richness remains largely unchanged (Dornelas et al. 2014). There is also a presumption within many homogenization discussions that sites tend to become more similar in composition due mostly to the increasing presence of ubiquitous exotic species rather than the loss of endemic native species (Olden and Rooney 2006). To explore if these patterns were present in fish assemblages within Floridian coastal waters we calculated historical and current species richness within each sub-zone. We calculated species richness as the simple sum of all species recorded as present within each sub-zone, repeating this calculation for each time frame. We also labeled each species as either native or exotic using USGS Non-Indigenous Aquatic Species database (nas.er.usgs.gov). We also categorized each species as using coral reefs or multiple habitat types using the methods and lists of Luiz et al. (2013) and Alvarez-Filip et al. (2015). This categorization allowed us track whether species lost from sub-zones between time frames were more often those that utilized coral reefs for part of their life cycle.

Changes in richness within a sub-zone over time can occur in two ways. Species can either be absent from the historical list and present in the current, or vice versa. The spatial pattern by which these temporal shifts in species presence occur dictates the extent to which compositional similarities are altered (Olden and Poff 2004; Olden and Rooney 2006). Thus, for example, across sub-zone compositional similarity can increase (homogenize) through the addition of the same species across all sub-zones across time frames. Or, similarity can increase when species found only in on or a few sub-zones disappear between time frames. Differentiation can occur if formerly common species are lost in only a few scattered sub-zones, or if species enter the presence-absence record at only one or a few sub-zones. In order to track such changes, we recorded the number of sub-zones in which each of the 452 species were lost or gained between time frames. We then calculated the percentage change in sub-zone occupancy between time frames by dividing the change in number of sub-zones by the number occupied in the historical time frame. We plotted all species according to their percentage change in sub- zone occupancy, as well as the number of sub-zones they occupied in the historical time frame. This graphic allows one to visualize how species contribute to homogenization and/or differentiation.

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Results

Overall, sub-zones along Florida's Atlantic Coast and Keys experienced a net gain of 67% in species richness. Three sub-zones experienced modest net gains in species richness across time periods, however Florida Bay experienced a massive increase

(Figure 1). Most other sites experienced a slight loss in species richness, or very little change through time (Figure 1). Fifty-six species were observed within at least one sub- zone in the historical time period but were not observed in the current period. Over half (29) of these ‘disappearing’ species were originally seen only within one sub-zone, suggesting that perhaps their loss from the current time species list is due to lack of detection and not true loss. However, 11 species were found within three or more sub-zones in the historical time frame but not at any in the current; which is not as likely due to lack of detection. Seventy-two species were absent in the historical time period but were recorded as present in the current period. Over two-thirds (55) of these ‘appearing’ species were found only within one sub-zone in the current time period, suggesting that they were simply not detected in the historical period. However, nine species appear for the first time within three or more sub-zones in the current time period. The exotic species Pacific lionfish (Pterois volitans) and brassy chub (Kyphosus vaigiensis) appear for the first time in the current time period and occupy five or more sub-zones indicating substantial range expansion into the Florida east coast marine habitats. These two species account for the largest percentage increases in number of sub-zones occupied within Figure 2. Of the 324 species that were present within at least one sub-zone across both time frames, 90 showed no change in the number of sub-zones they occupied, with another 96 showing <20% shifts (positive or negative) (Figure 2). Most of the 82 species that expanded the number of sub-zones they occupied through time moved into one more sub- zone, however six increased by three or more sub-zones. No species that were originally found within four or more sub-zones expanded to occupy any more sub-zones.

In terms of species composition, total branch length for the historical dendrogram was 4.3 whereas total length for the current dendrogram was 4.5, indicating slight overall differentiation in fish assemblage composition across sub-zones through time (Figure 3).

The Florida Bay sub-zone was consistently different in fish assemblage composition as compared to all other sub-zones. The shift of the Saint Mary's Reef to Cape Canaveral sub-zone to become a member of the Looe Key, Biscayne National Park, Long Key and Marquesas Key super-group indicates that it currently has a species composition substantially more similar to the super-group than it did historically. In contrast, Jupiter Inlet to Key Biscayne shifts from grouping with the other super-group to becoming its own unique branch indicating that it has become much more distinct in species composition between time frames (Figure 3). More subtle changes in compositional similarity occur within a super-group consisting of Marathon, Islamorada, Key West, Dry Tortugas, Key Largo, and Cape Canaveral to Jupiter Inlet (Figure 3). Within this super-group there is a shift toward the sites becoming more similar to each other in their species composition across time frames as evinced by a reduction in number of clusters (Figure 3).

Discussion

We show that the fish assemblages along the Atlantic coast of Florida and the Keys have substantially shifted in their assemblage composition through time without experiencing changes in species richness. Although there has been consistent focus on the loss of species and biomass within marine ecosystems, the possibility of changes in spatial diversity have largely gone unexplored (c.f., Alvarez-Filip et al. 2015, Edelist et al. 2006). We show that changes in the Florida Atlantic coast fish assemblage is characterized by larger-scale differentiation while at a smaller scale some regions have homogenized. In particular, Jupiter Inlet to Key Biscayne substantially differentiated from the other sub-zones through time, while a super-group of mostly coral-dominated sites homogenized. Contrast this pattern to what we observed amongst another super-group of coral dominated sub-zones that experienced approximately similar changes in richness through time, but did not homogenize. These sub-zones are some of the more geographically isolated of the set (e.g., Marquesas Key) or are under active protection as National Marine Sanctuaries or National Parks (e.g., Looe Key and Biscayne National Park).

These shifts in assemblage similarity are driven mostly by species losses and not by gains. Change in species presence within sub-zones was largely negative, including within the super-group that showed evidence of homogenization. Although some species are expanding into new sub-zones, this spread seems to be smaller in spatial extent than the range contractions that many other species are undergoing. The two species that spread the most across time frames were exotic species, however we recorded very few exotic species in our dataset. Our results suggest the need to more closely examine the fish assemblages associated with the coral reefs around the Keys, Biscayne Bay, and scattered along the most southern Florida coastline. Narrowing the habitat focus of analysis, and thus also the suite of fish species considered, will make it far easier to examine the role of changes in coral cover, predator diversity, and ocean temperatures in producing homogenization across habitats.

No matter the mechanism, given the economic importance of the fish assemblages in Florida related to dive tourism, our results imply a changing relationship between people and the marine fish they pay to see. We envision the feedback between fish assemblage composition the cultural connections people maintain with these assemblages as a two-way interaction (Figure 4A,B). Existing research links diver activity to their efforts to conserve the species and habitats they enjoy (Dearden et al. 2007; Arin and Kramer 2002) and this provides a framework for understanding when spatial compositional changes will elicit conservation actions from divers and when it will not. For example, novice divers may fail to recognize assemblage change since they are unfamiliar with how unique any given site was previously from all others in terms of fish assemblage. They thus likely will not experience an emotional response to experiencing a homogenized (or differentiated) fish assemblage and will not go on to engage in behaviors that lead to remediation of the site (Anderson and Loomis 2012). Such divers may also fail to recognize their own behaviors are contributing to fish assemblage change or species losses, and they may have low social inclination towards habitat conservation. These feedbacks for novice divers thus either fails to stem changes that lead to assemblage change, or even encourages further change (Figure 4A).

In contrast, divers that regularly visit the same sites over many years have an expectation of what a healthy baseline fish assemblage looks like (Anderson and Loomis 2012). When that baseline begins to change, these more experienced divers undergo an emotional response that may result in changes of their behavior. We posit that experienced divers that value a particular dive site because it is unique in its fish assemblage, and notice that this uniqueness is lost, may value that site less and stop visiting with obvious economic consequences. These experiences may also motivate these more experienced divers to change their own or others behaviors in ways that reduce or even reverse homogenization. For example, divers who are responding to perceived assemblage change may be more likely to participate in diver ‘best practices’, and also facilitate a social community that has similar adherence levels (Figure 4B).

If changes in fish assemblages excites enough of an emotional and economic response from citizens and industry, political actions could arise. Those who rely on eco- tourism are some of the most politically active demographics, which is reflected in the often mandatory requirement of state natural resource managing bodies to have individuals on the boards of these stakeholders (South Atlantic Fishery Management Council Accessed 14 June 2017). If conservation and natural resource managers aim to find compromises that satisfy both conservationists and stakeholders, we must to take into account the cultural and economic influences that are associated with natural systems. Future research on the cultural and social effects on biological homogenization should aim to gather information from individuals who engage with fish assemblages in both a casual and dedicated manner, and connect their perceptions to what species they observe to their willingness to support conservation and management actions.

Literature Cited

Airoldi L, Balata D, Beck MW (2008) The Gray Zone: Relationships between habitat loss and marine diversity and their applications in conservation. Journal of Experimental Marine Biology and Ecology 366 (1-2):8-15. doi:10.1016/j.jembe.2008.07.034

Airoldi L, Turon X, Perkol-Finkel S, Rius M (2015) Corridors for aliens but not for natives: effects of marine urban sprawl at a regional scale. Diversity and Distributions 21 (7):755-768. doi:10.1111/ddi.12301

Alevizon WS, Porter JW (2015) Coral loss and fish guild stability on a Caribbean coral reef: 1974-2000. Environmental Biology of Fishes 98 (4):1035-1045. doi:10.1007/s10641-014-0337-5

Alvarez-Filip L, Dulvy NK, Gill JA, Cote IM, Watkinson AR (2009) Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. Proceedings of the Royal Society B-Biological Sciences 276 (1669):3019- 3025. doi:10.1098/rspb.2009.0339

Anderson LE, Loomis DK (2012) Normative Standards for Coral Reef Conditions: A Comparison of SCUBA Divers by Specialization Level. Journal of Leisure Research 44 (2):257-274

Arin T, Kramer RA (2002) Divers' willingness to pay to visit marine sanctuaries: an exploratory study. Ocean & Coastal Management 45 (2-3):171-183. doi:10.1016/s0964-5691(02)00049-2

Ault JS, Smith SG, Bohnsack JA, Luo JG, Zurcher N, McClellan DB, Ziegler TA, Hallac DE, Patterson M, Feeley MW, Ruttenberg BI, Hunt J, Kimball D, Causey B (2013) Assessing coral reef fish population and community changes in response to marine reserves in the Dry Tortugas, Florida, USA. Fisheries Research 144:28-37. doi:10.1016/j.fishres.2012.10.007

Bhat MG (2003) Application of non-market valuation to the Florida Keys marine reserve management. Journal of Environmental Management 67 (4):315-325. doi:10.1016/s0301-4797(02)00207-4

Bruckner AW (2005) The importance of the marine ornamental reef fish trade in the wider Caribbean. Rev Biol Trop 53:127-137

Dearden P, Bennett M, Rollins R (2007) Perceptions of diving impacts and implications for reef conservation. Coastal Management 35 (2-3):305-317. doi:10.1080/08920750601169584

Dornelas M, Gotelli NJ, McGill B, Shimadzu H, Moyes F, Sievers C, Magurran AE (2014) Assemblage Time Series Reveal Biodiversity Change but Not Systematic Loss. Science 344 (6181):296-299. doi:10.1126/science.1248484

Edelist D, Rilov G, Golani D, Carlton JT, Spanier E (2013) Restructuring the Sea: profound shifts in the world's most invaded marine ecosystem. Diversity and Distributions 19 (1):69-77. doi:10.1111/ddi.12002

Green EP, Bruckner AW (2000) The significance of coral disease epizootiology for coral reef conservation. Biological Conservation 96 (3):347-361. doi:10.1016/s0006-3207(00)00073-2

Hassell NS, Williamson DH, Evans RD, Russ GR (2013) Reliability of Non-Expert Observer Estimates of the Magnitude of Marine Reserve Effects. Coastal Management 41 (4):361-380. doi:10.1080/08920753.2013.804028

Hermoso V, Clavero M, Kennard MJ (2012) Determinants of fine-scale homogenization and differentiation of native freshwater fish faunas in a Mediterranean Basin:

implications for conservation. Diversity and Distributions 18 (3):236-247. doi:10.1111/j.1472-4642.2011.00828.x

Johns G, Lee DJ, Leeworthy V, Boyer J, Nuttle W (2014) Developing economic indices to assess the human dimensions of the South Florida coastal marine ecosystem services. Ecol Indic 44:69-80. doi:10.1016/j.ecolind.2014.04.014

Lamy T, Galzin R, Kulbicki M, de Loma TL, Claudet J (2016) Three decades of recurrent declines and recoveries in corals belie ongoing change in fish assemblages. Coral Reefs 35 (1):293-302. doi:10.1007/s00338-015-1371-2

Lane D, Jones R, Mills D, Wobus C, Ready RC, Buddemeier RW, English E, Martinich J, Shouse K, Hosterman H (2015) Climate change impacts on freshwater fish, coral reefs, and related ecosystem services in the United States. Clim Change 131 (1):143-157. doi:10.1007/s10584-014-1107-2

Levin PS, Holmes EE, Piner KR, Harvey CJ (2006) Shifts in a Pacific ocean fish assemblage: the potential influence of exploitation. Conserv Biol 20 (4):1181- 1190. doi:10.1111/j.1523-1739.2006.00400.x

Luiz OJ, Madin JS, Robertson DR, Rocha LA, Wirtz P, Floeter SR (2012) Ecological traits influencing range expansion across large oceanic dispersal barriers: insights from tropical Atlantic reef fishes. Proceedings of the Royal Society B- Biological Sciences 279 (1730):1033-1040. doi:10.1098/rspb.2011.1525

Magurran AE, Dornelas M, Moyes F, Gotelli NJ, McGill B (2015) Rapid biotic homogenization of marine fish assemblages. Nature Communications 6. doi:10.1038/ncomms9405

Manzello DP, Berkelmans R, Hendee JC (2007) Coral bleaching indices and thresholds for the Florida Reef Tract, Bahamas, and St. Croix, US Virgin Islands. Mar Pollut Bull 54 (12):1923-1931. doi:10.1016/j.marpolbul.2007.08.009

Nyitrai D, Martinho F, Dolbeth M, Baptista J, Pardal MA (2012) Trends in estuarine fish assemblages facing different environmental conditions: combining diversity with functional attributes. Aquat Ecol 46 (2):201-214. doi:10.1007/s10452-012-9392-1

Olden JD, Poff NL (2004) Ecological processes driving biotic homogenization: Testing a mechanistic model using fish faunas. Ecology 85 (7):1867-1875. doi:10.1890/03-3131

Olden JD, Rooney TP (2006) On defining and quantifying biotic homogenization. Global Ecology and Biogeography 15 (2):113-120. doi:10.1111/j.1466- 822x.2006.00214.x

Perera-Valderrama S, Hernandez-Arana H, Ruiz-Zarate MA, Alcolado PM, Caballero-Aragon H, Gonzalez-Cano J, Vega-Zepeda A, Cobian-Rojas D (2016) Condition assessment of coral reefs of two marine protected areas under different regimes of use in the north-western Caribbean. Ocean & Coastal Management 127:16-25. doi:10.1016/j.ocecoaman.2016.04.001

Riegl B, Bruckner A, Coles SL, Renaud P, Dodge RE (2009) Coral Reefs Threats and Conservation in an Era of Global Change. In: Ostfeld RS, Schlesinger WH (eds) Year in Ecology and Conservation Biology 2009, vol 1162. Annals of the New York Academy of Sciences. pp 136-186. doi:10.1111/j.1749- 6632.2009.04493.x

Rota CT, Fletcher RJ, Dorazio RM, Betts MG (2009) Occupancy estimation and the closure assumption. J Appl Ecol 46 (6):1173-1181. doi:10.1111/j.1365-2664.2009.01734.x

Schmitt EF, Sluka RD, Sullivan-Sealey KM (2002) Evaluating the use of roving diver and transect surveys to assess the coral reef fish assemblage off southeastern Hispaniola. Coral Reefs 21 (2):216-223. doi:10.1007/s00338-002-0216-y

Schmitt EF, Sullivan KM (1996) Analysis of a volunteer method for collecting fish presence and abundance data in the Florida keys. Bulletin of Marine Science 59 (2):404-416

Sebens KP (1994) Biodiversity of Coral Reefs-what are we losing and why. Am Zool 34 (1):115-133

Semmens BX, Buhle ER, Salomon AK, Pattengill-Semmens CV (2004) A hotspot of non-native marine fishes: evidence for the aquarium trade as an invasion pathway. Marine Ecology Progress Series 266:239-244. doi:10.3354/meps266239

Threlfall CG, Law B, Banks PB (2012) Sensitivity of insectivorous bats to urbanization: Implications for suburban conservation planning. Biological Conservation 146 (1):41-52. doi:10.1016/j.biocon.2011.11.026

Thrush SF, Gray JS, Hewitt JE, Ugland KI (2006) Predicting the effects of habitat homogenization on marine biodiversity. Ecological Applications 16 (5):1636- 1642. doi:10.1890/1051-0761(2006)016[1636:pteohh]2.0.co;2

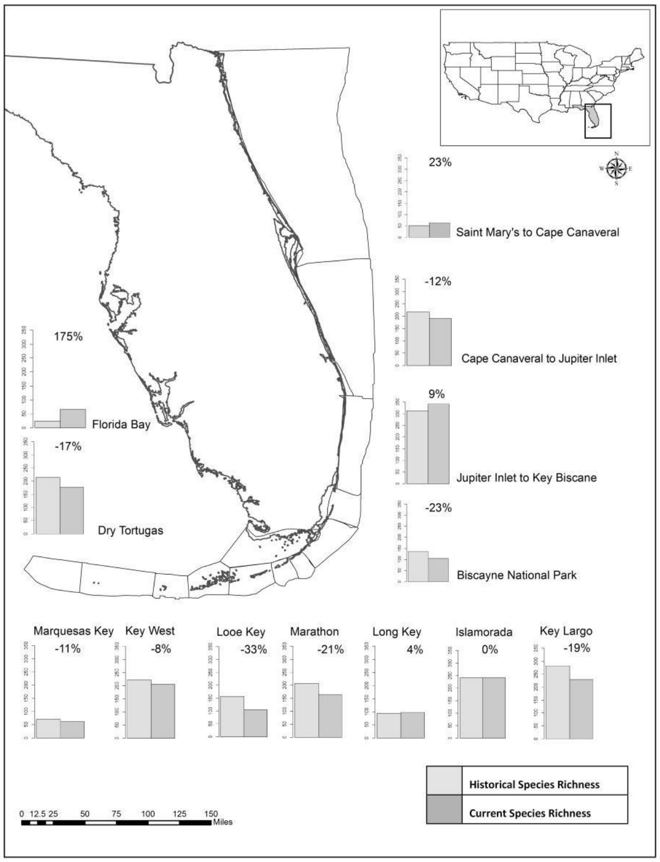


Figure 1. Map of Florida showing the REEF sub-zones considered in this study. Bar graphs correspond to sub-zone with historical species richness in light grey and current species richness in dark grey. Overall changes in species richness from historical to current time frames are reported as percentages. Positive percentages indicate net gain in species richness while negative percentages indicate net loss in species richness.

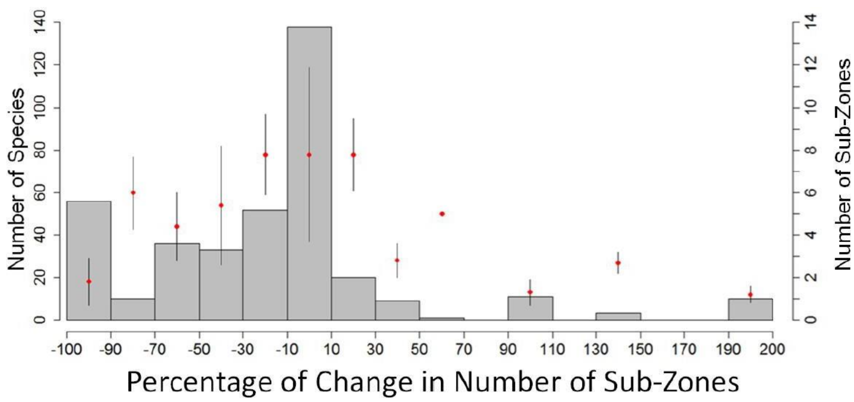


Figure 2. Histogram showing the number of species that show a percentage change in number of sub-zones occupied between the historical and current time periods. Negative percentage change indicates species' range contraction, and positive change indicates species' range expansion. The y-axis indicates the number of species that fall within each percentage change bin. Also shown on the yy-axis is the average number of sub- zones species within each percentile grouping occupied in the historical time period (red dots = average, bars = standard deviation). Many changes in percentage of sub-zones occupied are likely due to failure to detect species in surveys. However, large percentage decreases from species that occupied many sub-zones historically are likely true species declines, and large percentage gains from species that occupied few sub-zones historically likely represent true range expansions.

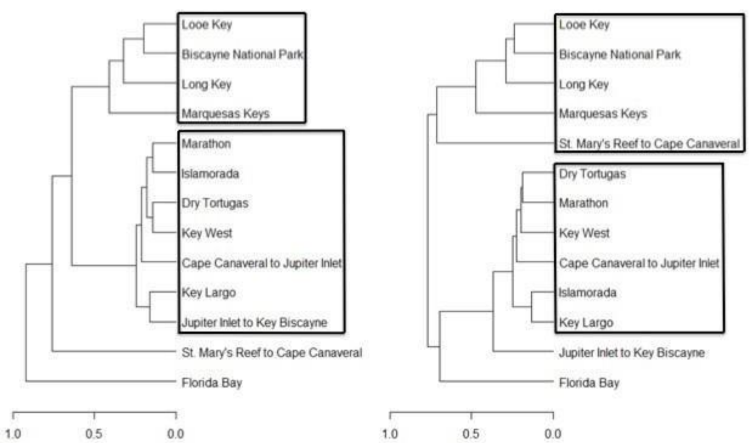
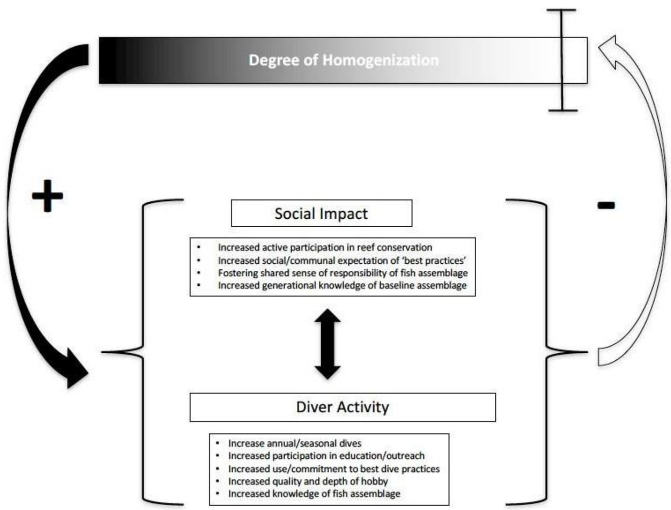


Figure 3. Two dendrograms comparing fish assemblage similarity between reefs from historical (left) and current (right) time frames. Shorter branch lengths indicate higher degree of similarity while longer branch lengths indicate lower degree of similarity, with similarity derived using Bray-Curtis scores.


Degree of Homogenization
+
-
Social Impact
• Increased active participation in reef conservation
• Increased social/communal expectation of 'best practices'
• Fostering shared sense of responsibility of fish assemblage
• Increased generational knowledge of baseline assemblage
Diver Activity
• Increase annual/seasonal dives
• Increased participation in education/outreach
• Increased use/commitment to best dive practices
• Increased quality and depth of hobby
• Increased knowledge of fish assemblage

A

B

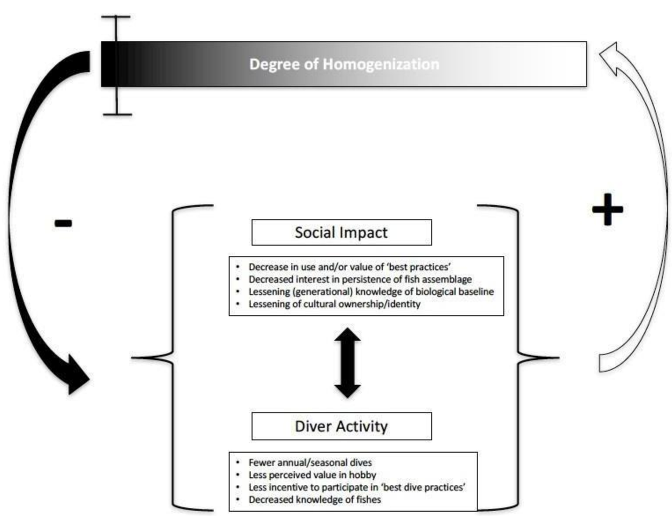

Degree of Homogenization
-
+
Social Impact
• Decrease in use and/or value of 'best practices'
• Decreased interest in persistence of fish assemblage
• Lessening (generational) knowledge of biological baseline
• Lessening of cultural ownership/identity
Diver Activity
• Fewer annual/seasonal dives
• Less perceived value in hobby
• Less incentive to participate in 'best dive practices'
• Decreased knowledge of fishes

Figure 4. Conceptual diagrams illustrating the interplay of SCUBA and snorkel diver cultural and social behavior, and its feedback with biological homogenization. We envision that divers' experience levels will play a strong role in how they respond to homogenization, depicted as a bar above the diagram with increases from right (light) to left (dark). Diver activity levels (experienced or novice) tends to engender particular behaviors relative to adherence to 'best practices', interest in fish species ecology, knowledge of biological baselines, and claiming cultural identity in the fish themselves or dive site. Given this connection, we posit that experienced divers will respond with increased conservation vigilance when realizing that fish assemblages they frequently encounter are homogenizing. The increase in conservation action will then tend to decrease homogenization levels (panel A). In contrast, novice divers may not notice that assemblages have homogenized, and thus will do nothing to prevent further homogenization, or even act in ways that further increase homogenization (Panel B).

Chapter 3: The evidence base surrounding marine mammal–fisheries trophic interactions

Abstract

Rebounding marine mammal populations are conservation successes however, these recovering populations may clash with commercial and artisanal fisheries through ecological interactions that can have economic impacts for the fisheries involved. Here we investigate how marine mammals and fisheries interact through depredation, direct competition, indirect competition and facilitation. We conducted a systematic review of 187 published peer-reviewed articles on marine mammal and fishery interactions. Studies tended to be short in duration, with the majority of studies one year or less in duration. Spatially, there was a regional bias in study location, with 76% of studies coming from three global regions (out of thirteen), the northeast Pacific, northwest Atlantic / Caribbean, and Mediterranean. The most common interaction type reported in the literature was direct competition (58% of all interactions), followed by depredation (27%), indirect competition (13%), and facilitation (2%). Studies on depredation were primarily field-based, while those on indirect competition and facilitation were largely model-based. Odontocetes were the most frequently interacting group, followed by pinnipeds and mysticetes. Although there are many endangered marine mammals that interact with fisheries, we found that the most common interactions reported in the literature occurred with those listed as ‘least concern’. Geographic biases in the study of fishery-marine mammal interactions have created serious gaps un the knowledge-base that supports management.

Key words: whale, dolphin, pinniped, fishery, trophic interaction, systematic review

Introduction

The recovery of marine mammal populations over the last several decades is considered a substantial conservation success (Clapham et al. 1999; Magera et al. 2013). Cetaceans and pinniped species in particular have rebounded toward historically high abundances after having been driven to near extinction from over-exploitation in the previous century (Magera et al. 2013). As many of these species are top or meso-predators in oceanic ecosystems, and often serve as ecosystem engineers (Roman et al. 2014), their recovery increases predation rates of their prey and alters the trophic structure of oceanic food-webs (Surma and Pitcher 2015). When the resurgence of marine mammals results in reductions in the populations of prey that have commercial value, there exists a conflict between conservation and fisheries (Hirsch et al. 2011). The possibility of such a conflict has been used as justification for marine mammal culling, or re-opening or relaxing limits on marine mammal harvests (Smith et al. 2015). Such reactions are highly controversial as it is not always clear that the newly abundant mammals are responsible for declines in fishery catch instead of changes in climate, habitat, or other food web alterations (Lennox et al. 2018). In addition, there is some evidence that, even when marine mammals are feeding on commercially valuable fish, they are not consuming enough individuals to substantively impact the biomass available to fisherman (Gerber et al. 2009; Morissette et al. 2012). The growing concern over the influence of resurging populations of marine mammals on fisheries, and the policy reactions that ensue, has led to a substantial increase in published research efforts over the last decades. Here we conduct a systematic review of this literature summarizing how and where evidence has accumulated over the past 30 years, and conversely, where evidence gaps lie.

Marine mammals interact with fisheries in complex ways throughout their lifetimes, but only a few of these interactions can potentially reduce fishery yields (Schradin et al. 2017; Figure 1). The most obvious form of interaction is when marine mammals consume fish that have been caught on longline or within nets before they can be hauled onto the ship, termed depredation (Read 2008). In some cases, this results in complete loss of fish or damage to fish and fishing gear, resulting in a loss of revenue to the fishery (Hucke-Gaete et al. 2004). Marine mammals can also prey on species that are the target of a fishery, creating a direct competitive interaction between the mammals and the fisherman. This type of interaction is herein termed direct competition. For example, killer whales often consume the same salmon species that are the target of a lucrative fishery in the Pacific Northwest (Chasco et al. 2017). Similarly, grey seal predation has been linked to the delayed recovery of collapsed Atlantic cod populations in the Northwest Atlantic (e.g., Mohn and Bowen 1996). Marine mammals can have indirect trophic interactions with a fishery if the targeted fish population competes with mammals for a shared food resource (herein termed indirect competition). In this case, the fishery may lose potential yield if targeted fish populations decline following marine mammal-induced a reduction in their prey base. For example, baleen whales often consume low trophic level species ("forage fish"), which are often important prey for many commercially important finfish species (Morissette et al. 2010). Marine mammals can also provide benefits to fisheries in various ways (herein termed facilitation). Consumption of higher trophic level fishes by marine mammals could lead to increases in exploited populations of lower trophic level species (e.g., Veit et al. 2017). Another means of facilitation recently identified is the role of marine mammals in recycling limiting nutrients in the surface ocean, most notably iron in the Southern Ocean, which could lead to increased primary production and cascading bottom up increases in exploited fish populations (Nicol et al. 2010)

Our primary goal is to document which of these interactions is most commonly studied within the published literature on marine mammal–fishery interactions. We also seek to characterize which marine mammals and fish species are the subjects of these research publications, and where geographically the evidence for the interaction(s) were gathered. This information can indicate which interactions, involving what species, most often to occur, and where. But it could also reflect an unequal allocation of research effort toward interactions in certain regions that are either easier to document or are considered of more concern by fisheries managers or marine conservation biologists; possibilities we evaluate here.

A key element of the potential conflict between marine mammals and fisheries is the conservation status of the mammals themselves (Marshall et al. 2015). There is clear evidence that marine mammal populations that have received harvest protection have increased over the last 40 years (Magera et al. 2013), yet several species are still considered at risk of extinction (Davidson et al. 2012). It is not clear from existing reviews how often species that are considered of conservation concern have populations that interact with fisheries. Thus, we categorized publications with respect to the International Union for the Conservation of Nature (IUCN) red list categorization of the marine mammal species.

There is some evidence that marine mammal–fisheries interactions are more common when particular gear types are used in the fishery, or in larger commercial fishery operations. For example, depredation events are often more reported in longline fisheries (Sigler et al. 2008). Similarly, in the sablefish fishery, sperm whale depredation is thought to be common due to the large spatial scale that the fishery occupies (Peterson et al 2017). We evaluate if the current evidence base suggests that interactions type is often associated with particular gear or the scale of the fishery. We also evaluate whether there are gear types that are more often associated with particular species of marine mammals overall, and in particular, with species that are of conservation concern.

Finally, we recorded two aspects of how data was assembled and used to evaluate marine mammal–fisheries interactions; study length and analytical approach. Long-term ecological studies provide needed evidence for how ecological interactions change through time and how individuals may alter their feeding rates or trophic interactions as they move between life or reproductive stages (Lindenmayer et al 2012; Sijm et al. 1992). Capturing such context is particularly important in marine mammal-fisheries interactions as the focal species are often exceptionally long-lived (e.g., whales), or have complex life histories (e.g., salmon). Thus, we recorded the length of time over which information used in each analysis was collected. The difficulty in tracking these types of interactions, especially over long time scales, often necessitates the use of models, particularly ecosystem models (Smith et al. 2011). Such models allow researchers to quantitatively evaluate the long-term outcomes of trophic interactions, and how management and policy options can alter these outcomes (Mendoza et al. 2006). The use of models in this context is thus a critical form of evaluative evidence (Nelson et al. 2009), and we therefore recorded whether a publication was model-based, or whether is based on an analysis of field-based observations.

Methods

A single person (ADS) searched primary literature published between 1980 and December 2018 using Web of Science and Google Scholar. We developed search terms intended to capture a wide range of publications on marine mammal interactions with fisheries. We did not constrain studies by location. The terms we entered in the search engines were: TOPIC = ("Marine mammal" OR pinniped* OR whal* OR cetacea* OR "sea lion" OR sea-lion OR dolph* OR otter* OR dugon* OR seal* OR "fur seal" OR "grey seal") AND (fisheries* OR fishery) AND (depredat*) NOT "otter trawl" NOT "whale shark" NOT "whale worm"; TOPIC = (("Marine mammal" OR pinniped* OR whal* OR cetacea* OR "sea lion" OR sea-lion OR dolph* OR otter* OR dugon* OR seal* OR "fur seal" OR "grey seal") AND (fisheries* OR fishery) AND (compet* OR interact* OR oppos* OR conflict*)); and (((("Marine mammal*" OR pinniped* OR whal* OR cetacea* OR "sea lion" OR sea-lion OR dolph* OR otter* OR dugon* OR seal* OR "fur seal" OR "grey seal") AND (fisheries* OR fishery) AND (compet* OR interact* OR oppos* OR conflict* OR consum* OR trophic)))).

We acknowledge that these search terms will not capture every published article on marine mammal interactions with fisheries. However, by using these search terms, we found 1,147 unique articles. We included only search terms and articles in English. Limiting search results to articles published in English may introduce language bias. By limiting our results to English we may introduce regional bias as well.

Screening and Study Inclusion Criteria

During screening, we checked for consistency with inclusion criteria following steps recommended by Preferred Reporting Items for Systematic Reviews and Meta-Analysis (PRISMA) (Moher et al. 2009). Owing to the large number of candidate studies, we performed screening in three steps. In the initial step, we checked for consistency with criteria by reading all titles. In the second step, we read all remaining abstracts. In the third screening step, we downloaded and read the full text of all remaining candidate articles (N = 235), and based on this reading, decided on their disposition (include or exclude). After the final screening step, we were left with 187 articles that we included in our systematic review (Figure 2).

Prior to article screening, we developed a set of inclusion criteria (Table 1). Each article had to meet all criteria to be included in our final database. Articles had to be original research involving marine mammal–fishery interactions. These interactions could be directly observed in nature or hypothesized to occur based on empirical data. For articles to be included, authors had to discuss either a hypothesized or directly observed marine mammal–fishery interaction.

Data Extraction

From each included article, we used a data form to extract author, interaction type, taxon, conservation status of mammals, study duration, gear type and size of fishery (Table 1). We piloted data extraction using 10% of the articles to ensure standardized application of definitions across articles. One researcher was responsible for the data extraction (ADS). One article could fall under multiple categories. For example, each paper could provide information on multiple species or interactions. We extracted over 739 lines of data from the included 187 articles.

We defined interaction types as depredation, direct competition, indirect competition and facilitations following Figure 1 and descriptions above. Long-term studies were defined as four or more years of consecutive field monitoring. Short-term studies were defined as three or fewer years of consecutive field monitoring. For studies that utilized models we assigned the model run years as the duration of the study. Fisheries were categorized as either commercial or artisanal, where the former is defined as large operations while the latter as small-scale local operations. Gear types were assigned by both the kind of gear that was used as well as the trophic interaction associated with the gear. We identified fish and marine mammals down to the species level using Jefferson et al (2011). IUCN status was used to label species as 'least concern', 'vulnerable', 'endangered', 'critically endangered' or 'data deficient' according to IUCN (2019).

We defined 13 ocean regions following Lewison et al. (2014), who used empirical data from peer-reviewed articles, agency and technical reports and symposia proceeding published between 1990 and 2008 to identify the global distribution and magnitude of marine mammal bycatch from fisheries. We use this by-catch interaction map as a guide to where marine mammals may also be directly or indirectly influencing fisheries catch (Figure 3).

Results

Of the 187 papers identified in our search, 60% were classified as field-based studies, with the remaining 40% classified as model-based studies. The most well studied regions were the northeast Pacific, Mediterranean, and northwest Atlantic / Caribbean, comprising 76.5% of the studies found in our search (Figure 4). All of the remaining regions had fewer than 15 studies, with fewer than 5 studies in four regions. The southwest Pacific did not have any studies.

In their review on bycatch studies, Lewison et al. (2014) identified six regions with a high occurrence of marine mammal bycatch: the northeast Pacific, Mediterranean, southwest Pacific, eastern tropical Pacific, southwest Atlantic, and Western Indian Ocean. We found that there was also a high occurrence of studies on marine mammal and fishery interactions in the northeast Pacific and Mediterranean (Figure 3). The top sites for by-catch as reported by Lewison et al. (2014) were the northeast Pacific, Mediterranean, southwest Pacific, eastern tropical Pacific, southwest Atlantic, and Western Indian Ocean. Although we found high publication rates related to marine mammal-fisheries interactions in two of these regions (the northeast Pacific and Mediterranean), the other regions were less well covered in our review, with one region of high bycatch having no studies on marine mammal – fishery trophic interactions (southwest Pacific). Furthermore, some of the more well studied regions in our review were not well covered in terms of bycatch data (northwest and northeast Atlantic, and Southern Ocean; Figure 4). Thus, although there was some coherence in study locations

for marine mammal bycatch and trophic-fishery interactions, there were many regions well studied for bycatch but not for trophic-fishery interactions, or vice-versa.
Enough information to determine study duration was available in 79 of the 187 studies. Of those 79, the majority of studies occurred over one year or less (51%), while long-term studies (>10 years) comprised 23% of studies (Figure 5). As the duration of study increased, the likelihood of it being a modeling study increased. Field studies accounted for 83% of studies ≤ 1 year, and 46% of studies between 2 to 5 years in length. For studies longer than 5 years, however, only 19% of the studies were field-based (Figure 5).

From the 187 studies, we tabulated a total of 738 trophic interactions between marine mammals and fisheries. The majority of studies in our review focused on single marine mammal (65%), whereas there was a roughly even split across studies focusing on one or multiple fish species (49 % and 51%, respectively). Thus, a single study with multiple mammals, fish species, or both, could result in many trophic-fishery interactions. Direct competition was the most common interaction (N=427), followed by depredation (N= 199; Figure 6). Indirect competition (N = 95) and facilitative interactions (N = 17) were less frequently addressed in published literature we assembled. Field-based studies accounted for 53% of the total number of interactions, but there was a bias in the interaction type, and whether it came from a modeling or field based study. Depredation accounted for 27% of all interactions, and was primarily covered in field-based studies (95% of depredation interactions). Indirect competition and facilitation accounted for 13% and 2% of all interactions respectively, and were primarily found in model-based studies (5% and 17% of the interaction for each, respectively, were from modeling studies). Direct competition accounted for 46% of all interactions, and was more evenly distributed by study type, with 45% of the direct competition interactions coming from field-based studies (Figure 6).

Odontocetes were the most frequently interacting group of marine mammals across studies (N=363 interactions or 49% of all interactions; Figure 7). Killer whales, sperm whales, and bottlenose dolphins were the most common species involved in these interactions composing 44% of all odontocete interactions (Figure 7). Pinnipeds were a close second in interaction frequency to odontocetes (35% of all interactions), although the number of interactions were spread out over a greater number of species, with only two species within the top 10 (harbor seals and gray seals were the 8th and 9th most frequently interacting species, respectively; Figure 7). Mysticetes were the least frequently interacting group (14% of all interactions), although humpback, minke and fin whales were the 4th, 5th and 6th most frequently studied mysticete species (N=40; Figure 7).

Direct competition was the most common interaction across groups, with odontocetes being the most frequently interacting group in direct competition (N=195), followed by pinnipeds (N=161) and mysticetes (N=68) ;Figure 8). The fisheries associated with direct trophic interactions targeted anchovy and Antarctic tooth fish, with cod, herring krill, and sardines also represented in the literature base. Depredation was the second most common interaction for odontocetes (N=133) and pinnipeds (N=66), but there were no documented cases of depredation for mysticetes. Of the depredation interactions involving nets, only pinnipeds were involved, and the associated fisheries most often involved were those targeting Patagonian tooth fish and sablefish. For indirect competition, pinnipeds were most frequently documented (N=35), closely followed by odontocetes and mysticetes (N=28, and 24, respectively; Figure 8). All groups were involved in facilitative interactions, with mysticetes (N=8) and odontocetes (N=7) more frequently documented in facilitation than pinnipeds (N=2).

Of the studies where specific information on the fishery type was available (N=187), the majority (95%) involved commercial fisheries as opposed to artisanal fisheries. However, only 25% of all studies specified which gear type was observed in their study. With regard to the fishing gears, depredation was most commonly observed in longline fisheries (30% of all depredation studies), with killer whales and sperm whales the most common species involved in longline depredation. Pinnipeds and bottlenose dolphins were most commonly associated with net depredation. Purse seine gear was often associated with direct competition. Facilitation did not show bias toward gear type. Facilitation was found across net and hook gears.

In regards to IUCN status, the category 'least concern' composed the majority of species (57%) for marine mammals with 'data deficient' species as the second most common status type (18%). We found that the least common IUCN status was 'critically endangered' (N=1; Figure 9). Globally, we found that 43% of all endangered marine mammals interact with fisheries (Figure 9). We also found that 39% of all vulnerable marine mammals participate in fishery interactions and 30% of near threatened species participate in fishery interactions. The endangered and vulnerable species listed in Table 3 were found to interact with fisheries through direct competition, primarily through depredation. Endangered and vulnerable species interacted with several different gear types including but not limited to gill nets and longline. The critically endangered species, the vaquita, interacted through depredation as well in the mackerel fishery that used longlines.

Discussion

The global recovery of marine mammals is a conservation success. Studies found that marine mammal populations were recovering and as they recovered they had increased interactions with fisheries which often lead to their mortality (Read and Wade 2000). As marine mammals recover, their interactions with humans over shared natural resources will increase.

I found that direct competition was the most common conflict involving marine mammals and fisheries, and that odontocetes and pinnipeds compose the majority of direct competition and depredation interactions. Magera et al. 2013 found that 23% of all toothed whale populations are recovering, which suggests that the resurgence of odontocetes may be leading to increasing occurrences of fisheries depredation. Studies that focused on depredation often did not report the quantity of fish removed from the catch. In order to better understand the biological and economic impact of depredation, fishermen and managers should be sure to quantify the amount of catch lost per depredation event. By quantifying the amount of catch lost per depredation event, managers will gain more insight into setting appropriate catch limits and will be able to better inform fisheries models that account for whale depredation. Mysticeti whales were less frequently documented as involved in conflicts with fisheries, which is likely due to at least two factors. First, baleen whales were hunted to near extinction during the commercial whaling eras (Mackintosh 1965), leaving many populations at low current

Commented [JW11]: Get rid of WE and replace with I here and throughout abundances and much harder to observe. Second, Mysticeti whales are filter feeders that will not interact directly via depredation with longline or net fisheries; whereas toothed whales and pinnipeds have feeding strategies and prey preferences that highly overlap longline fishing targets.

The baleen whales were the least frequently studied group of marine mammals. This may be due to their historically low abundances after commercial whaling drove stocks of baleen whales to near extinction. In addition to their low abundances, due to their large size, baleen whales travel great distances in the open ocean which makes it difficult to directly observe them (Lascelles et al. 2014).

I found that facilitation was the least published marine mammal-fishery interaction. This may be because indirect interactions are harder to directly observe in the wild. Although it is difficult to observe and by extension quantify indirect interactions such as facilitation, it is still important to account for these interactions when considering policy and management (Whipple et al. 2000). By under reporting occurrences of facilitation and other indirect marine mammal-fishery interactions, I potentially bias the influence of indirect interactions in fisheries (Baskett 2007).

Marine mammals and fisheries are part of a greater ecosystem that includes multiple marine mammal species and multiple fisheries. Therefore, one might expect that research on marine mammal-fisheries interactions will commonly include information on indirect and direct interactions between all species components of the ecosystem. However, the published evidence I summarize here was skewed towards considering a single marine mammal species. In contrast, I found that most articles did include multiple fish species in their analysis. Articles should include multiple marine mammals because more than one species of marine mammals contribute to food web dynamics in the ecosystem. It is also possible that by only looking at a single marine mammal species in these models may bias our understanding of how that species contributes to ecosystem dynamics. By only including a single marine mammal species, I lose the complexity of how the food web functions.

Marine mammals are widely distributed across the oceans, but most evidence related to their interactions with fisheries stem from only a few oceanic regions. Lewison et al. (2014) found that fishing effort and marine mammal bycatch was largely concentrated along continental coastlines with relatively few open ocean fisheries. In this literature review, I use the marine mammal bycatch from Lewison et al. (2014) as a proxy for expectations regarding where marine mammal interactions are likely to occur. Marine mammal bycatch is a good proxy for predicting where marine mammal and fishery interactions might occur because it provides direct evidence of fishery-marine mammal interactions in space and time. Additionally, the distribution of marine mammals and the subsequent interactions with fisheries is likely influenced by both oceanographic and physiological forces (Block et al. 2011). For example, the distribution of fishing effort and by extension the distribution of marine mammals interacting with those fisheries is known to be non-random with greater fishing effort concentrated in productive Eastern Boundary Currents and shallow coastal zones (Middleton 2000). Our results indicate that where fishing effort and marine mammal bycatch is highest, there is also more published evidence of marine mammal–fishery interactions. For example, there were high occurrences of marine mammal-fishery interactions in the northwest Atlantic / Caribbean and Mediterranean. These regions host some of the most productive fisheries. This implies that research effort is following trends of fishing effort. Regional biases may also be due to the uneven distribution of research interest and effort. For example, the Northeast Pacific was a highly studied region. This may be due to the high interest in researching endangered salmon and killer whales.

However, there were regions from Lewison et al. (2014) that had high fishery activity coinciding with high diversity of marine mammal bycatch, but for which I did not find a similarly large number of published evidence of interactions. For example, the southwest Pacific, eastern Tropical Pacific, southwest Atlantic, and western Indian Ocean were regions in Lewison et al. (2014) that had high occurrences of marine mammal bycatch and fishery activity, but for which I found few published articles. This result suggests that these regions are under-studied when, in fact, marine mammal and fishery interactions in these places are common. Finally, I found some regions that have several published articles on mammal-fishery interactions but for which Lewison et al. (2014) indicate that such either mammals are rare there or the fisheries in that region are less active. This discrepancy suggests that, in these regions, either there is a very strong and consistent interaction that is of high concern and thus under active investigation; or, that research effort is high in these locations for reasons other than the common occurrence of interactions.

Most studies occurred over one year or less, which suggests that the existing evidence base likely misses a tremendous amount of information on long-term dynamics between marine mammals and fisheries. The foraging dynamics of marine mammals likely changes across years based on changes in prey distributions and abundance, but also based on ontogenetic changes in mammal foraging. Thus, there is a need for long-term studies in order to capture the temporal variability of the mammal-fishery interactions. Additionally, long term studies are needed in order to improve the data that may be used in modelling these interactions. Long-term studies in our review tended to be model-based, and constructing models of these interactions is a powerful way to explore and predict the outcome of marine mammal-fishery interactions given the limited availability of long-term observational studies (Bax 1998).

Long-term studies were able to gain insight into more complex ecological phenomena (Machado *et al*. 2018; Gavrilchuk *et al*. 2014; Scheinin *et al.* 2014; Bearzi *et al*. 2006). For example, one long-term study was able to show evidence of a trophic cascade following consumption of certain prey items by rorqual whales (Gavrilchuk *et al*. 2014). The long-term nature of the study provided researchers with enough data to be able to observe trends that would not have been obvious through short-term studies. We define long-term studies as more than five years, which appears to be the length of time necessary to observe long-term trends in data sets. For example, in a study by Reid *et al.* 2005, the authors looked at predator consumption of icefish data for over five years and found that while consumption of icefish by marine mammal predators was high in some years of low krill abundance but not all, the pattern of icefish consumption was explained better by putting it in the context of marine mammal recovery. The authors highlight that the study of exploited species should be explored in the context of changing ecosystems. In order to observe these changes and the effects they have on the rest of the relationships in the ecosystem, a long-term study approach should be used like the one in Reid *et al*. 2005. More long-term studies may provide us with direct insight into dynamics that are harder to quantify through short term studies such as indirect interactions like trophic cascade.

I found that well-studied oceanic regions, like the Mediterranean and northeast Pacific, tended to be the subject of published articles that utilized both modelling and field-based observations. However, regions such as the eastern Indian Ocean, and southeast Pacific Ocean have very few published articles on marine mammal-fishery interactions and, what information does exist, is from field-based observations only. Regions where there are only field-based observations may be missing indirect trophic interaction complexities that can be shown through models.

The diverse set of challenges in studying the trophic ecology of marine systems are well acknowledged, particularly for organisms such as marine mammals that can travel great distances making direct observations difficult (e.g., Hays et al. 2016). Because marine mammals travel great distances it makes them inherently more difficult to account for in modelling studies. It is therefore not surprising that field-based studies on trophic conflict between marine mammals and fisheries focus on interactions that are directly observable (i.e., depredation), and I found that the vast majority of depredation studies were field-based. In contrast, studies on more complex trophic interactions that span multiple species or trophic levels are more difficult using field-based observations alone. In such cases, models are an essential tool for understanding the complex trophic interactions. Our finding that studies on indirect competition and facilitation were largely model based, therefore makes sense.

I found that marine mammals with the IUCN status of ‘least concern’ interact most frequently with fisheries. This finding makes sense given that a more abundant species is more likely to have a large impact on fishery through depredation or competition. Nevertheless, there are several marine mammals that are considered ‘vulnerable’ or ‘endangered’ that interact with fisheries. In this instance managers are faced with a fisheries-conservation conflict, forcing them to consider tradeoffs between maximizing economic yield while also not jeopardizing the recovery of marine mammals. For example, in European fisheries the recovery of marine mammals is a conservation success; however, as recovery progresses, the intensity of conflicts over shared resources intensifies and prey species may become endangered (Rauschmayer et al. 2007). Similarly, in the Atlantic salmon and brown trout fisheries the smolt of these endangered species is eaten as a prey item by recovering marine mammals (Koed et al. 2006). In this case conservation and fisheries management must be flexible enough to adapt to the changing circumstances of predator-prey dynamics.

Most fisheries were commercial as opposed to artisanal, which may indicate that more research is needed on artisanal fishery and marine mammal interactions. Although artisanal fisheries are small scale operations, they still have a local impact on marine mammal-fishery interactions (Moore et al. 2010). High seas and industrial fisheries have been the focus of more data collection than coastal or small-scaled fisheries (Lewison et al 2013) largely due to investments in research by industrial fisheries and the national and regional management organizations that oversee these fisheries (Lewison and Crowder 2007). There is also the issue of spatial scale of artisanal fisheries. Artisanal fisheries will not come into as frequent contact with marine mammals as commercial fisheries. For example, the artisanal fisheries in Nepal struggle with conservation and management of the Ganges River dolphin. Although this fishery is small, it still contributes to the anthropogenic activities that threaten the endangered dolphin. As a way to combat these conservation challenges, artisanal fisheries in Nepal have tried to switch gear types used in the fishery to avoid interacting with the Ganges River dolphin (Paudel et al. 2016). In addition to switching gear types, artisanal fisheries will try to avoid the same location where endangered marine mammals feed (Dudgeon 2003).

I have gone through the literature of marine mammal and fishery interactions and found that marine mammals primarily interact through direct competition with fisheries. I also found that odontocetes primarily interact through direct competition in the form of depredation. This makes sense when comparing recovery efforts across marine mammal species as odontocetes are slowly rebounding in population sizes. Although there are many endangered marine mammals that interact with fisheries, I found that the most common interactions occurred with marine mammals listed as ‘least concern’. Given the recovery of these predator species and the clear need for healthy fisheries we must learn more about where these interactions occur in the high seas as opposed to limiting our observations to coastal interactions. We must also invest in monitoring these interactions over several years to be sure to capture the complexity of the food-web dynamics in the ecosystem. By quantifying these observations we gain more insight into fishery-conservation tradeoffs which allows us to inform future sustainable fisheries management and conservation of marine mammals.

Literature Cited

Baker, C.S. and P.J. Clapham. 2004. Modelling the past and future of whales and whaling. Trends in Ecology and Evolution 19(7):365 - 371

Baskett, ML. 2007. Simple fisheries and marine reserve models of interacting species: An overview and example with recruitment facilitation. California cooperative oceanic fisheries investigation reports. 48:71 – 81.

Bax, N. J. 1998. The significance and prediction of predation in marine fisheries. ICES Journal of Marine Science 55:997-1030.

Bearzi, G., Politi, E., Agazzi, S. and A. Azzellino. 2006. Prey depletion caused by overfishing and the decline of marine megafauna in eastern Ionian Sea coastal waters (central Mediterranean). Biological Conservation 127(4): 373 – 382.

Block, B. A., I. D. Jonsen, S. J. Jorgensen, A. J. Winship, S. A. Shaffer, S. J. Bograd, E. L. Hazen, D. G. Foley, G. A. Breed, A. L. Harrison, J. E. Ganong, A. Swithenbank, M. Castleton, H. Dewar, B. R. Mate, G. L. Shillinger, K. M. Schaefer, S. R. Benson, M. J. Weise, R. W. Henry and D. P. Costa. 2011. Tracking apex marine predator movements in a dynamic ocean. Nature 475:86-90. 68

Bowen, W. D. and D. Lidgard. 2012. Marine mammal culling programs: review of effects on predator and prey populations. Mammal Review. 43(3): 207 – 220.

Chasco, B. E., I. C. Kaplan, A. C. Thomas, A. Acevedo-Gutierrez, D. P. Noren, M. J. Ford, M. B. Hanson, J. J. Scordino, S. J. Jeffries, K. N. Marshall, A. O. Shelton, C. Matkin, B. J. Burke and E. J. Ward. 2017. Competing tradeoffs between increasing marine mammal predation and fisheries harvest of Chinook salmon. Scientific Reports 7(14): 1 - 14.

Commented [JW12]: Pages?

Clapham, P. J., S. B. Young and R. L. Brownell. 1999. Baleen whales: conservation issues and the status of the most endangered populations. Mammal Review 29:35-60.

Davidson, A. D., A. G. Boyer, H. Kim, S. Pompa-Mansilla, M. J. Hamilton, D. P. Costa, G. Ceballos and J. H. Brown. 2012. Drivers and hotspots of extinction risk in marine mammals. Proceedings of the National Academy of Sciences of the United States of America 109:3395-3400.

Dudgeon, D. 2003. The contribution of scientific information to the conservation and management of freshwater biodiversity in tropical Asia. Hydrobiologia 500:295-314.

Estes, J. A., E. M. Danner, D. F. Doak, B. Konar, A. M. Springer, P. D. Steinberg, M. T. Tinker and T. M. Williams. 2004. Complex trophic interactions in kelp forest ecosystems. Bulletin of Marine Science 74:621-638.

Gavrilchuk, K., Lesage, V., Ramp, C., Sears, R., Berube, M., Bearhop, S., and G. Beauplet. 2014. Trophic niche partitioning among sympatric baleen whale species following the collapse of groundfish stocks in the Northwest Atlantic. Marine Ecology

Progress Series. 497:285-297.

Gerber, L. R., L. Morissette, K. Kaschner and D. Pauly. 2009. Should Whales Be Culled to Increase Fishery Yield? Science 323:880-881.

Hays, G. C., L. C. Ferreira, A. M. M. Sequeira, M. G. Meekan, C. M. Duarte, H. Bailey, F. Bailleul, W. Don Bowen, M. J. Caley, D. P. Costa, V. M. Eguiluz, S. Fossette, A. S. Friedlaender, N. Gales, A. C. Gleiss, J. Gunn, R. Harcourt, E. L. Hazen, M. R. Heithaus, M. Heupel, K. Holland, M. Horning, I. Jonsen, G. L. Kooyman, C. G. Lowe, P. T. Madsen, H. Marsh, R. A. Phillips, D. Righton, Y. Ropert-Coudert, K. Sato, S. A. Shaffer, C. A. Simpfendorfer, D. W. Sims, G. Skomal, A. Takahashi, P. N. Trathan, M. Wikelski, J. N. Womble and M. Thums. 2016. Key Questions in Marine Megafauna Movement Ecology. Trends in Ecology & Evolution 31:463-475

Hirsch, P. D., W. M. Adams, J. P. Brosius, A. Zia, N. Bariola and J. L. Dammert. 2011. Acknowledging Conservation Trade-Offs and Embracing Complexity. Conservation Biology 25:259-264.

Hucke-Gaete, R., C. A. Moreno, J. Arata and C. Blue Whale. 2004. Operational interactions of sperm whales and killer whales with the Patagonian toothfish industrial fishery off southern Chile. CCAMLR Science 11:127-140. 69

Jefferson, T. A., M. A. Webber and R. L. Pitman. 2011. Marine mammals of the world: a comprehensive guide to their identification. Elsevier.

Julien Collet, G. R., Anais Janc, Christophe Guinet, Henri Weimerskirch. 2018. Influence of depredating cetaceans on albatross attraction and attendance patterns at fishing boats. Marine Ecology 605:49 - 59

Juni, P., F. Holenstein, J. Sterne, C. Bartlett and M. Egger. 2002. Direction and impact of language bias in meta-analyses of controlled trials: empirical study. International Journal of Epidemiology 31:115-123.

Kristin Kaschner1, Konstantinos I. Stergiou, Gunna Weingartner1 and A. S. Kumagai. 2004. Trophic levels of marine mammals and overlap in resource utilization between marine mammals and fisheries in the Mediterranean Sea. CISEM 25:51 - 58.

Koed, A., H. Baktoft and B. D. Bak. 2006. Causes of mortality of Atlantic salmon (Salmo salar) and brown trout (Salmo trutta) smolts in a restored river and its estuary. River Research and Applications 22:69-78.

Lewison, R. L. and L. B. Crowder. 2007. Putting longline bycatch of sea turtles into perspective. Conservation Biology 21:79-86.

Lewison, R. L., L. B. Crowder, B. P. Wallace, J. E. Moore, T. Cox, R. Zydelis, S.
Lotze H., Coll M., Mager A., Ward-Paige C., and L. Airlodi. 2011. Recovery of marine

animal populations and ecosystems. Trends in Ecology and Evolution. 26:11.

Lindenmayer, D. B., G. E. Likens, A. Andersen, D. Bowman, C. M. Bull, E. Burns, C. R. Dickman, A. A. Hoffmann, D. A. Keith, M. J. Liddell, A. J. Lowe, D. J. Metcalfe, S. R. Phinn, J. Russell-Smith, N. Thurgate and G. M. Wardle. 2012. Value of long-term ecological studies. Austral Ecology 37:745-757.

Machado, R., de Oliveria, L.R., Ott, P.H., Denuncio, P., Haimovici, M., Cardoso, L., Danilewicz, D., Moreno, I., and M. Borges-Martins. 2018. Changes in the feeding ecoogy of South American sea lions on the southern Brazilian coast over the last two decades of excessive fishing exploration. Hydrobiologia. 819(1):17 – 37.

Mcdonald, A. Dimatteo, D. C. Dunn, C. Y. Kot, R. Bjorkland, S. Kelez, C. Soykan, K. R. Stewart, M. Sims, A. Boustany, A. J. Read, P. Halpin, W. J. Nichols and C. Safina. 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. Proceedings of the National Academy of Sciences of the United States of America 111:5271-5276.

Mackintosh, N. 1965. The stock of whales. Fishing news.

Magera, A. M., J. E. M. Flemming, K. Kaschner, L. B. Christensen and H. K. Lotze. 2013. Recovery Trends in Marine Mammal Populations. PLOS One 8(12): 1 - 12.

Marshall, K. N., A. C. Stier, J. F. Samhouri, R. P. Kelly and E. J. Ward. 2016. Conservation Challenges of Predator Recovery. Conservation Letters 9:70-78.

Mendoza, G. A. and H. Martins. 2006. Multi-criteria decision analysis in natural resource management: A critical review of methods and new modelling paradigms. Forest Ecology and Management 230:1-22. 70

Middleton, J. F. 2000. Wind-forced upwelling: The role of the surface mixed layer. Journal of Physical Oceanography 30:745-763.

Misaki, M. K. and M. Komastu. 2001. The Truth Behind the Whaling Dispute. Tokyo: Institute of Cetacean Research.

Mohn, R. and W. D. Bowen. 1996. Grey seal predation on the eastern Scotian Shelf: Modelling the impact on Atlantic cod. Canadian Journal of Fisheries and Aquatic Sciences 53:2722-2738.

Moore, JE., Cox, TM., Lewison, RL., Read, AJ., Bjorkland, R., McDonald, SL., Crowder, LB., Aruna, E., Ayissi, I., Espeut, P., Joynson-Hicks, C., Pilcher, N., Poonian, CNS., Solarin, B., and Kisza, J. 2010. An interview-based approach to assess marine mammal and sea turtle captures in artisanal fisheries. Biological Conservation. 143:795-805

Morissette, L., V. Christensen and D. Pauly. 2012. Marine Mammal Impacts in Exploited Ecosystems: Would Large Scale Culling Benefit Fisheries? PLOS One 7(18): 1 - 12.

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Morissette, L., K. Kaschner and L. R. Gerber. 2010. 'Whales eat fish'? Demystifying the myth in the Caribbean marine ecosystem. Fish and Fisheries 11:388-404.

Nelson, E., G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D. R. Cameron, K. M. A. Chan, G. C. Daily, J. Goldstein, P. M. Kareiva, E. Lonsdorf, R. Naidoo, T. H. Ricketts and M. R. Shaw. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7:4-11.

Nicol, S., A. Bowie, S. Jarman, D. Lannuzel, K. M. Meiners and P. Van Der Merwe. 2010. Southern Ocean iron fertilization by baleen whales and Antarctic krill. Fish and Fisheries 11:203-209.

Paudel, S., J. C. Levesque, C. Saavedra, C. Pita and P. Pal. 2016. Characterization of the artisanal fishing communities in Nepal and potential implications for the conservation and management of Ganges River Dolphin (Platanista gangetica gangetica). Peerj 4:23.

Peterson, M. J. and D. Hanselman. 2017. Sablefish mortality associated with whale depredation in Alaska. Ices Journal of Marine Science 74:1382-1394.

Rauschmayer, F., H. Wittmer and A. Berghoefer. 2008. Institutional challenges for resolving conflicts between fisheries and endangered species conservation. Marine Policy 32:178-188.

Read, A. J. 2008. The looming crisis: Interactions between marine mammals and fisheries. Journal of Mammalogy 89:541-548.

Read, A.J. and P.R. Wade. 2000 Status of marine mammals in the United States. Conservation Biology. 14:929 - 940

Reid, K., Hill, S.L., Diniz, T.C.D., and M.A., Collins. 2005. Mackerel icefish Champsocephalus gunnari in the diet of upper trophic level predators at South Georgia: implications for fisheries management. Marine Ecology Progress Series. 305:153 – 161.

Roman, J., J. A. Estes, L. Morissette, C. Smith, D. Costa, J. Mccarthy, J. B. Nation, S. Nicol, A. Pershing and V. Smetacek. 2014. Whales as marine ecosystem engineers. Frontiers in Ecology and the Environment 12:377-385.

Schradin, C. and L. D. Hayes. 2017. A synopsis of long-term field studies of mammals: achievements, future directions, and some advice. Journal of Mammalogy 98:670-677.

Scheinin, A.P., Kerem, D., Lojen, S., Liberzon, J., and E. Spanier. 2014. Resource portioning between common bottlenose dolphin (Tursiops truncatus) and the Israeli bottom trawl fishery? Assessment by stomach contents and tissue stable isotopes analysis. Journal of the Marine Biological Association of the United Kingdom 94(6):1203-1220.

Sigler, M. F., C. R. Lunsford, J. M. Straley and J. B. Liddle. 2008. Sperm whale depredation of sablefish longline gear in the northeast Pacific Ocean. Marine Mammal 71. Science 24:16-27.

Sijm, D., W. Seinen and A. Opperhulzen. 1992.Life-cycle biomagnification study in fish. Environmental Science & Technology 26:2162-2174.

Smith, A. D. M., C. J. Brown, C. M. Bulman, E. A. Fulton, P. Johnson, I. C. Kaplan, H. Lozano-Montes, S. Mackinson, M. Marzloff, L. J. Shannon, Y. J. Shin and J. Tam. 2011. Impacts of Fishing Low-Trophic Level Species on Marine Ecosystems. Science 333:1147-1150.

Smith, L., R. Gamble, S. Gaichas and J. Link. 2015. Simulations to evaluate management trade-offs among marine mammal consumption needs, commercial fishing fleets and finfish biomass. Marine Ecology Progress Series 523:215-232.

Stewart, K. R., R. L. Lewison, D. C. Dunn, R. H. Bjorkland, S. Kelez, P. N. Halpin and L. B. Crowder. 2010. Characterizing Fishing Effort and Spatial Extent of Coastal Fisheries. PLOS One 5:8.

Surma, S. and T. J. Pitcher. 2015. Predicting the effects of whale population recovery on Northeast Pacific food webs and fisheries: an ecosystem modelling approach. Fisheries Oceanography 24:291-305.

Sutherland, W. J., A. S. Pullin, P. M. Dolman and T. M. Knight. 2004. The need for evidence-based conservation. Trends in Ecology & Evolution 19:305-308.

Sutherland, W. J. and C. F. R. Wordley. 2017. Evidence complacency hampers conservation. Nature Ecology & Evolution 1:1215-1216.

Tabitha C. Y. Hui, Y. M., Yumi Kobayashi, Yoko Mitani, and Kazushi Miyashita. 2017. Dietary analysis of harbour seals (Phoca vitulina) from faecal samples and overlap with fisheries in Erimo, Japan. Marine Ecology. 38(3): 1 - 12.

Veit, R. R. and N. M. Harrison. 2017. Positive Interactions among Foraging Seabirds, Marine Mammals and Fishes and Implications for Their Conservation. Frontiers in Ecology and Evolution 5(8):1-8

Whipple, S., Link, J., Garrison, L., and Fogarty, M. 2000. Models of predation and fishing mortality in aquatic ecosystems. Fish and Fisheries 1:22-40.

Williams, F. C. C. a. S. 2002. Overexploiting marine ecosystem engineers: potential

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consequences for biodiversity. Trends in Ecology and Evolution. 17(1):40-44.

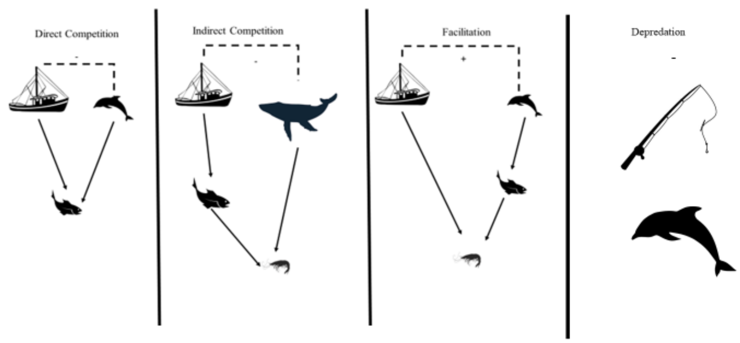

Figure 1. Conceptual figure showing direct competition, indirect competition, facilitation, and depredation. Direct competition occurs when fisheries and marine mammals share a prey species. Indirect competition can occur when a marine mammal feeds one trophic level below the fishery. Facilitation occurs when either the marine mammal or fishery releases prey from competition. Depredation occurs when marine mammals remove catch from lines or nets before they are hauled onto the ship.

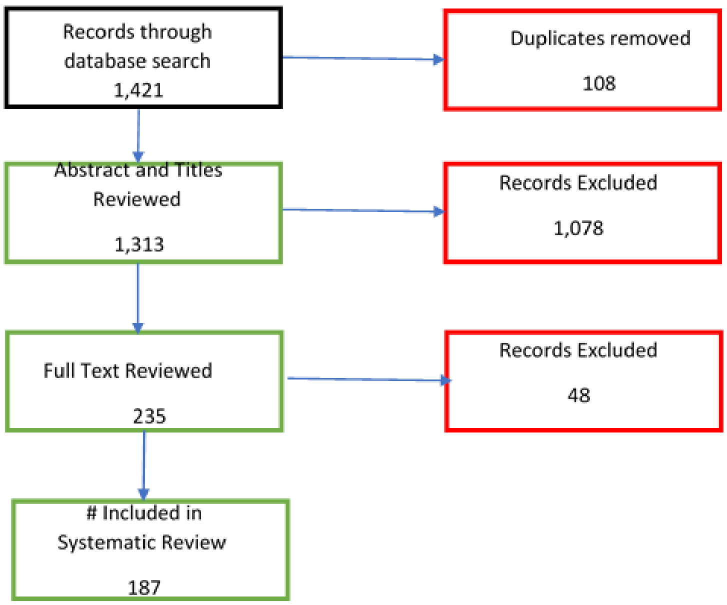

Figure 2. Conceptual figure of the search framework considered in this systematic review. Steps two and three had the same criteria for inclusion and exclusion. After three steps of screening, we were left with 114 articles for this review.

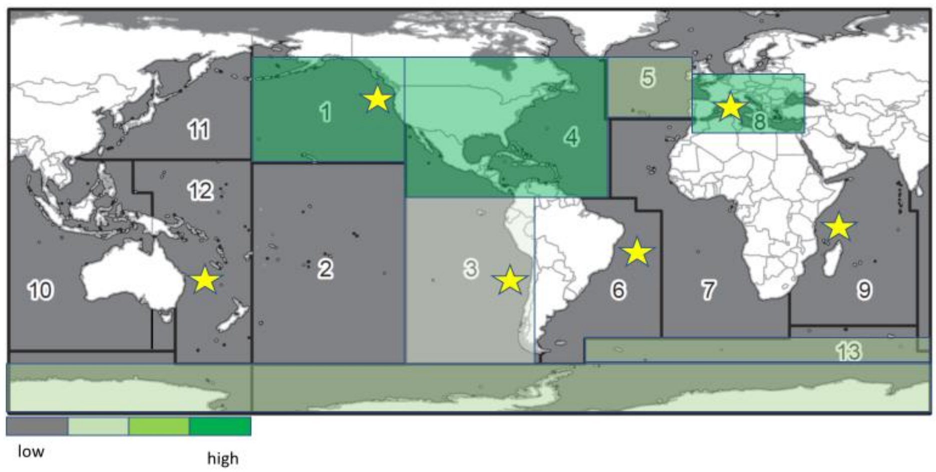

Figure 3. Map from Lewison et al. 2013 illustrating where studies occurred by ocean. 1, northeast Pacific; 2, southeast Pacific; 3, eastern Tropical Pacific; 4, northwest Atlantic/Caribbean; 5, northeast Atlantic; 6, southwest Atlantic; 7, eastern Atlantic; 8, Mediterranean; 9, Western Indian Ocean; 10, eastern Indian Ocean; 11, northwest Pacific; 12, southwest Pacific; and 13, southern Ocean/Antarctica. Grey represents areas where < 5 studies occurred, light green represents areas where more than 5-9 studies occurred, medium green represents areas where 10-19 studies occurred and dark green represents areas where ≥ 20 studies occurred. The stars represent areas of high marine mammal by-catch from Lewison et al (2013).

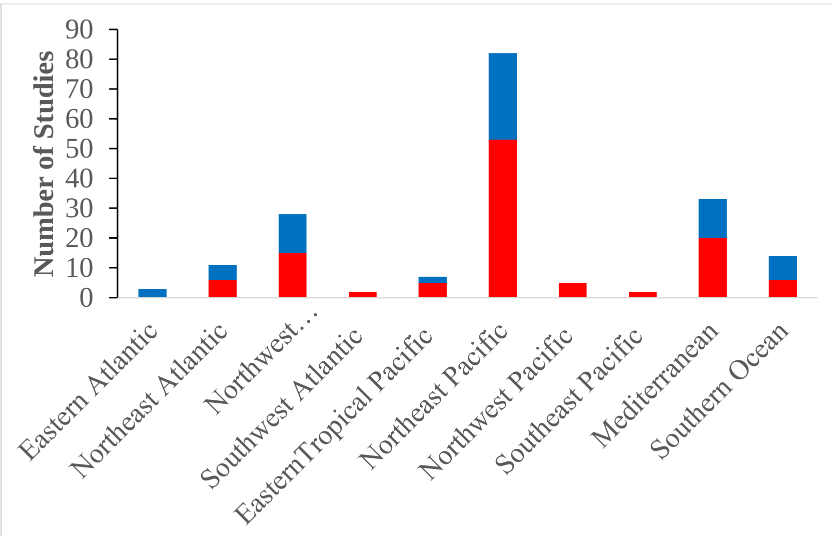

Figure 4. The number of field (red) and modelling (blue) studies and where in the world they are found.

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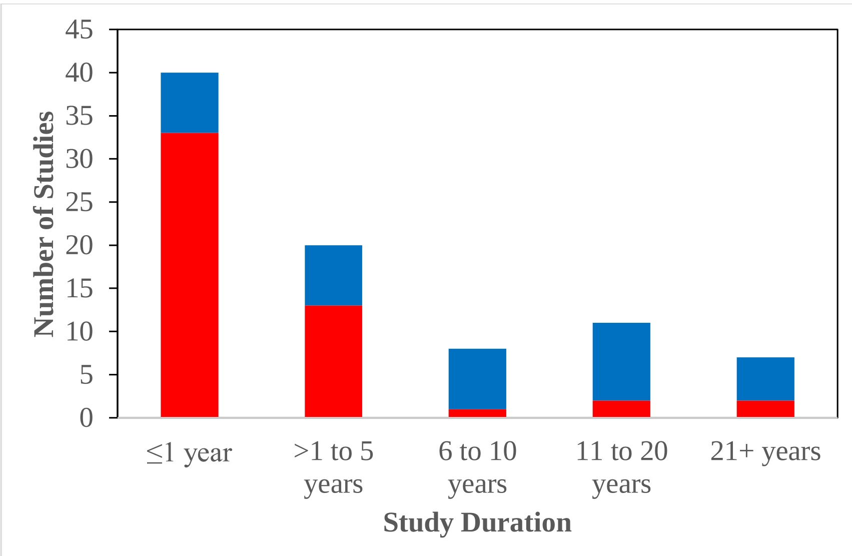


Figure 5. Study duration (in years) by field and model studies. Duration was determined from 79 of the 187 studies in our review. Blue represents model studies and red represents field studies.

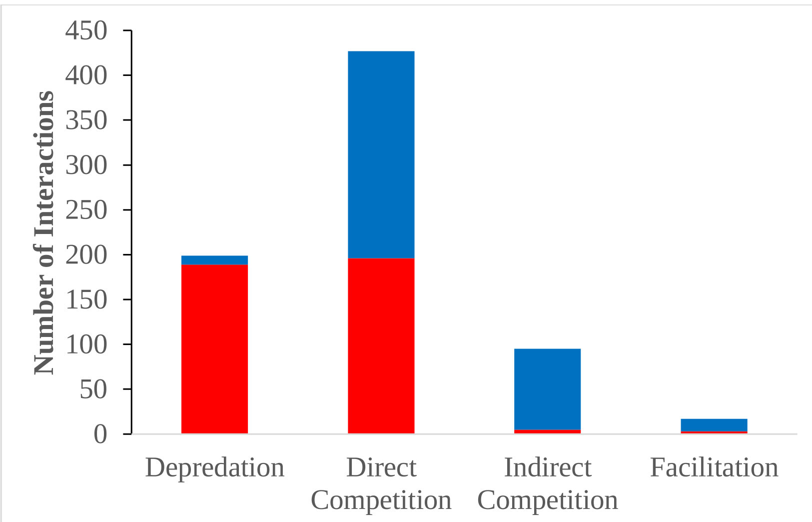
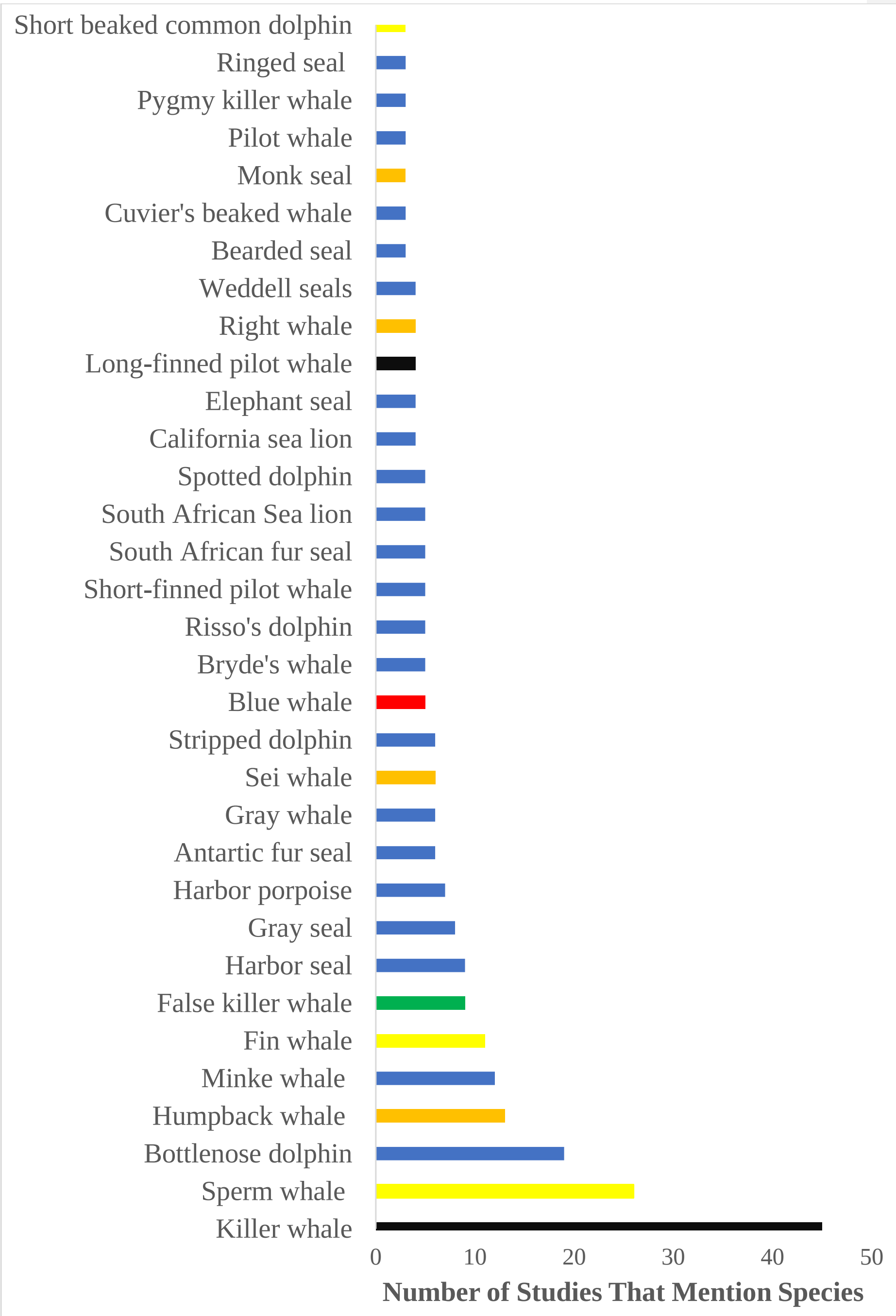


Figure 6. Frequency of interaction types across studies. The number of interactions is greater than the number of studies because a single study could contain multiple interactions. Blue represents model studies and red represents field studies.


Short beaked common dolphin
Ringed seal
Pygmy killer whale
Pilot whale
Monk seal
Cuvier's beaked whale
Bearded seal
Weddell seals
Right whale
Long-finned pilot whale
Elephant seal
California sea lion
Spotted dolphin
South African Sea lion
South African fur seal
Short-finned pilot whale
Risso's dolphin
Bryde's whale
Blue whale
Stripped dolphin
Sei whale
Gray whale
Antartic fur seal
Harbor porpoise
Gray seal
Harbor seal
False killer whale
Fin whale
Minke whale
Humpback whale
Bottlenose dolphin
Sperm whale
Killer whale
0
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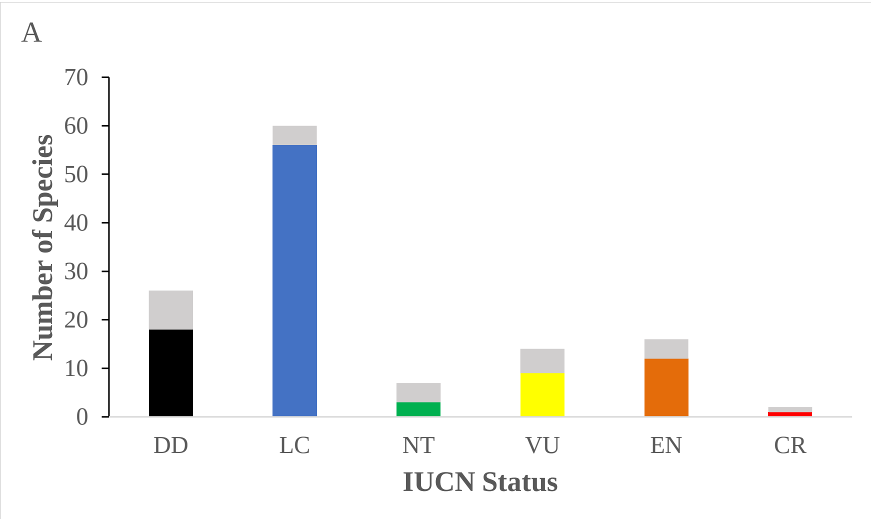
Number of Studies That Mention Species

Figure 7. The occurrence of species across studies, ordered by frequency of occurrence. Species mentioned less than three times are not shown. Colors correspond to IUCN status where black is 'data deficient', red is 'critically endangered', orange is 'endangered', yellow is 'vulnerable', green is 'near threatened' and blue is 'least concern'.



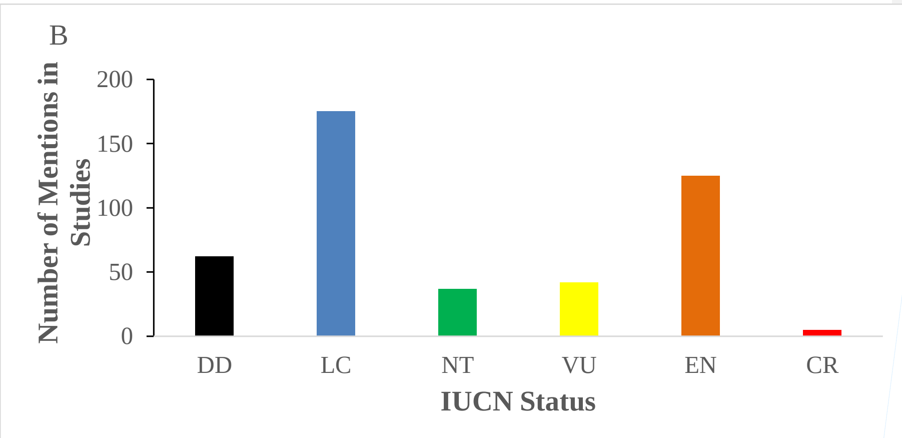


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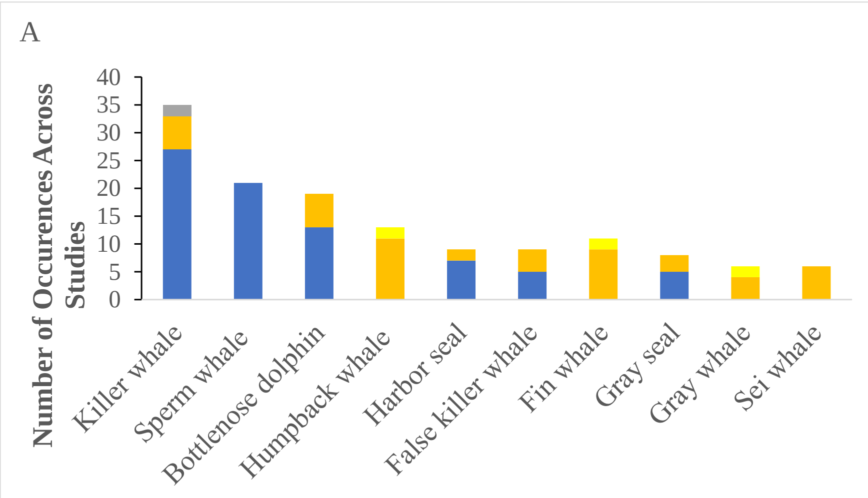
Figure 8. A) The IUCN status of species found in this systematic review. The colors correspond to the IUCN status whereas the grey region represents the total number of species within the specific category globally. B) The frequency of studies that mention

each IUCN status. The number of studies can be more than the total number of studies because studies will mention more than one marine mammal.



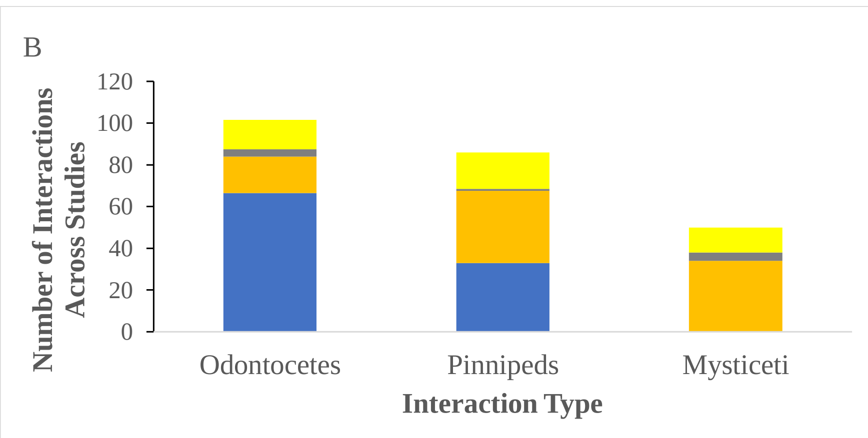


Figure 9. A) The top ten species and the kind of trophic interaction each species is involved in. Blue represents depredation, orange represents direct competition, grey represents facilitation, and yellow represents indirect competition. Bar chart B shows the order and the kind of trophic interaction each order is involved.

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Table 1. Inclusion and exclusion criteria for articles considered in this review

Criteria	Description
Marine Mammal Species	All marine mammal species are viable for inclusion.
Fishery	Artisanal or commercial fisheries were accepted.
Article Type	Modeling papers and field papers were accepted. Reviews and meta-analyses were not included.
Date	None
Language	English

Table 2. I extracted 23 unique pieces of information from each article. I provide descriptions of the data here for the systematic review.

Category	Description
A#	Individual paper ID number
Mammal species	Marine mammal species responsible for interaction
Fish species	Focal fish species considered in the interaction; can include mammals if they are hunted
Fishery	Fishery type such as long-line, trawl, seine, etc.
Site	Specific geographic location of the system in that study
Region	Country of the system in the study
Sea	Sea or ocean of the system that the study is discussing
Governmentbody	Governmental body of country of the site where the system resides
MPA	Marine protected area status
Gear	Gear type(s) used in the fishery such as circle hooks, non-lethal deterrents, nets, etc.
Length of Study	How many years the study was conducted
Study Type	Modeling paper or field paper (model = M, field = F)
Management top	Top down management style used to manage fishery (yes=1, no=0)
Management bottom	Bottom up management style used to manage fishery (yes=1, no=0)
Missing data type	If article mentioned missing data, this is the specific data mentioned (eg. foraging data, fishery data)
Suggested Gear Change	If the article mentioned gear changes to fishery this is the suggested gear change
Scale	Commercial fishery, recreational fishery (if commercial = C, if artisanal = A)
Multi Mam Species	Does the paper consider multiple species? (if yes =1, if no=0)

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Multi Fish Species	Does the paper consider multiple species? (if yes =1, if no=0)
Interaction Type	Specific fishery interaction such as 'direct competition', 'indirect competition', 'bycatch', 'facilitation' etc.
Paper Title	Title of the article
Study years	Specific years of study (eg 2002-2010)

Conclusion

At its most basic interpretation, conservation is the act of using a resource sustainably such that none of the resource goes to waste as well as the protection of species for their continued healthy existence. Conservation is not only philosophically a good practice, but has immense economic value associated with it in regards to natural resource management. In fact, this has been realized by governments all over the world and has translated into national and international natural resource policies and management. By viewing these policies and management goals through the lens of ecosystem based management, we discover a paradigm that is able to account for the tradeoffs that inherently exist among interest groups involved in conservation and natural resource use.

By studying ecosystem based management, we are able to quantify, measure, and account for tradeoffs among and between conservation initiatives. This is seen at the state level through ecosystem based fisheries management (EBFM) in the north east and mid-Atlantic United States. When a common core definition of EBFM exists among stakeholders, it becomes possible to manage state natural resources at regional scales. Although there are logistical, political, and economic restrictions in place that inhibit the implementation of certain components of EBFM, it is through studying these barriers that

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In regards to fisheries management, resources should be spent on engaging stakeholders early on in the decision-making process. Although engaging stakeholders is already a well implemented component of EBFM in the north east and mid-Atlantic regions, respondents of the survey also noted that engaging stakeholders was one of the hardest components to implement. To ease in the transition of involving stakeholders it may be beneficial to analyze the pedagogical methods used in interacting with stakeholders. Some of the barriers may be emerging from a poorly implemented way of engaging stakeholders. By examining the pedagogy of engagement, managers and scientists may find a better way forward in involving stakeholders in the EBFM process.

Through the work of this dissertation I find that ecosystem based management is a framework well suited for a wide range of socio-ecological systems. I find that tropical coral reef fish assemblages off the coast of Florida are sensitive socio-ecological feedbacks. By examining these feedbacks through an ecosystem based approach, we are able to account for the biological, social, and economic factors that contribute to the persistence of healthy ecosystems.

The compositional change in tropical fish communities influences the socio-economic aspects of the ecosystem through a series of socio-ecological feedbacks. These feedbacks can be analyzed through an ecosystem based management approach. Because EBM accounts for ecotourism, it becomes possible to link socio-economic dynamics to the ecological functioning of the ecosystem. For example, recreational divers directly interact with the environment and tropical reef fish communities which contributes to understanding the ecological baseline of a community. When the community begins to change, these changes are picked up by the recreational divers. If a baseline understanding of a health community composition is in place among the recreational diving community, changes to the community may elicit behavioral response among divers. This behavioral response may translate into conservation action to protect the healthy ecological baseline of the fish community.

Because the recreational diving industry is part of a sensitive socio-ecological system with coral reefs, one can manipulate aspects of the social and economic components of the system to affect conservation outcomes. By educating divers on the historical species compositional baseline of the reefs, one prepares a generation of divers to be sensitive to any compositional changes on the reefs. Should any compositional changes occur, well-educated divers will notice and may be inspired to take conservation action to preserve the natural species composition of the reef. By instilling a value system that reef communities are sensitive to change and are worth protecting, one can indirectly protect the local economy associated with that particular reef. For example, if a community of divers are interested in preserving the uniqueness of the reef, it ensures that the reef will continue to be a popular and unique diving experience for future generations of divers. Future generations of divers may plan extensive trips to see a unique reef community.

Finally, when we apply ecosystem based management to a global phenomenon such as marine mammal and fishery interactions, we are able to analyze large socio-ecological phenomena. By analyzing these dynamics using an ecosystem based framework, we are able to take large complex interactions and distill them into their components without losing the complexity that exists. By studying marine mammal and fishery interactions through an ecosystem based approach, we discover dynamics that permeate systems. This allows us to be able to more sustainably manage our natural resources at a global scale.

Through the work of this dissertation, I found that fisheries and marine mammals most often engage in direct competition for shared resources; however, there were a few instances of positive interaction via facilitation that also occurred. By identifying these dynamics, it becomes possible to understand the complex socio-ecological dynamics that permeate the system. It is through understanding these socio-ecological dynamics that informative policy and management can be created. For example, research should be invested in studying marine mammal and fishery dynamics that are facilitative like that of the mysticete whales fertilizing the upper layers of the ocean which facilitates more fish and by extension fishery productivity. Conservation management should aim to preserve marine mammal and fishery interactions that induce a positive feedback such as those involving facilitation. By protecting facilitation interactions and the species involved, managers are both protecting the ecological components of the ecosystem as well as the economic components.